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REPORT

Assessment of the Underground Disposal of Tailings

by

Nora M. Hutt and Kevin A. Morin
Morwijk Enterprises Ltd. and
Normar Enterprises

Prepared for
the Atomic Energy Control Board
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ASSESSMENT OF THE UNDERGROUND DISPOSAL OF TAILINGS

A report by Nora M. Hutt and Kevin A. Morin, Morwijk Enterprises Ltd. and Normar Enterprises, under contract to the Atomic Energy Control Board.

ABSTRACT

The Atomic Energy Control Board (AECB) of Canada is facing the issue of long-term disposal of uranium tailings. One option that has not been examined in sufficient detail for the AECB is the retrieval of tailings from surface impoundments and subsequent placement of those tailings in underground workings of mines.

In order to examine the issue of underground disposal in greater detail, the AECB commissioned this study. It is based on a detailed literature review and a questionnaire distributed to knowledgeable people. The literature review involved a search of 34 computerized databases and libraries, identifying thousands of potentially relevant titles. An evaluation of abstracts and retrieved documents led to a selection of approximately 250 for inclusion in this report. Forty-one people responded to the questionnaire, providing valuable references, contacts, suggestions, and most importantly their thoughts and concerns on underground disposal of tailings.

This report is structured like a catalogue of facts and information, with each paragraph presenting some concept, concern, theory, or case study involving the retrieval or placement of tailings. All relevant information, findings, interpretations, conclusions, and recommendations gathered during the course of this study are included. The Table of Contents illustrates the striking number of relevant topics and acts like a "flowchart" or checklist to ensure that an underground-disposal submission by a mining company has addressed relevant topics.

As explained in the report, the underground disposal of uranium tailings is not a reasonable alternative for many uranium mines. This stems primarily from one fact: for many mines, all tailings generated by the operation cannot be returned underground. As rock is mined and processed, it "swells" in volume so that a cubic meter, for example, of intact ore rock becomes 1.5 or more cubic meters of tailings. Therefore, some tailings would remain on the surface at many minesites. The report explains in detail the implications of disturbing surface-impounded tailings for the purpose of placing only some of the volume underground. The cumulative environmental, safety, and monetary liabilities of such a partial scheme can be discouraging in some cases.

The major conclusion drawn from this study is that partial retrieval of surface-impounded tailings and subsequent placement in underground workings is justified only if one or more key "intangible costs" are lessened significantly. Examples of such costs are long-term environmental impacts and concerns over long-term impoundment stability, which from another viewpoint could become tangible costs for future generations.

RÉSUMÉ

La Commission de contrôle de l'énergie atomique (CCEA) du Canada est aux prises avec le problème de l'évacuation à long terme des résidus d'extraction de l'uranium. Une solution, selon la CCEA, n'a pas été suffisamment étudiée, soit le retrait des résidus des réservoirs de retenue et leur enfouissement dans des mines souterraines.

La présente étude, commandée par la CCEA, a pour but d'examiner plus en détail la question du stockage souterrain. Elle est fondée sur une recherche documentaire approfondie et sur un questionnaire distribué à des personnes bien informées. Pour la recherche documentaire, on a consulté 34 bibliothèques et bases de données informatisées, ce qui a permis de relever des milliers de titres possiblement pertinents; après avoir évalué les résumés et les documents retenus, on en a choisi environ 250 pour l'étude. Quant au questionnaire, il a été rempli par 41 personnes, qui ont fourni des références intéressantes, indiqué des personnes ressources, fait des propositions et, plus important encore, exprimé leur point de vue et leurs préoccupations relativement au stockage souterrain des résidus.

Le rapport se présente comme un catalogue de faits et de renseignements; dans chaque paragraphe, on retrouve un concept, une préoccupation, une théorie ou une étude de cas concernant le retrait ou l'enfouissement de résidus. Tous les renseignements ainsi que toutes les découvertes, les interprétations, les conclusions et les recommandations recueillies au cours de l'étude y sont mentionnés. La table des matières donne une idée du très grand nombre de sujets pertinents et peut servir de diagramme de processus ou de liste de vérification pour s'assurer qu'une demande de stockage souterrain présentée par une société minière traite des sujets pertinents.

Comme il est mentionné dans le rapport, le stockage souterrain des résidus d'extraction de l'uranium ne constitue pas, pour beaucoup d'exploitants de mines d'uranium, une solution raisonnable, surtout parce qu'il leur est impossible d'enfouir sous terre tous les résidus produits. Quand le minerai est extrait et traité, son volume augmente; ainsi, un mètre cube de minerai intact, et donc compact, donne, après transformation, au moins 1,5 mètre cube de résidus en vrac. Dans de nombreux sites miniers, certains résidus doivent donc rester à la surface. Dans le rapport, on explique en détail ce qui se produit lorsqu'on retire une partie des résidus d'un réservoir de retenue pour les enfouir sous terre. Les responsabilités financières et environnementales, auxquelles s'ajoutent celles sur le plan de la sécurité, peuvent parfois s'avérer très décourageantes.

La principale conclusion qui se dégage de l'étude est que le retrait d'une partie des résidus des réservoirs de retenue et leur enfouissement ne sont justifiables que s'ils entraînent une réduction substantielle d'au moins l'un des principaux «coûts intangibles». Parmi ces coûts figurent les conséquences environnementales à long terme et les préoccupations concernant la stabilité à long terme des réservoirs, qui risquent par ailleurs de se transformer en coûts tangibles pour les générations à venir.

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ACKNOWLEDGEMENTS

We would like to thank the staff of the Atomic Energy Control Board for the opportunity to assist them in their work through this report, and at the same time learn much ourselves about the retrieval and placement of tailings. In particular, we appreciate the efforts of Dr. Rao Avadhanula, Dave Corkill, Jane Naisbitt, Michael Grace, and Marsha Scallen who assisted us beyond the call of duty during our visit to the AECB offices (and who weighed us down with dozens of reports). We also thank Norm Mailer of Atomic Energy of Canada Ltd. at Pinawa, whose efforts at compiling and sending dozens of publications to us greatly improved this report. We gratefully acknowledge the information and references provided by the staff of the Mine Environment Neutral Drainage (MEND) Program: Grant Feasby, Gilles Tremblay, and Marcia Blanchette.

1. INTRODUCTION, OBJECTIVES, AND METHODOLOGY

In 1993, the Atomic Energy Control Board (AECB) of Canada identified a need for a literature review on a particular option for uranium-tailings disposal. Due to recent mine closures in the Elliot Lake Uranium District of Ontario, reasonable options for the stable, safe disposal of uranium tailings have been identified and considered by mining companies and regulatory agencies. However, the AECB required additional details on the option of tailings disposal in mined-out underground workings. This report provides those details.

The disposal of tailings into underground workings has a basic appeal because it returns the tailings to their original location beneath the earth's surface. Such disposal would typically involve two steps: the retrieval of tailings from a surface impoundment and the placement of these tailings into the workings. There are, of course, variations on the theme. For example, operating mines which process ore underground would not necessarily have to hoist the tailings to a surface impoundment. Nevertheless, the two-step process of surface retrieval and underground placement is the main focus of this report.

With little forethought, the retrieval of tailings from a surface impoundment and placement into underground workings appear rather simple and easy. However, there are many practical and theoretical issues that can render this disposal option inappropriate for a mine. For example, disturbance of stable surface-impounded tailings in some cases could lead to greater environmental effects than would be realized by careful closure of the undisturbed impoundment.

This report examines the concept of underground disposal of tailings through a detailed literature review involving thousands of references and through a questionnaire sent to knowledgeable people. This approach provided a wealth of information on the concept, its advantages and disadvantages, and the issues that must be considered before attempting such disposal. This report presents this information, the references from which the information is drawn, and the results of the questionnaire.

In order to locate published information pertaining to retrieval of surface-impounded tailings and underground placement of tailings, a detailed computer-based search was conducted of 34 databases and libraries (Appendix A). Eight keywords were used to specifically identify the topics of interest. In total, approximately 2400 matches to the keywords were found and an additional 200 publications were located in libraries. The titles and abstracts for all matches and publications were reviewed in order to determine which would be retrieved and examined further for this report. After reviewing these documents, approximately 250 were finally used.

More than 100 copies of a questionnaire (Appendix B) were mailed internationally. Forty-one people responded, providing valuable references, contacts, suggestions, and most importantly their thoughts and concerns on underground disposal of tailings (Table 1-1). We appreciated their efforts in responding to the questionnaire.

All of this work has culminated in this report which examines the concept of underground disposal of tailings and the accompanying concerns and issues. The following chapter discusses the general problem of tailings disposal and stability, and provides a general overview of the advantages and disadvantages of underground disposal. Chapters 3 and 4 then focus on the detailed issues involved in retrieval of surface-impounded tailings and the placement of tailings into underground workings. Chapter 5 then examines a few additional issues such as environmental monitoring and permafrost.

Table 1-1
Summary Results of the Questionnaire for This Report
(see Appendix B for more details)

Issue	Section in this report	Respondents mentioning issue
General: Retrieval of tailings from a surface impoundment	3	88%
- increase or re-start of metal leaching and/or acid generation	3.1.1	35%
- degradation of water quality in tailings pond and effluent	3.1.2	25%
- change in water flows and volumes (hydrology and hydrogeology)	3.1.3	10%
- degradation of groundwater quality	3.1.4	20%
- change in geotechnical stability of dams and containment structures	3.1.5	15%
- degradation of air quality	3.1.6	13%
- method of excavation	3.2.1	13%
- reprocessing	3.2.2	3%
- transportation to underground workings	3.2.3	8%
- final cleanup and reclamation	3.2.4	15%
- costs	3.3	35%
- (no concerns over retrieval)		12%
General: Placement of tailings into underground workings	4	93%
- change in reactivity of placed tailings or wall rock during/after placement	4.1.1	23%
- degradation of water quality in placed tailings and surrounding groundwater system	4.1.2	43%
- change in groundwater flow through/from mine	4.1.3	8%
- degradation of air quality in mine	4.1.4	8%
- preparation and examination of underground workings and surrounding rock mass	4.2.1.1	8%
- maximization of tailings volume	4.2.1.2	33%
- coordination with active mining operations	4.2.1.3	5%
- detoxification of tailings	4.2.1.4	3%

Table 1-1 (continued)		
- stabilization of placed tailings	4.2.1.5	15%
- handling of water	4.2.1.6	13%
- air-phase requirements	4.2.1.7	10%
- sterilization of ore	4.2.1.8	5%
- length of time to complete placement	4.2.1.9	5%
- methods of placement	4.2.2	5%
- costs	4.3	20%
- (no concerns over placement)		7%
General: Biological implications	5.1	20%
General: Monitoring and corrections of problems	5.2	5%
General: Precedent	5.3	5%
General: Permafrost	5.4	0%

2. THE TAILINGS DILEMMA AND RELATED CONCERNS

2.1 The Dilemma

In 1980, 11.1% of Canada's Gross National Product (GNP) was derived from mining production and extraction of about 60 different minerals from Canadian rocks and soil (Hamel and Howieson, 1982). For uranium in particular, Canada is a major producer, supplying over 20% of the total world production of uranium as of 1982. As with any production process, waste products were inevitably generated.

Many mining operations which excavate ore from the earth often process or concentrate the ore nearby. This generates waste products that must be handled and disposed. Mining operations which grind ore and then remove the target element(s) can produce large volumes of crushed rock at sand, silt, or clay size known as "tailings". Because most uranium ores consist of less than 10% uranium compounds (often less than 1% in Elliot Lake), most of the mined ore rock is eventually disposed of as tailings after much of the uranium is extracted.

Tailings are often delivered to a surface impoundment as a slurry consisting mostly of water. Because of the saturated state of the tailings and their geotechnical properties which preclude high stacking slopes, impoundments typically cover relatively large areas. Other operational factors such as accelerated evaporation also justify the use of large surface impoundments. Nevertheless, at mine closure, surface tailings impoundments present significant problems for safety and stability. The AECB is finalizing workshop notes on this topic (SNC-Shawinigan Inc. et al., 1995). In fact, the decommissioning and reclamation of surface-impounded tailings can often represent the largest and most costly part of mine closure.

How can tailings be permanently decommissioned with maximum safety and minimal environmental and visual impact? The answer is unique to each mine, but there is a common suite of options that can be considered. One option is to retrieve the tailings from the impoundment and return them to underground workings. In fact, this is sometimes carefully performed during active mining to enhance the stability of workings, when it is commonly referred to as "backfilling" (e.g., Hassani, 1989). After mining, the objective can be simply to dispose of as much tailings volume as possible with little regard for stability.

To understand the enormous task of decommissioning surface-impounded uranium tailings, an impression of the volume of uranium tailings that has been generated is needed. The volume of uranium tailings requiring long-term management may in fact be the largest obstacle to using underground workings as primary disposal facilities (Fry, 1982), because there may well be insufficient underground volume available. Consequently, underground disposal is not a general option, but may be appropriate on a mine-by-mine basis.

SECOR Inc. (1981) indicated there were approximately 98,000,000 tons of uranium tailings in Canada. These tailings covered a land-surface area of 21 km^2.

Haw (1982) produced a map showing the location of uranium-tailings site in Canada as of 1982 (Figure 2-1). At that time, there were approximately 120,000,000 tonnes of surface-impounded uranium tailings. Similarly, Hamel and Howieson (1982) estimated a total of some 130,000,000 tons of uranium tailings present in Canada in 1982 covering about 10 km^2.

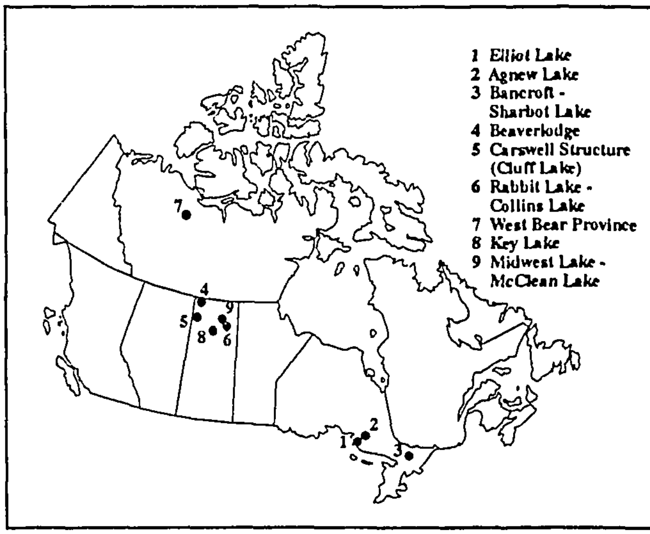

FIGURE 2-1. Canadian Uranium Tailings Sites (from Haw, 1982).

Chakravatti et al. (1982) reported that there was over 35,000,000 tonnes of uranium tailings alone in the Denison Mines tailings impoundments near Elliot Lake, Ontario. These tailings encompassed an area approximately 2.8 km long by 1 km wide.

In 1993, for the Atomic Energy Control Board of Canada compiled recent statistics on uranium tailings and estimated the total at 194,000,000 tonnes covering 1,180 ha (Table 2-1). The immense cost associated with decommissioning this volume of tailings is implied in the data.

For the United States, Thomson et al. (1986) estimated that there were 300 million metric tonnes of uranium tailings in the United States. These tailings, although considered relatively low-level radioactive waste, continue to release radon gas into the atmosphere and have other, conventional air and waterborne contaminants as their main source of concern (Thomson et al., 1986). There are similar concerns for Canadian uranium tailings.

For comparison, Rubin et al. (1981) estimated that 60,000,000 tons of coal-mine and coal-processing wastes were produced annually in the United States. An estimated surface area of 174,000 acres was in use at that time to hold these wastes.

As another international illustration of the size of the uranium-tailings disposal problem, Lloyd (1980) reported that, in the Witwatersrand area of South Africa from 1952 to 1979, 1.3×10^9 tons of uranium tailings slimes were produced. These tailings were predominantly silica with approximately 0.005% uranium and roughly 1.8% FeS_2 grading 70% at 75 μm grain size.

Uranium mining is not the only segment of the Canadian mining industry to have a large tailings management problem. Hamel and Howieson (1982) reported that there was a total of 3.4×10^9 tonnes of hard-rock mine tailings in Canada in 1980, covering 230 km^2. Saskatchewan potash mines have in excess of 250,000,000 tonnes of tailings and lesser amounts of brine stored in dyked disposal ponds (Tallin et al., 1990). Environmentally and aesthetically acceptable, economic and safe long-term disposal of these mine tailings is being called for.

Table 2-1
Locations and Quantities of Uranium Tailings in Canada
(compiled by AECB, 1993)

Area and Site	Site Status	Years of Operation	Tailings (tonnes)	Area (ha)
Elliot Lake, ONTARIO				
Stanrock	Decomm(1)	1957-1964	5,700,000	52
Spanish American	Idle(1)	1958-1959	450,000	10
Pronto	Decomm(1)	1955-1960	2,100,000	47
Agnew Lake	Decomm(2)	1977-1983	(3)	-
Quirke	Decomm(1)	1956-61; 1967-90	46,000,000	192
Panel	Decomm(1)	1958-61; 1977-90	16,000,000	123
Denison	Decomm(1)	1957-1992	63,000,000	307
Stanleigh	Operating	1958-60; 1983-	15,000,000	50
Nordic	Idle(1)	1957-1968	12,000,000	87
Lacnor	Idle(1)	1957-1960	2,700,000	25
Bancroft, ONTARIO				
Dyno	Idle(1)	1958-1960	360,000	3
Bicroft	Idle(1)	1958-1963	2,000,000	8
Faraday	Decomm(1)	1957-1962	1,800,000	16
Madawaska	Decomm(1)	1976-1982	2,000,000	15
SASKATCHEWAN				
Cluff Lake	Operating	1980-	1,500,000	47
Rabbit Lake Pit	Operating	1984-	2,400,000	8(4)
Rabbit Lake Surface	Idle(2)	1975-1985	6,500,000	53
Key Lake	Operating	1982-	2,600,000	48
Larado	Idle(1)	1957-1960	550,000	14
Gunnar	Idle(1)	1955-1964	4,400,000	75
Beaverlodge	Decomm(1)	1953-1982	6,000,000	(5)
NORTHWEST TERRITORIES				
Port Radium	Decomm(2)	1933-1960	910,000	-
Rayrock	Idle(1)	1957-1959	80,000	-
TOTAL			194,000,000	1,180

NOTES:

Decomm(1) Decommissioning activities underway; licensed by the AECB

Decomm(2) Decommissioning complete

Idle(1) Decommissioning activities underway; not licensed by the AECB

Idle(2) Under Care and Maintenance; licensed by the AECB

(3) Sludges mixed with waste rock

(4) Tailings disposal into mined open pit

(5) Tailings deposited into lake

2.2 The Primary Concerns

Many of the concerns over tailings disposal converge into four areas: costs, environmental effects, standards for disposal, and techniques for disposal. These primary concerns are introduced below.

Kegel (1975) felt most people can understand that underground mining is more expensive and more dangerous than surface mining. Therefore, the disposal of mining waste on the surface would seem to be less dangerous and less expensive than returning the wastes underground. However, in order to better characterize the expense and liabilities, a wider examination of uranium-tailings traits and behaviour is required, including "intangible" costs.

The wider examination of Canadian uranium tailings between 1982 and 1987 was directed, promoted, and funded by the National Uranium Tailings Program (NUTP). For example, the Canadian Uranium Mill Waste Disposal Technology Manual (Steffen, Robertson and Kirsten, 1987) was developed to give guidance on the maintenance of surface impoundments for long-term segregation of uranium tailings. Several of the Canadian references in this report were produced under the NUTP or are dependent on information developed under it.

During the mining process, there are disruptions of stable and pseudo-stable physical and geochemical conditions (discussed further in Section 3.1). For example, Constable and Snodgrass (1987) wrote that the mining and milling of uranium-bearing ores result in:

❶ the removal of mineral assemblages from stable reducing environments;

❷ the mobilization of some reduced minerals by leaching and oxidation reactions;

❸ the precipitation of many mobilized substances as secondary precipitates, usually on surfaces of host minerals and solids (i.e. gypsum);

❹ the disposal of waste materials into some type of tailings management area, in which aerobic conditions may dominate, especially in unsaturated zones. If water-flow pathways allow water to percolate through the tailings, mobilized substances may be transported to ground and surface waters systems (i.e. oxidation of reduced sulfides); and,

❺ the potential release of radioisotopes into the environment through atmospheric and aquatic pathways (i.e., some concerns include ^{226}Ra due to its radiotoxicity, thorium species because of their long half-life , and ^{210}Pb and ^{210}Po due to their potential for a large "dose commitment" which is a measure of the damage to the human body).

As a result, these disruptions of physical and geochemical conditions by mining often lead to tailings that are reactive and interact with the surrounding environment. If tailings are reactive and are capable of producing detrimental effects in the environment, then a mining company may have to actively limit and control the tailings' interactions with the environment. Chakravatti et al. (1982) reported that there are two main areas of interactions associated with the final close-out of uranium tailings and the long-term impacts of reclamation options:

❶ the potential for acidic drainage from the oxidation of iron-sulfide minerals within the tailings; and,

❷ the potential for the release of heavy metals and radionuclides from the tailings management system into the biosphere.

Both of these events can cause severe and sometimes irreversible damage to the surrounding environment if not dealt with adequately when choosing and/or designing a final uranium tailings disposal site.

Similar to Chakravatti et al. (1982), Bell (1989) categorized tailings into three types: (1) chemically innocuous, (2) radioactive, and (3) acid generating. It should be noted that a tailings impoundment may be both acid generating and radioactive, such as in Elliot Lake, depending on the original ore composition. Bell's (1989) summary of the environmental interactions and resulting reclamation objectives for each type of tailings are presented in Table 2-2.

Table 2-2
Objectives for Tailings Reclamation
(adapted from Bell, 1989)

Chemically Innocuous	Radioactive	Acid Generating
Compatibility with area land use	Control direct radiation	Limit oxidation
Compatibility with aesthetics of area	Control indirect radiation	Limit contaminant flushing
Long term routing of surface drainage	Control radon emissions	Control surface water seepage
Long term dust control	Control wind blown particles	Control groundwater seepage
	Control surface water seepage	Dust control
	Control surface water erosion	
	Control direct access	
	Control groundwater seepage	
	Long term containment integrity	Long term containment integrity

The objectives outlined in Table 2-2 should be maintained for at least 200 years, and up to 1,000-2,000 years "where practical" (Bell, 1989). Once these objectives have been achieved for a surface impoundment, during operation or after closure, then subsequent disturbance of the tailings could jeopardize any positive reclamation that has been accomplished. Disturbances include the retrieval of surface-impounded tailings for the purpose of placing them underground. On the other hand, if the reclamation objectives cannot be reasonably attained, then the disturbance of tailings for placement underground may provide greater benefits and long-term protection to the surrounding environment. A major disadvantage of underground placement, however, arises if a mine has insufficient volume for complete placement of tailings. In this situation, the mine is faced with closure of both tailings-filled underground workings and a residual surface impoundment.

Intangible costs associated with the release of acidic water (Scott et al., 1972), heavy metals, and/or radionuclides (Bearman, 1979) include chronic and lethal effects on terrestrial and aquatic life and their habitats (Kepford, 1977), and reduced property values along affected lakes, rivers and streams due to their undesirable condition for recreational use or permanent habitation. Today, these effects no doubt play a role in the selection of a tailings disposal options, but it is often difficult to place a monetary value on them. More obvious costs can include replacement of equipment eroded by acidic water, additional treatment costs at municipal and industrial water treatment plants, and the public, industrial and private expense associated with corrosion of bridges, culverts, boat hulls, etc. (Scott et al., 1972).

Since environmental interactions and impacts as well as costs are factors to be considered in tailings disposal, another concern becomes the degree to which tailings must be physically, geochemically, and biologically isolated from their surroundings (Rogers et al., 1982; Brookins, 1983). In other words, the "standards" according to which tailings should be disposed of and isolated are important. However, there are no standards on which all regulatory agencies, mining companies, and research scientists agree, although most groups generally agree that mine tailings can be a source of air, water (surface and ground), and soil pollution having "regional, economic, psychological and sociological impacts" (National Research Council, 1975). It is also important to remember that natural "contamination" of surface and ground waters above various standards also occurs. In areas known to contain a rich mineral content, which are obviously prime targets for mining activity, background levels may be already much higher than government regulatory levels even before mining commences. To further examine the issue, what "standards" for closing a tailings impoundment should be used in areas with pre-existing elevated levels of potential contaminants?

A study conducted by Lee and Stuebing (1990) of River Toads (*Bufo juxtasper*) near a copper mine in east Malaysia indicated that liver uptake of Ag, Cd, Cr, Co, Cu, Ni, Mn, Pb, and Zn could not be explicitly attributed to the copper mine. Toads were gathered from rivers in four locations. These sampling locations included: (1) one site which was "directly effected by discharge from the copper mine", (2) a site within 1 km of the mine "but not affected by the copper mine", (3) a wildlife reserve "known to contain mud volcanoes rich in minerals", and (4) a national park "composed principally of sandstone with no known source of heavy metal pollution". The results of the Lee and Stuebing (1990) study showed variable levels of all parameters, and correlation between the concentrations of heavy metals found in the toad livers and contamination by the copper mine seemed to be vague. The levels of some heavy metals were actually higher in areas not associated with mining activities.

The Lee and Steubing (1990) study is one of many examples that highlight the problems with identifying impacts and attributing them to mining activity. From the examples, it is clear that the local environment should be characterized prior to mining and on-going monitoring should occur during and after mining. This information can then be used by mining companies and regulatory agencies to develop site-specific "standards" for tailings disposal and mine closure. This is probably the best answer to the development of disposal standards: there are many site-specific factors at a mine which would dictate flexibility in developing any set of standards. One matter of flexibility should be on the disposal/closure technique, such as relatively impermeable covers over a surface impoundment or disposal of surface-impounded tailings into underground workings.

This section has briefly introduced the major concerns over tailings disposal, namely obvious and intangible costs, environmental effects, disposal standards, and disposal techniques. Subsequent sections of this report examine these concerns and issues in greater detail, except standards which are left to regulatory agencies and mining companies. While standards are not explicitly examined or recommended in this report, the following presentation of case studies, successes, and failures may suggest a minimum "standard practice" in the industry.

2.3 Advantages and Disadvantages of Surface Disposal of Tailings

Environmentally safe disposal and closure of tailings have been topics of international discussion for decades. Many options for safe closure of surface impoundments involve additions such as an impermeable top cover or a bentonite slurry wall around the perimeter. Despite the number of options, the primary limitation of any surface-disposal option is the long-term effects of exposure to physical and chemical weathering at or near the earth's surface. Surficial storage of tailings leaves them indefinitely susceptible to potential, often inevitable disturbances and/or dispersion (Kilborn and Beak, 1979).

Some specific problems that can be caused by weathering processes include (Down and Stocks, 1977):

❶ degraded safety and stability of tailings impoundments;

❷ water pollution, such as contamination of surface and ground waters by leachate from tailings impoundment (e.g., acidity, toxic heavy metals, dissolved salts, suspended solids, milling reagents; Table 2-3);

❸ air pollution due to wind blown tailings, as wind blown tailings can also contaminate surface waters and surrounding soil; and,

❹ difficult closure and reclamation due to the chemical nature and content of the tailings.

These and other problems have often been described in terms of "pathways" from a surface impoundment (Figure 2-2) along which, or at the end of which, environmental effects occur.

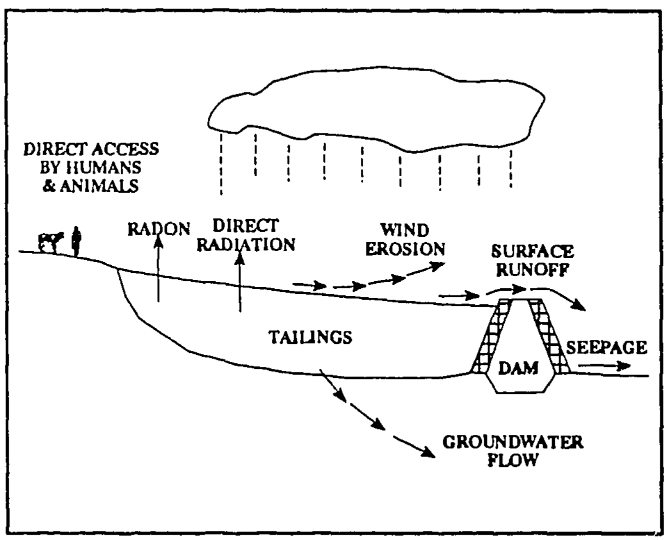

FIGURE 2-2. Potential Contaminant Exposure Pathways (from Bell, 1989)

Table 2-3
Chemical Analysis Of Effluents From Selected Tailings Areas
(from Kilborn and Beak, 1979)

	Active Tailings Areas & Seepage	Idle Tailings Areas Seepage	Idle Tailings Area Runoff	*Federal* Regulation Limits
pH	6.5 - 9.0	2.0 - 6.5	3.0 - 4.5	6.0 min.
TDS (ppm)	1,000 - 2,500	1,000 - 6,000	300 - 1,500	-
SO_4 (ppm)	600 - 1,500	600 - 3200	200 - 1,000	-
NO_3 (ppm)	100 - 200	5 - 10	5 - 10	-
NH_3 (ppm)	5 - 15	0.05 - 3	0.05 - 3	-
Ca (ppm)	250 - 450	250 - 600	50 - 350	-
Fe^{+2} (ppm)	0.1	10 - 400	5 - 20	-
Fe^{+3} (ppm)	0.1	10 - 500	5 - 20	-
Pb (ppm)	0.05	1.0	0.5	0.2
Zn (ppm)	0.1	1.0	0.5	0.5
Cu (ppm)	0.1	1.0	0.5	0.3
Mn (ppm)	0.01 - 1.0	1 - 3.5	0.5 - 1.5	-
Ni (ppm)	0.1	1.0	0.5	0.5
$^{226}Ra^{*}$ (pCi/L)	10	10 - 300	1 - 100	10
U (ppm)	1.0	1.0	1.0	-

* Dissolved

These pathways and their causes can arise (1) in the short term and remain permanently active, (2) in the short term and eventually become inactive, (3) in the long term and remain active, (4) in the long term and eventually become inactive, or (5) periodically through time. This complex scenario is usually simplified to only short-term and long-term processes. For example, Steffen, Robertson and Kristen (1986b) reported that, when dealing with surficial uranium tailings ponds, there are two types of "disruptive forces" that work towards the degradation of an impoundment and creation of pathways. The forces are:

❶ short-term disruptions such as floods, fires, earthquakes, tornadoes and glaciation causing stress on the impoundment which can exceed design criteria; and,

❷ long-term disruptions such as wind erosion, water erosion, weathering, composition, intrusion by plant roots, and intrusions by animals and/or man which can cause slower persistent degradation of the impoundment structures.

Any particular pathway can be subdivided into several related pathways. For example, the International Atomic Energy Agency (1981) developed a list of radionuclide/human-exposure pathways for releases from uranium-tailings impoundments (Figure 2-3):

❶ atmospheric pathways including inhalation of radon and its daughters, inhalation of airborne radioactive particulates, and external irradiation;

❷ atmospheric and terrestrial pathways including ingestion of contaminated foodstuffs, and external radiation; and

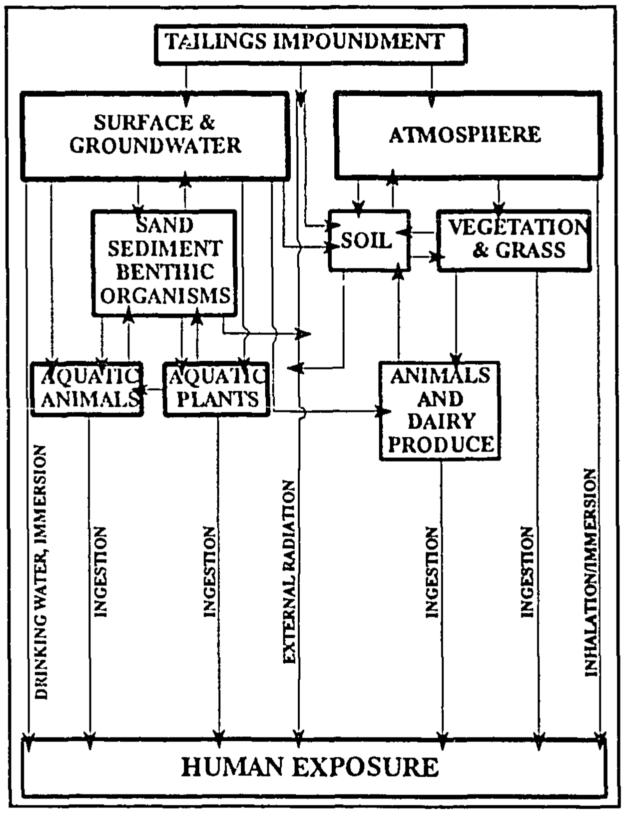

Figure 2-3. Contaminant Exposure Pathways From Uranium Tailings to Humans (from IAEA, 1981)

❸ aquatic pathways including ingestion of contaminated water, ingestion of irrigated foodstuffs, fish and other aquatic biota, and external radiation.

These exposure pathways for humans have also been discussed by many others (Bell, 1989; Haw 1982; Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects, 1986; Rogers et al., 1982; Kilborn and Beak, 1979; Down and

Stocks, 1977).

Kennedy et al. (1977) also outlined some exposure pathways for radioactive contaminants escaping from uranium tailings, and included their associated levels of radiation:

❶ airborne radon and its daughters (90%);

❷ direct gamma radiation (10%);

❸ airborne tailings dusts;

(i) direct inhalation - tailings are not airborne long enough to cause radiation exposure by inhalation,

(ii) deposition on land surface - can cause surficial contamination of lands and limit use until cleanup,

❹ waterborne (minor);

(i) solid tailings,

(ii) dissolved solids,

❺ physical removal for off-site use (usually minor).

Similarly, Ramsey (1982) cites five "mechanisms of transport" for contaminants from a tailings disposal site:

❶ wind borne erosion and transport;

❷ water borne erosion and transport;

❸ groundwater dissolution and transport in surface, perched or unsaturated and water table groundwaters;

❹ noble gas emanation; and,

❺ encroachment and/or intrusion by plants, animals, or man.

Ramsey (1982) emphasized that these mechanisms must be addressed during an assessment of the effectiveness of contaminant isolation at a permanent tailings disposal site, and case studies confirm the importance of several mechanisms (e.g., Ford, Bacon and Davis Utah, 1977).

Wind blown tailings from surface impoundments can cause serious dust problems for people living near a mine as well as for those working at the minesite. Dust control at the active and abandoned mines must be dealt with especially if the tailings are not kept under water.

Pidgeon (1982) indicated that the long-term chemical processes taking place in uranium tailings, climatic conditions and groundwater conditions will be the main factors affecting the release of contaminants from a surficial tailings impoundment. If uranium tailings were placed underground, the effect of surficial climatic conditions would not greatly affect the tailings except for potential changes in groundwater chemistry, flow and direction. These points are discussed in detail in Sections 3 and 4 of this report.

The potential for surface water and soil contamination from mining and milling operations can result in the uptake of heavy metals by plants which in turn may be eaten by animals and humans. Uranium mining and milling operations have several non-radioactive contaminants associated with their tailings. Some non-radioactive elements which may be found in uranium tailings include Si, Al, Ca, Mg, K, Na, Fe, S, C, H (as H^+) as major constituents and Cu, Ni, As, Ba, Co, Mn, Pb, Ti, Se, Sr, and Mo as minor constituents (Snodgrass et al., 1985).

In many countries, soil pollution by Cd, Zn, Pb, Cu, As, and Hg has occurred. Severe cases have been reported in Japan due to several large mining and smelting operations (Asami, 1988). Concurrently, serious public health concerns have arisen in Japan over the significant use of river water for rice-farming irrigation, which may be contaminated by heavy-metal runoff from mining and smelting operations. At the end of 1984, 126 regions of agricultural land in Japan, totalling 6,170 ha, were contaminated with Cd, Cu, and/or As above the maximum allowable limits defined by law (Asami, 1988).

The significance of any particular pathway is dependent on the site-specific conditions at a minesite, including the characteristics of the tailings, characteristics of the disposal site, climatic conditions, ground and surface water flows and movement (Haw, 1982). Local conditions and site-specific pathways can be identified through site characterizations and investigations. For example, inhalation pathways may be generally more significant in dry climates, whereas surface and groundwater pathways may be more critical in wet climates (International Atomic Energy Agency, 1981).

In any case, it may be overly optimistic to believe that the problems and pathways can be controlled through time by engineered structures and designs. Chapters 3 and 4 provide some evidence of this. Down and Stocks (1977) rather pessimistically explained:

➊ regardless of how carefully an impoundment is designed and constructed, there is always the risk of failure;

➋ properly built impoundments can be extremely expensive, especially if designed to water retention standards;

➌ impoundments are usually a major source of air and water pollution during operation and in some cases after closure. During operation, extensive water collection and treatment facilities may be needed to control the release of contaminated water into the biosphere. Continued operation of these facilities may be needed after the dam has been abandoned. The cost of operating and maintaining these facilities after closure, over the long term, may be very high;

➍ as "engineering" structures, up until recently these impoundments have seldom been designed for minimum visual impact, and can be very prominent and not aesthetically pleasing; and,

➎ if the tailings contain pyrite and other sulfide minerals, they present problems associated with oxidation resulting in acidic water, potential for contaminant seepage, difficulties with revegetation, and, if a vegetation cover is established, toxicity of vegetation to animals.

In a report for the National Uranium Tailings Program, McInnis (1985) reported that flooding caused by heavy precipitation storms, is a natural event that will probably release the greatest amount of radionuclides from Canadian uranium tailings impoundments. Heavy storms were followed, in order of decreasing level of release, by wind storms, earthquakes, drought and forest fires. Glaciation was not be ranked with the other natural events due its unpredictability. Carbon dioxide (CO_2) caused by the greenhouse effect was also not ranked because it's impacts on tailings sites were not fully known. The likelihood of partial or complete release of radionuclides into the environment at several Canadian sites was estimated by McInnis (1985) and is outlined in Table 2-4.

Table 2-4
Potential for Radionuclide Release at Several Canadian Uranium Tailings Impoundment Sites
(from McInnis, 1985)

Location	Release of Impoundment Contained Radionuclides In Years:		
	Several 100	1,000 - 2,000	10,000-100,000
Lorado	Severe (Most)	-	Severe/ Total
Gunnar	Significant	-	Severe/ Total
Elliot Lake	Significant	Severe	Severe/ Total
Bancroft	Significant	-	Severe/ Total
Key Lake	Significant	-	Severe/ Total
Rabbit Lake	Significant	-	Severe/ Total
Beaverlodge	Should not occur	-	Severe/ Total

In addition to release caused by natural catastrophes and on-going geochemical processes, the issue of dam stability is prominent. The AECB held a workshop in October 1993 to review and summarize the state-of-the-art knowledge on containment structures (SNC-Shawinigan Inc. et al., 1995). The breach or failure of a dam can have disastrous effects on human life and the surrounding environment.

There are several documented examples of dam failures. For example, a dam breach occurred on July 16, 1979 at United Nuclear Corporation's Church Rock site, New Mexico, which released approximately 94,000,000 gallons of tailings liquid and an estimated 1,100 tons of tailings solids. Other examples of tailings impoundment failures include the El Cobre district of Chile where more than 250 people were killed in March, 1965. Those and other examples are discussed in Section 3.1.5.

Hamel and Howieson (1982) noted that, as tailings are deposited in impoundments they build in height and usually have a consistency comparable with natural sands or quicksand deposits. As a result, stability of tailings as well as their containment dams are concerns, and are examined in more detail in Chapters 3 and 4.

In order to better determine if the benefits to mankind and the environment outweigh the financial costs incurred in moving surface-impounded tailings underground, the risks posed by tailings should be considered. For example, Culver et al. (1982) cited research in the Elliot Lake area showing that the radiation exposure level for persons living adjacent to a surface tailings area, prior to any reclamation, was approximately 10 mrem/a. They also noted that emissions of radon and radioactive dust reach background levels over a distance of less than 1 km from the tailings, and, since the tailings are often relatively isolated, general human exposure to radiation is frequently low. However, radiation exposure of surrounding vegetation and wildlife would have to be evaluated separately.

In general, a common opinion on risks seems to be that direct exposure to uranium tailings over long periods of time would be required to produce significant radiation-initiated health effects (Fry, 1982). Thus risk seems to be only a long-term issue unless a catastrophic natural event or structural failure occurs. Consequently, the most significant concern over surface-impounded uranium tailings appears to be the shear volume of tailings requiring a reasonable degree of management and long-term containment.

2.4 Advantages and Disadvantages of Underground Disposal of Tailings

Tailings are typically disposed of permanently in surface impoundments, but there are several pertinent disadvantages. As a result, another possible disposal option, which is the focus of this report, is the retrieval of tailings from a surface impoundment and subsequent placement into underground workings. Like other surficial options, the concept of underground disposal has been discussed for decades. An example of this is a brief article in the Engineering Mining Journal (1978) which discussed continuing research at the Anglo American Corp. laboratories in Johannesburg, South Africa, into the feasibility of using mined out stopes as a disposal site for mill tailings. Various literature and technical reviews (e.g., Mohiuddin, 1985) have concluded that underground disposal of radioactive waste is preferred and pursued by many countries, although there is vocal opposition (e.g., Herrmann et al., 1985).

The placement of uranium mill tailings underground as a final disposal alternative would provide (Kilborn and Beak, 1979):

❶ cut-off from human access to the tailings if the underground workings are sealed after disposal, thereby eliminating some potential contaminant exposure pathways such as direct human contact, exposure through inhalation, and injection of contaminated soils;

❷ reduction in, or limited water seepage from the tailings as the seepage water would be moving through a low fractured rock groundwater system, in many cases; and,

❸ the potential for adsorption of groundwater-borne radionuclides and other contaminants onto underground rock formations (i.e., fracture surfaces) prior to returning to the biosphere.

Other advantages cited by Down and Stocks (1977) are:

❶ reduction in ground subsidence;

❷ reduction in the amount of tailings left on the surface and their associated environmental hazards; and, ❸ when used as backfill, an increase in the recovery of the ore body (e.g., pillar removal).

Also, Franklin et al. (1982a) reported that underground airborne levels of radon actually decreased after placement of coarse tailings.

Although this literature review only addresses underground placement of uranium tailings for the purpose of waste disposal, the aforementioned geotechnical advantages to this practice may be important and should be addressed by mines either operating or preparing for closure (discussed later in this section and in Section 3.1.5).

However, this disposal option is not without major difficulties (Kilborn and Beak, 1979):

❶ the "swell factor" of the tailings, which limits the volume that can be returned underground;

❷ the placement of the tailings in a stable form; and,

❸ the costs associated with reopening an abandoned mine.

For the closure plan of Denison Mines in Elliot Lake, Golder Associates in association with Senes Consultants and Cumming Cockburn (1992) wrote:

> "removal [of uranium tailings] to the Denison underground workings is not a viable, 'stand-alone' solution because the underground workings cannot hold all of the tailings. Partial removal of the tailings underground does not provide any significant environmental benefit...and would incur an enormous cost".

For Rio Algom Ltd. in Elliot Lake, Senes Consultants (1991b) reviewed underground disposal of uranium mill tailings at the Quirke and Panel Mines. The result of investigations into the underground closeout alternative was that disposal of surface-impounded tailings underground was "conceptually appealing" but the option was found to be "expensive with minimal to no environmental benefit received for the large expenditure". An intensive assessment of the issues, benefits, and costs were not presented in the documents.

During operation, the carefully designed and executed placement of tailings underground is often referred to as "backfilling". Backfilling of underground mine workings has been a common practice for many decades in most segments of the mining industry, and published papers on backfilling provided important information for Chapter 4 of this report.

Bell (1989) noted that underground disposal of tailings has been ongoing for many years in Canada for use as backfill, but the type of tailings which are placed are generally non-acid generating. The justification for this choice comes from the geotechnical requirements of stability and integrity rather than from environmental concerns, although there are counter-arguments (Section 4.1.1). In any case, tailings are sometimes mixed with cement or other compounds (Section 4.2.1.5) to increase strength. This in turn minimizes the subsidence of the rock above the mine.

As an example, Placer Dome Inc.'s Campbell Mine in Balmertown, Ontario, has a 1,300 ton backfill plant, which was built in 1987 for improved tailings/cement mixing and which recovers tailings from the milling process. Ninety percent of the lower levels are mined using a cut-and-fill method. Tailings which are not recovered go to a series of tailings disposal ponds (Werniuk, 1990).

Other examples of underground placement of uranium tailings include those at the closed Beaverlodge Mine near Uranium City, Saskatchewan (Haw, 1982). Thomson et al. (1986), in their report on the geochemical/environmental feasibility of disposing uranium tailings in underground workings, mentioned that underground disposal of uranium tailings was an extensive practice at the Grants Mineral Belt in New Mexico. The placement of uranium tailings there was primarily for ground stabilization, and has been ongoing since the early 1970's (Thomson and Heggens, 1982).

In Poland, the Legnica deposit of brown coal lies under 25 villages occupying 200 km^2.

This deposit is a potential exploitable resource, but mine collapse and land subsidence are major concerns. The optimum design for the mine involves placement of some type of fill for roof support (Kurzydlo, 1991). To stabilize the ground as mining proceeds, flotation tailings from copper mines are under consideration for use as backfill. The tailings would be combined with additives to bind the water, forming a self-hardening fill.

When no backfill is used, Yamaguchi and Yamatoni (1983) noted that surface subsidence would be only of a moderate amount when underground caving advanced slowly in small increments (Figure 2-4), with few, if any, cracks visible in the surface soil. If the caving did not progress slowly but became "hung up" in an area for a period of time, then there could be a violent plug-like fall or "rock bursts", resulting in wide, deep cracks at the surface. While this concern of surface subsidence is warranted in relatively shallow mines, collapse of deeper workings may have no effect on the surface. Nevertheless, significant collapse in deeper workings can still open new fracture pathways through which surface and ground waters can flow and create new hydraulic connections (Thomson et al. 1986). For isolation of the environment from the mine workings and from any tailings that may be placed in them after closure, the formation or enhancement of aqueous pathways by subsidence should be minimized (Section 4.1.3).

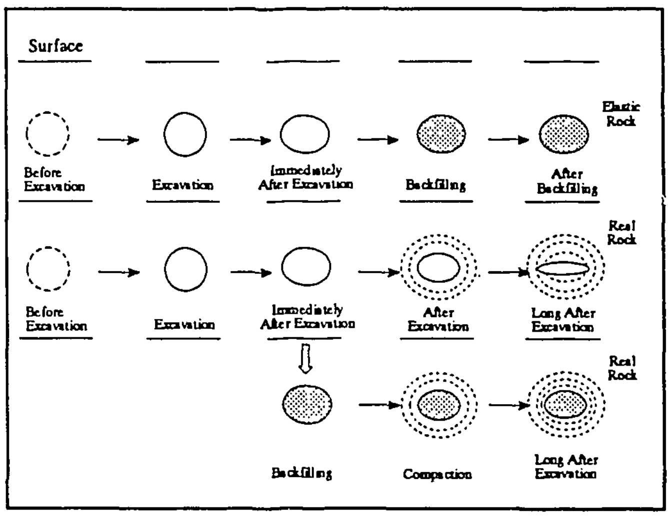

FIGURE 2-4. Effect of Subsidence and Compaction on Underground Openings (from Yamaguchi and Yamatoni, 1983).

Because of inflow of groundwater, most mines will flood to some extent when active dewatering ceases. The impact of flooding on the stability of underground structures and any placed tailings should be evaluated (Section 4.2.1.5). For example, Golder Associates Ltd.

(1991a) evaluated three mining operations in Elliot Lake, Ontario owned by Rio Algom Ltd.: Quirke 1, Quirke 2, and Panel. Golder (1991a) identified land areas which should only be used for mining activities, due to the potential for underground rock bursts, fracture formation to the surface, and subsidence over the long term.

Ackman (1982) sites several advantages and disadvantages for both underground and surface disposal of coal-mine sludge from the treatment of acidic rock drainage (ARD). These advantages and disadvantages can be extended to tailings (Table 2-5) with the benefit of comments by other researchers. An inspection of Table 2-5 shows that surface disposal has relatively few advantages relative to underground disposal. These issues are discussed further in Section 3 and 4.

Table 2-5
Underground vs. Surface Disposal of Coal-Mine Sludge as An Analog for Uranium Tailings

Advantages	Disadvantages/Concerns
Underground Disposal	
Reclamation of minimal surface area[1]	Surface access to underground disposal site may be prohibited[1]
Filling of void to reduce subsidence[1]	Topography may make pipeline installation and pumping uneconomical[1]
Addition of alkaline sludge would neutralize acid water[1]	Bulkheads used to retain sludge can fail causing flooding of adjacent active workings[1]
High security potential[2]	Seepage from disposed sludge seeps into active workings increasing volume of water that must be treated[1]
Removal of problems associated with surface disposal[3]	Induced inundation would severely inhibit future mining operations[1]
Reduce seismic activity associated with subsidence[3]	Environmental impacts[1]
	Additional costs of disposal[3]
Surface Disposal	
No sludge transport[1]	Large areas effected and reclamation difficult[1]
	Seepage from surficial containment facility into surface and ground waters resulting in migration of radionuclides[1,4]
	Dam stability[1]
	Short-life expectancy of decades instead of centuries[2]
	Seismic activities can severely damage[5]
	Embankment erosion, failure, flood damage[5]
	Human intrusion[5]
	Potential acid generation from oxidation of pyrite in tailings[4]
	Nearby stream migration may under cut pond[5]
	Migration of radionuclides into the air[4]

[1] Ackman, 1982
[2] Schneider, 1988
[3] Wood, 1983
[4] Culver et al., 1982
[5] Task Committee on Low-Level Radioactive Waste Management, 1986

3. RETRIEVAL OF TAILINGS FROM A SURFACE IMPOUNDMENT

The concept of underground disposal of tailings, as examined in this report, consists of two steps: retrieval of tailings from a surface impoundment and subsequent placement into underground workings. The concerns and issues over surface-disposed tailings were introduced in Chapter 2, and this chapter further examines those concerns and issues, including the implications of residual surface tailings which cannot be placed underground due to lack of space. Chapter 4 discusses the pertinent concerns and issues for placing the tailings in underground workings.

This chapter contains three primary sections. Section 3.1 discusses in detail many of the concerns related to retrieving tailings from stable or pseudo-stable surface impoundments. In simple words, that section contains discussions of "what to watch for" and "what to possibly expect". Section 3.2 discusses technical aspects involved in the retrieval of tailings, and thus is more of a "how to" discussion. Section 3.3 on costs addresses "how much".

Nearly 90% of respondents to the questionnaire for this report (Appendix B) indicated that one or more issues under the retrieval of surface tailings were critical to the success of underground placement (Table 1-1). Consequently, an ill-conceived act of retrieving tailings could lead to overall failure of this disposal option and thus the following topics warrant careful consideration.

3.1 Chemical/Physical Characteristics of Uranium Tailings and the Disturbance of Stable/Pseudo-Stable Conditions in the Surface Impoundment

A surface impoundment of tailings approaches a state of stability, or balance, with the surrounding environment over decades or centuries. This "state of stability" includes steady, or steadily decreasing, pathways and environmental interactions (Sections 2.2 and 2.3) in a generally predictable manner. The optimum state of stability is a lack of any pathways, resulting in an environmentally inactive impoundment. However, the optimum state is rarely achieved, at least over years to centuries. Nevertheless, a generally stable or "pseudo-stable" state can sometimes be attained in a few years or decades.

In any case, the retrieval of tailings from an impoundment in steady or pseudo-steady state will likely cause major physical, geochemical, and biological upheavals. Such upheavals can result in the accelerated degradation of conditions within and around the area. The following subsections discuss this in greater detail.

3.1.1 Acceleration or re-start of metal leaching and/or acid generation

Retrieval of surface-impounded tailings can lead to the acceleration or re-start of metal leaching and/or acid generation over that of stable/pseudo-stable conditions (Section 3.1). Over one-third (35%) of the respondents to the questionnaire (Appendix B) considered this factor a major issue in the concept of underground disposal of tailings.

Weathering of tailings within an impoundment can result in the chemical breakdown of primary minerals into more stable forms within the impoundment environment. The oxidation of pyrite and other sulfide minerals, the resulting formation and mobilization of sulphur and metal products, and the associated degradation of other minerals are all examples of mineral chemical weathering (Wood, 1983). More general examples are mineral dissolution/precipitation and ion exchange which affect levels of radioactive and non-radioactive aqueous species and minerals within a mass of tailings (Sections 3.1.2 and 3.1.3). Leaching, a very general form of chemical weathering, tends to reduce the size of the tailings particles, particularly those with the more vulnerable or chemically reactive minerals. The leached products are then carried along water-related pathways (Section 2.3). In contrast to these processes which mobilize contaminants, other geochemical processes operating within a tailings mass can remove the products of sulfide oxidation and leaching from water (Dubrovsky et al., 1984) and store them as relatively stable secondary minerals (Wood, 1983). This is a key form of "stability" and "pseudo-stability" referred to in Section 3.1.

Snodgrass et al. (1985) explained that chemical kinetic and thermodynamic processes result in a slowing of acid generation as water moves downward through a tailings impoundment (Section 3.1.4). A protective surface layer of mineral precipitants and/or leached and inert tailings forms between the deeper unoxidized tailings and the atmosphere, minimizing further pyrite oxidation and leaching. This is another form of stability referred to in Section 3.1.

For retrieval of tailings from an impoundment, some methods use water to dislodge the tailings from their settled location (Section 3.2.1). The aforementioned processes which establish geochemical stability can be altered significantly with the introduction of retrieval water, particularly if its water chemistry differs from that of the impoundment's surface water (Section 3.1.2) and groundwater (Section 3.1.3). Furthermore, any surficial layer of mineral precipitants or inert tailings would likely be removed at the start of retrieval. This disruption of stability may then result in increased mobilization of contaminants (Sections 3.1.2 and 3.1.3) transported along environmental pathways (Section 2.3).

In addition to water-based retrieval methods, there are others that require little water such as simple excavation by shovel (Section 3.2.1.1). However, this can cause the introduction of air into the tailings, the exposure of additional tailings to the atmosphere, or remove an inert surficial layer, which in turn can accelerate or restart leaching and/or acid generation.

In addition to contaminants placed in tailings during the milling process and from the original ore rock, the disposal of other mining waste materials into tailings impoundments may alter the expected chemical characteristics of resulting seepage. One important waste from the perspective of geochemical reactivity is sludge from treatment plants. The implications of this sludge are discussed below.

MacDonald et al. (1989), in a paper on Canadian mineral industry effluent treatment sludges, reported that 42 effluent treatment facilities were in use: 28 by base metal mines, 8 by precious metal mines, and 6 by uranium mines. They estimated that 140,000 dry tonnes/year of treatment sludge were produced by these facilities, with disposal usually being at the minesite into tailings impoundments or sludge ponds.

The chemical composition of sludge from treatment ponds can vary considerably depending on the initial chemical composition of the water, the type of treatment process and the added reagents (Ackman, 1982). Disposal of the sludge can depend mostly on its physical characteristics such as settling behavior and the final volume. Depending on reagents and the process, the sludge may range from "granular dense sludge" to "gelatinous voluminous flocs" (Ackman, 1982). Sludge which is disposed into tailings impoundments can affect the physical and chemical characteristics of the overall tailings mass to be relocated underground.

Through analyses of sixteen representative industry sludges, Ca, Fe, and Zn were identified as the main elements with Cu, Pb, Ni, Al, Mg, Mn, Bo, As, Cd, and Sb present in lesser concentrations (MacDonald et al., 1989). During leaching studies, contaminant releases from the sludge were found to be related to pH and the initial concentration of the metals, and this is an important observation on reactivity of tailings. The metals producing the highest releases were Zn (5,200 mg/L), Fe (770 mg/L), and Cu (93 mg/L) (MacDonald et al., 1989).

In the uranium industry, water which is subjected to treatment includes acidic mine drainage and tailings decant, with the addition of lime as the predominant treatment reagent method. Barium chloride is added to precipitate radium, and in some cases polymer addition is used as settling aids where mechanical solid/liquid separation is practiced instead of using a settling pond (MacDonald et al., 1989). The sludge containing the concentrated contaminants and added reagents may then be placed in a tailings impoundment.

Busse (1974) discussed a water collection and treatment system implemented at Heath Steele, New Brunswick. Contaminated water was entering the Northwest Miramichi River which is a major salmon spawning river. Busse (1974) described the use of hydrated lime to precipitate the dissolved metals after pumping to the tailings impoundment area, causing metal hydrates to form and remain as reactive waste in the impoundment.

Ackman (1982) reports that the chemical composition of coal mine ARD sludge was generally composed of hydrated ferrous or ferric oxides, gypsum, hydrated aluminum oxide, varying amounts of sulfate, calcium carbonate, bicarbonate. Traces of silica, phosphate, manganese, titanium, copper, and zinc were present, but "highly variable and nonuniform". Ferric hydroxide, the main component of most coal ARD sludges, is usually responsible for the poor compaction of most sludges due to its hydrous nature and electrostatic charge (Ackman, 1982). Formation of ferric hydroxide begins at approximately pH 4 and above through the oxidation of ferrous hydroxide, or the oxidation and/or hydrolysis of ferrous iron (Ackman, 1982).

At this point, a more detailed examination is warranted of potential radioactive and non-radioactive contaminants that can occur in tailings.

3.1.1.1 Radioactive contaminants

Sethness and Holmes (1979) reported that the radioactivity held in uranium mill tailings is 85-97% of the original radioactivity of the ore. Down and Stocks (1976) also found that tailings from uranium mines in Colorado contain up to 70% of the radioactivity of the original ore in the form of residual radium and other decay products, most notably thorium. While this radioactivity may be released relatively slowly from an impoundment in steady state (Section

3.1), retrieval of tailings from such an impoundment can accelerate the release by exposing more tailings to physical and chemical weathering. However, the release of radioactivity is complex due to the myriad chemical elements and their isotopes, as illustrated below.

The Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects (1986) indicated that there are three principal sources of external gamma radiation from uranium tailings: ^{226}Ra, ^{214}Pb, and ^{214}Bi. Many researchers echo the concern over ^{226}Ra and refer to it as potentially the most harmful radioactive parameter because it emits highly penetrating gamma radiation and produces ^{222}Rn gas (Down and Stocks, 1976; Rogers, 1978b). Radon gas is an alpha emitter, can migrate along some pathways faster than most other radioactive contaminants, and decays to ^{214}Pb and ^{214}Bi. Radon is very soluble in water and can be easily inhaled.

The major radionuclide contaminants of concern when dealing with leaching of uranium mill tailings are those of the ^{238}U decay series isotopes (Table 3-1), principally ^{238}U, ^{234}U, ^{230}Th, ^{226}Ra, ^{210}Pb, ^{210}Po (Snodgrass et al., 1982), ^{222}Rn, and daughters of ^{222}Rn (Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects, 1986). The isotopes of the ^{232}Th decay series isotopes also receive attention, including ^{232}Th, ^{228}Ra, and ^{228}Th (Snodgrass et al., 1982). The third decay series of ^{235}U often receives little attention in studies of tailings leaching, but detectible levels of ^{227}Ra and ^{227}Ac, for example, have been reported (Morin et al., 1988b).

As a non-metallic tailings example, Bloomfield (1984) reported on the concentrations of radium in phosphogypsum tailings taken from samples collected from approximately 305 m of drill core obtained from nine Florida tailings sites. The concentrations ranged from 8 to 38.0 pCi/g (average of 20.2 pCi/g) with measured pH from 2.10 to 3.35 (average of 2.57). These levels could degrade the quality of nearby surface waters and underlying groundwater systems if leached and transported.

Although these radionuclides are commonly found at very low levels in the earth's crust, they are usually concentrated in uranium ores and thus in their tailings. The aqueous and solid-phase concentrations of many of these radionuclides will not decrease considerably over a substantial period of time through decay due to their long half lives. However, there are exceptions. ^{222}Rn has a half life of approximately 4 days, is a noble gas, and is quite mobile in the environment. As mentioned earlier, ^{222}Rn can be considered a significant environmental concern/hazard due to the large concentrations usually present (Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects, 1986). Such large concentrations, despite a short half life, indicates parent nuclides are constantly decaying to radon gas (and its daughters) and highlight the previous statement that migration of radioactive contaminants can be complex.

Table 3-1
Range of Radioactivity Levels at Selected Uranium Mines/Mills
(from Kilborn and Beak, 1979)

Source		pH	^{226}Ra (pCi/L)	^{230}Th & ^{232}Th (pCi/L)	Uranium (mg/L)	^{210}Pb (pCi/L)
Mine Water	(F)	2 - 4	7 - 170	5 - 17,000	0.1	--
Surface Runoff	(F)	2 - 4	15	7,000	3	--
Seepage	(F)	2 - 4	1 - 12	3 - 10,000	1 - 8	--
Mill Waste to Neutralization	(F)		2 - 30,000	70,000 - 360,000	1 - 4,300	
Neutralized Tailings	(F)	6 - 9	200 - 800	20 - 40	1	3 - 7
Discharge to Receiving Water	(F)	6 - 9	2 - 8	10 - 40	0.15	2 - 8
	(UF)	6 - 9	50 - 200	10 - 100	--	3 - 10

(F) Filtered using a 0.45 μm or 3.0 μm filter
(UF) Unfiltered

3.1.1.2 Non-radioactive contaminants

Hamel and Howieson (1982) felt that the most readily noticeable environmental damage from tailings is due to acid rock drainage (ARD) from sulfide-mineral oxidation. The sulfide minerals oxidize and can produce a pH of 4 or less in adjacent waters, with pH values to roughly -1 (negative 1) reported in the literature (Alpers and Nordstrom, 1991). This acidic water is capable of leaching various metals from tailings solids which are carried into surface waters. It is the disturbance of this process from quiescent or low levels to accelerated levels that represents a major disadvantage in retrieving stable and pseudo-stable tailings. This is discussed further in Sections 3.1.2 and 3.1.4.

The potential for surface water and soil contamination from mining and milling operations can result in the uptake of metals by plants which in turn may be eaten by wildlife and humans. Uranium mining and milling operations have several non-radioactive contaminants associated with their tailings. Some potential non-radioactive elements which may be found in uranium tailings include Si, Al, Ca, Mg, K, Na, Fe, S, C, H as major constituents and Cu, Ni, As, Ba, Co, Mn, Pb, Ti, Se, Sr, and Mo as minor constituents (Snodgrass et al., 1985), as well as acidity and Ni (Snodgrass et al., 1989; Kilborn and Beak, 1979), and ammonia, nitrate/nitrite, organic complexes, sulfate, Be, Cd, Cr, Cu, CN, P, V, Zn (Pidgeon, 1983), and pyrite and other sulfur-bearing minerals (Kilborn and Beak, 1979).

3.1.2 Degradation of surface-water quality in the tailings pond and effluent

The previous section (3.1.1) discussed potential contaminants that occur in tailings solids and the processes that can leach these contaminants from the solids. These leached contaminants can enter surface waters (this section) and groundwaters (Section 3.1.4). Disturbance of stable or pseudo-stable conditions through retrieval of tailings may lead to accelerated leaching and higher concentrations. One quarter (25%) of questionnaire respondents (Appendix B) considered this an important concern. However, surface-water quality can be degraded even if stable rates of geochemical processes are not disturbed. Case studies and laboratory experiments best demonstrate this point.

Kilborn and Beak (1979) discussed a leaching study conducted by Environment Canada which started in 1974 on uranium tailings from the abandoned Nordic tailings site, located near Elliot Lake, Ontario. The estimated hydraulic conductivity of the tailings was $1.8x10^{-7}$ m/s. The study found that increases in the "rate of precipitation" or percolation rate from 500 mm/a to 5,500 mm/a increased dissolved ^{226}Ra levels by approximately 19% from 107 pCi/L to 127 pCi/L. The increase in ^{226}Ra concentrations was accompanied by increased weathering of pyrite. Such increases in contaminant concentrations could call for additional control measures such as water treatment.

Bench-scale experiments on 20-year-old tailings and fresh samples from the leaching pachucas, partial neutralization tanks and final neutralization tanks from a Rio Algom mill were carried out by Constable and Snodgrass (1987). The experiments were designed to characterize tailings leachability. Initial leaching resulted in dissolution of gypsum and minor amounts of carbonates, which otherwise might have remained stable, causing elevated levels of conductivity, total dissolved solids and pH in the leachate. Exposure of pyrite to oxygenated water resulted in pyrite oxidation and "colonization of the tailings by pyrite oxidizers", in addition to the dissolution of carbonate minerals and a rapid decrease in pH to 2-3. Constable and Snodgrass (1987) explained that higher pH values sometimes resulted from the flux of water diluting the mass of acidity produced by pyrite oxidation. After gypsum dissolution, the amounts of sulfate and Total Dissolved Solids (TDS) produced were less in continuous-mode lysimeters than in batch-mode lysimeters at equivalent pore volumes of leachate production. Therefore, Constable and Snodgrass (1987) concluded that a continuous mode of water application appears to have slowed the rate of colonization compared to a batch mode of application, but degradation of water quality was still apparent. When most of the gypsum dissolution was completed in the tailings containing little or no pyrite, the leachate stabilized at pH 5-6 (essentially the pH of the distilled water used as a wash).

Constable and Snodgrass (1987) then determined that the mass of TDS lost during the initial leaching phase was a function of the rate of water application only, which highlights the following issue on changes in water flow (Section 3.1.3). However, during pyrite oxidation, the rate of TDS production was independent of flushing rate. This implied that once the bacteria have colonized the tailings, the pyrite oxidation rate becomes independent of the hydraulic loading (flushing) rate, providing sufficient oxygen remains available to maintain pyrite oxidation. As a result, the degradation of water quality in a pond upon retrieval of tailings could pass through at least two stages.

Constable and Snodgrass (1987) also found that thorium was lost from the tailings faster than was gypsum, suggesting that the thorium species (^{230}Th, ^{232}Th, ^{228}Th) were precipitating mainly in the final neutralization tank as $Th(OH)_{4(s)}$. This compound is only partially encapsulated in the gypsum precipitate and, when placed in the tailings pile, this substance rapidly solubilizes when the pH becomes acidic. Furthermore, these researchers also noted that ^{226}Ra release corresponded to the depletion of pyrite in the top layer of experimental tailings, suggesting that, when pyrite oxidation (and associated sulfide production) was complete, dissolution of the sulfate mineral(s) containing ^{226}Ra begins. Retrieval of such tailings after potential contaminants have formed relatively stable secondary minerals can lead to re-dissolution of the minerals and resultant elevated aqueous concentrations in ponded water.

Constable and Snodgrass (1987) concluded that the aforementioned observations and results were:

> "consistent with dissolution of gypsum from tailings after placement in a tailings pile. The leaching of Ca, Mg, SO_4 and ^{226}Ra from $(Ba,Ra)SO_4$ sludges in settling pond sediments is consistent with dissolution reactions in sediment pore water and the diffusion of these constituents into the overlying water. Leaching from sludge sediments treated with a proprietary solidification process is consistent with the dissolution of a (Na,K)-(silicate, sulfate) precipitate which can be dissolved. The subsequent appearance of Ca in leachates from both sediments and tailings treated with this solidification process

suggests that the encapsulating material dissolved sufficiently to allow the $CaSO_4$ and $MgSO_4$ contained in these wastes to start to dissolve. Thus the solidification process provides, at least initially, neutralization capacity to the tailings, but may only delay development of acidification of the sludges and tailings, if pyrite is present".

In order to quantitatively estimate the effect of disturbing a stable or pseudo-stable tailings pond environment based on the previous observations, a thorough characterization of the physical and chemical properties of the tailings is required (e.g., Table 3-2). In performing such a characterization, two factors to consider are (1) the type of uranium ore and (2) the type of milling or concentrating, which both can affect the physical and chemical nature of the resulting tailings and their adsorption and neutralization capacities. Pidgeon (1982) classified the common uranium ore types as:

❶ sedimentary basin and sandstone-type deposits;
❷ uranium in quartz pebble conglomerates;
❸ vein and similar-type deposits; and
❹ other uranium deposits such as magmatic and surficial deposits.

On the effect of milling, EBA (1992) cited that in an acid-leach method of mineral extraction the ore is usually ground to < 0.5 mm diameter, while in a alkaline leach process ore is usually ground to < 0.075 mm diameter. This can affect rates of water movement and geochemical reactions.

Additionally, the milling process also has a significant effect on the resulting tailings waste by altering the chemical, physical and mineralogical characteristics of the rock (Pidgeon, 1982). Canadian uranium producers were found to condition uranium tailings prior to discharge into a disposal facility (Pidgeon, 1982). Pidgeon (1982) found that, in Canada, uranium producers recycle a maximum amount of water and milling solutions, impound all solids, neutralize acid water with lime, oxidize "ferrous iron to ferric iron by blowing air through the

Table 3-2
Properties of Uranium Tailings
(from EBA, 1992)

	Gunnar Coarse Tailings	Gunnar Fine Tailings	Key Lake	Rabbit Lake
Depth	0 - 9	0.09	4	20
Depth to Water Table (m)	4	1.5		0
Moisture Content (%)	12.7	22.7	130 - 367	90
Plastic Limit (%)	NP	NP		
Liquid Limit (%)	NP	NP		
Specific Gravity	2.67	2.7	2.64	2.6
Dry Density (kg/m^3)	1520	1460	900 (west) 300 (east)	
Calculated Void Ratio	0.7	0.3		
Gradation				
Sand (%)	73	43	30	50
Silt (%)	23	54	S + C = 70	S + C = 70
Clay (%)	4	60		
D10 (mm)		0.03		
D30 (mm)		0.05		
D60 (mm)		0.015		
Permeability (m/s)		1.70E^{-07}	5×10^{-7}(west); 1×10^{-6}(east); 5×10^{-9}(centre layer partly frozen)	10^{-3} to 10
Coefficient of Compressibility (m^2/kN)	0.04		0.0008	
Pore Water TDS	5000	5000	4500	8200
Mineralogy				
Quartz (%)			32	
Carbonate (%)			2	
Clay (%)			23	
Gypsum (%)			26	
Other (%)			11	
Salinity (ppt)	5	5	5	8
Fines			46 % (west) 85 % (east)	

pulp", and some mines use BaCl to reduce radioactivity in the discharge water by coprecipitating ^{226}Ra. Thus it is not surprising that tailings can be complex physically and chemically, beyond the level suggested by the ore characteristics alone.

A preliminary summary on the group of elements and radionuclides present in a tailings impoundment can, in many cases, be prepared after characterization of the primary and secondary minerals present within these tailings. The major minerals identified in uranium tailings located in Elliot Lake, for example, are quartz, sericite, mica, pyrite and gypsum with minor amounts of hydrous iron oxides (secondary minerals), aluminum hydroxides (secondary minerals), chlorapatite, monazite, calcite (secondary mineral or reagent) and barite (secondary mineral) (Snodgrass et al., 1985). In comparison, the major minerals found in high-grade uranium tailings located in Saskatchewan are quartz, gypsum, clays, micas, iron oxides, graphite, metal arsenides, and possibly nickel oxides and metal arsenates with minor amounts of metal sulfides, and selenides or selenates (Snodgrass et al., 1985). Section 3.1.1 discusses the implications of the primary and secondary minerals on the geochemical stability of an impoundment.

Hamel and Howieson (1982) compiled a list of physical and chemical parameters usually of interest in Canadian tailings (Table 3-3). Along with the parameters, they provided ranges of concentrations to further show the variability that can exist among tailings impoundment. As an example of how a set of parameters can affect surface-water quality, Hamel and Howieson (1982) reviewed the degradation of water quality in the Serpent River system and adverse effects on game fish as a result of the 1960's acid seepage from uranium mine sites in the Elliot Lake area. This problem has since been corrected.

Kilborn and Beak, (1979) reported that many contaminants impounded in tailings ponds, such as those located at Elliot Lake, result from neutralization, chemical precipitation and settling processes, including sludge from treatment plants (Section 3.1.1). The contaminants include the radionuclides of the uranium and thorium decay series and potential toxic and deleterious heavy metals. Snodgrass et al. (1985) during an investigation of Ontario and Saskatchewan uranium mill tailings concluded that the processes of mineral dissolution, recrystallization, and transport (Section 3.1.3) have the greatest influence on the flux of contaminants in the tailings. They noted that, to determine if dissolution and recrystallization are kinetically or thermodynamically controlled, evaluation of the hydraulic residence time of impounded water is necessary.

If surficial tailings water becomes unacceptably contaminated, water treatment offers a possible contingency for chemical control. Bragg et al. (1982) reported on methods of ^{226}Ra and ammonia removal from effluents created during uranium milling operations. They stated that there was technology available which allowed for the successful removal of ^{226}Ra, and for the most part ammonia, to levels which would satisfy environmental, and health and safety

Table 3-3
Some Characteristics Of Tailings
(taken from Hamel & Howieson, 1982)

Chemical/Physical Property	Range
Particle Size Distribution	
Sand (%)	1 - 97
Silt (%)	0 - 96
Clay (%)	0 - 40
Available Water Storage Capacity (%)	0.35
Bulk Density (g/cm³)	0.2 - 3.1
Particle Density (g/cm³)	0.01 - 4.29
pH of Ground Water	1.8 - 9.4
Cation Exchange Capacity (meq/100 g)	0.19 - 46.5
Organic Matter (%)	0.02 - 25
Electrical Conductivity (mmhos/cm)	0.1 - 22.4
Available Elements	
P (ppm)	0.1 - 400
K (ppm)	1 - 564
Ca (ppm)	40 - 52 480
Total Analysis	
N (%)	0.001 - 0.166
S (%)	0.01 - 38.87
Fe (%)	0.4 - 56.81
Al (%)	0.1 - 8.1
As (%)	0 - 0.2
Ca (%)	0.01 - 10.95
Mg (%)	0.04 - 5.0
Na (%)	0.01 - 2.9
K (%)	0.04 - 3.32
Mn (%)	0.01 - 4.0
Si (%)	4 - 37
Cd (ppm)	2 - 280
Cr (ppm)	20 - 7000
Co (ppm)	1 - 10 000
Hg (ppm)	0.005 - 1.2
Mo (ppm)	10 - 800
Ni (ppm)	10 - 546
Pb (ppm)	0.3 - 2810
Sb (ppm)	10 - 2000
Ti (ppm)	200 - 10 000
Zn (ppm)	1 - 5000
Cu (ppm)	1 - 15 000
Ra226 (pCi/g)	0.5 - 400

standards in place in 1982. The implications of wastes from treatment plants are discussed in Section 3.1.1.

This subsection has reviewed studies which indicate the ways by which surface-water quality in and around a tailings impoundment may be degraded upon retrieval of the tailings (and even without any disturbance). The impressive variety and combinations of physical and geochemical variables does not ease the task. However, the information in this subsection appears sufficient to indicate that some degradation upon retrieval should be anticipated.

3.1.3 Change in water flows and volumes (hydrology and hydrogeology)

From a physical perspective, a serious concern over retrieval is the alteration of hydrology and hydrogeology from previous, stable or pseudo-stable conditions (Sections 3.1.1 and 3.1.2). Depending on the method of retrieval (Section 3.2.1), surface-water flows can increase significantly and increased hydraulic heads could accelerate groundwater movement. One tenth of the respondents to the questionnaire for this report (Appendix B) expressed a concern over the physical implications of increased flows. The geochemical problems associated with increased flows are addressed in other sections (Sections 3.1.1, 3.1.2, and 3.1.4).

Hamel and Howieson (1982) felt that, at least in Canada, water is the major transport pathway along which contaminants will move from a tailings impoundment. Movement of water and dissolved salts from a uranium tailings impoundment is usually greatest during the period of active tailings deposition. Once the deposited tailings have begun to stabilize and become partially drained, the overall seepage from the impoundment is lower and the flow rate is controlled by the net infiltration of precipitation into the tailings mass (Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects, 1986). Disturbance of such a stable tailings environment by dredging or hydraulic methods (Sections

3.2.1.2 and 3.2.1.3) will typically increase the volume of water within a tailings impoundment. This excess water, in addition to a change in the physical state of the tailings, can result in increased volumes of runoff as surface water and of recharge to groundwater systems in the tailings area. The change in volumes of water may be accompanied by a general change in the tailings hydrology through changing patterns of drainage and ponding.

The most prudent approach would be to define the hydrology and hydrogeology of an impoundment prior to tailings retrieval (e.g., Figure 3-1), then estimate the changes during retrieval to ensure environmental and geotechnical protection and maintenance. Throughout retrieval, predictions should be compared to monitoring results to refine longer-term predictions. Finally, predictions for post-retrieval hydrology and hydrogeology may be required. This final step could be important because the tailings often "swell" in volume as compared to the volume of underground workings from which they are taken (Section 2.4) and thus there could be a residual amount of tailings remaining in the surface impoundment.

Hydrologic predictions are not easy and can involve a great deal of conceptual and numerical modelling. There are references to assist with the predictions. For example, Nelson et al. (1986) reviewed and developed methods applicable to uranium-tailings impoundments for (1) probable maximum precipitation and flood, (2) stability of

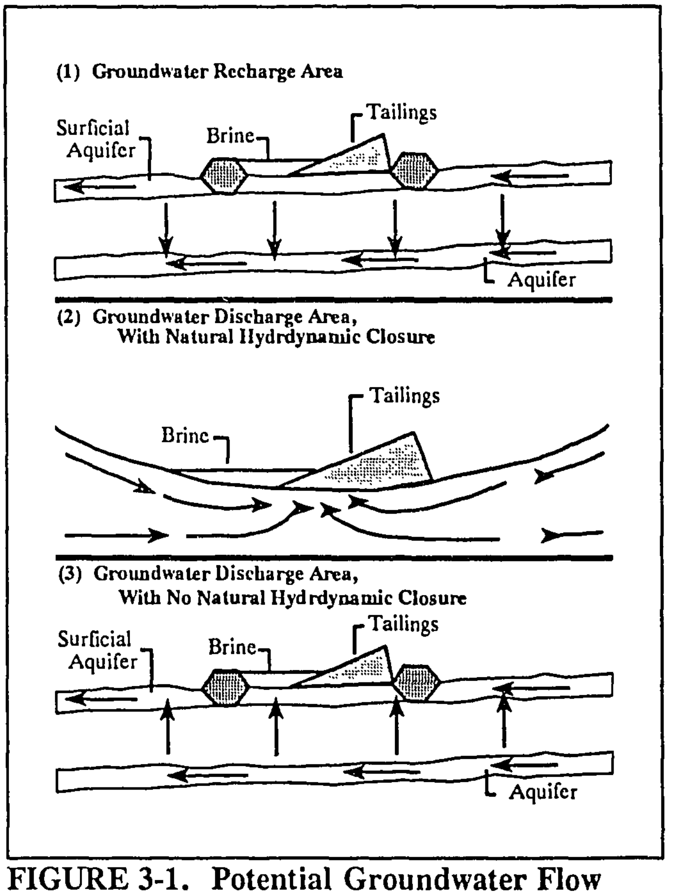

FIGURE 3-1. Potential Groundwater Flow Systems at Tailings Impoundments (from Tallin et al., 1990).

surface watercourses, (3) minimization of impoundment-surface gulley and sheet erosion, and (4) selection and sizing of riprap for embankment protection. Also, the Atomic Energy Control Board recently sponsored a workshop on the long-term stability of uranium-tailings retention structures (SNC-Shawinigan Inc. et al., 1995). If desired, hydrologic predictions can be refined further through risk-assessment techniques for uranium-mill tailings (Senes Consultants, 1986).

Hydrogeologic predictions during retrieval can be frustrated by laterally, vertically, and temporally variable degrees of saturation, a wide range of particle sizes, debris such as wooden trestles, and in some cases various layers of "caking" resulting in impervious areas (Sims, 1972). Furthermore, tailings particles generally becomes finer in size the farther from the tailings discharge pipe. This effect is caused by hydraulic deposition where coarser and/or heavier particles settled out of the slurry faster and sooner than the finer and/or lighter fraction, which can remain suspended for significant distances (Golder et al., 1992; EBA et al., 1992). In general terms, this gradation can result in marked contrasts in permeability laterally and vertically within the tailings pond. Both Chakravatti et al. (1982) and Golder et al. (1992) found such a trend within the uranium tailings situated at Denison Mines. Chakravatti et al. (1982) also reported that at Denison Mines diffusion of oxygen into coarser tailings was more extensive than in the finer-fraction areas of the tailings impoundments. The accelerated rate of oxygen diffusion resulted in an increase in acid generation from the coarse-fraction areas due to pyrite oxidation (Chakravatti et al., 1982). Increased pyrite oxidation in the coarse fraction of tailings could also be due to a higher pyrite content in this fraction. Similar findings were reported at Rio Algom Limited's closed Nordic Main impoundment (Dubrovsky et al., 1984). In any case, changes in hydrology and hydrogeology during and after retrieval should affect, and be affected by, the optimum closure strategy for the impoundment (MacLaren Plansearch Inc., 1987).

As an example, a change in groundwater conditions took place at metal-mining operations in the Coeur d'Alene Mining district of northern Idaho, upon the building and operation of tailings impoundments for previously deposited tailings (Williams, 1972). Tailings from the operations were originally spread on a valley floor, but two tailings ponds were constructed in 1968 for proper tailings management. The in-series ponds were built on a sand and gravel aquifer, and residents living down gradient of the ponds noted an increase in well water levels after their construction. The rise in well water levels suggested that the tailings ponds were a point source for inflow of water into the groundwater system. The groundwater also became acidic and contaminated with higher metals levels than were previously observed (Williams, 1972). The conditions in the groundwater system beneath the tailings ponds varied through the year. At times of high groundwater conditions, the tailings, embankment and substrata were hydraulically linked in a "saturated connected system". During periods of low groundwater conditions a "disconnected system" was present which consisted of an unsaturated zone below the tailings. Peripherally draining embankments helped control discharge of contaminated water into the groundwater system from the tailings ponds (Williams, 1972).

Tallin et al. (1990) described methods of controlling contaminant-laden surface and ground waters in and around an impoundment:

➊ Diversion ditching - control of runoff water and surficial seepage;

➋ Liners - reduction of seepage into the ground water system;

➌ Cutoff walls - "cutoff walls consist of a trench excavated through a permeable strata and backfilled with a low-permeability material, thus preventing lateral spreading of contaminants through the strata";

➍ Interceptor ditches - ditches to divert or collect seepage through a dyke or contamination moving within an aquifer;

➎ Buried drains (seepage collector drains) - a perforated pipe buried at the bottom of a trench under backfilled sand and gravel which act as a filter between natural soil and the pipe;

➏ Containment wells ("hydrodynamic containment") - "developing a ground water sink below the disposal site by pumping from one or more wells situated in an aquifer beneath the site".

These methods should be chosen on a site-specific basis, in any combination that will ensure minimization of contaminated seepage. Detailed designs for these methods are often required in the final reclamation plans of minesites with water-control and contaminant-control problem.

Nelson and McWhorter (1980) emphasized the influences that the configuration of subsurface strata beneath impoundments can have on groundwater movement. They also addressed the use of a liner to control effects on the groundwater system below an impoundment.

This subsection has illustrated the need for defining the physical hydrology and hydrogeology of a surface impoundment before, during, and perhaps after retrieval of tailings. In this way, physical changes in water flow and movement can be anticipated and combined with geochemical information to determine the potential for accelerated contaminant migration to surface and ground waters (Sections 3.1.1, 3.1.2, and 3.1.4).

3.1.4 Degradation of groundwater quality

Degradation of an underlying/adjacent groundwater system is a serious concern when dealing with the surface storage of tailings, and 20% of questionnaire respondents (Appendix B) agreed. One example (Williams, 1972) was presented at the end of the previous section (Section 3.1.3). Several processes which can degrade the quality of a groundwater system upon retrieval of tailings are similar to those for surface water (Section 3.1.2). However, there are additional processes which become more relevant for groundwater, as illustrated below.

One, more catastrophic way in which groundwater quality can be degraded beyond that caused by an impoundment is during a major spill or dam breach. A spill during retrieval could result in initial contamination of surface waters, soil, and vegetation, followed by contamination of the groundwater system. An example of the impact of spilled tailings would parallel the conditions observed in the Coeur d'Alene mining area of North Idaho, in which tailings were historically spread over valley floors as a method of disposal (Williams, 1972). This disposal method resulted in the leaching of metals from the tailings by rain and flooding of the local river and its tributaries. The outcome was the subsequent degradation of surface and groundwater systems in, and downgradient of, the mining area (Williams, 1972). A tailings spill during retrieval from an impoundment could have similar short- and potentially long-term consequences.

Osiensky et al. (1984) discussed the difficulty in monitoring groundwater and any associated contamination in fluvial environments, which can have a high degree of heterogeneity and fine layering in vertical and lateral dimensions. They found that, even after installation of a significant number of monitor wells, the delineation of the groundwater system and any contamination within the system can be very difficult. Some strata can incorrectly appear discontinuous due to their meandering nature. At a minesite where streams and creeks were diverted and former channels filled with overburden, a pseudo-fluvial type of groundwater system may develop. If tailings impoundments are built above or in the vicinity of a such a system, the assessment and monitoring of groundwater quality may yield unreliable results. In general, groundwater systems at minesites can have significant heterogeneity due to natural and artificial processes, and heterogeneity should be carefully considered and characterized.

The Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects (1986) reported that the contamination of groundwater by surface-impounded tailings is determined by:

❶ tailings composition;

❷ groundwater location, chemistry and flow rate;

❸ the chemical characteristics of the soils underlying the tailings (including retardation and adsorption); and

❹ long-term leaching of soluble tailings contaminants determined by the net infiltration of precipitation.

In order to more accurately determine the effect of radionuclide contamination on a groundwater system, Kilborn and Beak (1979) presented detailed lists of data which are required for assessment and modelling (Tables 3-4 and 3-5).

From a wider perspective, information should also be gathered on the local rock formations and geochemical conditions and processes operating naturally within the formations. For example, in a report for the Atomic Energy Control Board of Canada, Anderson (1992) reviewed literature and compiled information on naturally occurring uranium in the environment. From his literature review, Anderson (1992) found several comparison and correlations between uranium groundwater concentrations and other groundwater or surface water parameters:

❶ uranium and thorium concentration levels in groundwater are related;

❷ the amount of uranium dissolved in the groundwater is controlled by the colloidal material present in the groundwater, primarily $Al(OH)_3$, SiO_2 and $Fe(OH)_3$;

❸ the uranium concentration in groundwater is controlled by uranium mineral solubility, rather than adsorption which commonly controls trace elements;

❹ the concentration of uranium in groundwater is determined by kinetic factors controlling the redox potential and the dissolution of uranium minerals;

❺ the bicarbonate concentration in groundwater controls the solubility of uranium minerals;

❻ uranium concentration in groundwater is controlled by TDS and bicarbonate concentrations;

❼ uranium and radium concentrations in groundwater positively correlate; and,

❽ uranium solubility is controlled by the redox potential.

Table 3-4
Hydrogeologic and Geochemical Variables in Radionuclide Transport
(from Kilborn and Beak, 1979)

Hydrogeologic Properties of Porous Media	**Chemical Properties of Porous Geological Media**
Effective Porosity	Colloid Content
Intrinsic Permeability (Porous)	Type
Saturated	Distribution
Unsaturated	Surface Charge
Fractional Moisture Content	Electrical Potential
Grain Size Distribution	Cation Exchange Capacity (CEC)
Kinematic Viscosity	Selectivity Coefficient (K^{A}_{B})
Intrinsic Directional Fracture Permeability	Distribution Coefficient (K_d)
Average Fracture Spacing	Redox Potential (Eh)
Half Aperture Width	Equilibrium Constant (K_{eq})
Total Fracture Density	pH
Dispersivity	Resident Exchangeable Ions
Longitudinal	
Radial	**Geochemical Reaction Mechanisms**
Characteristic Pore Length	Chemisorption
Bulk Density	Physical Adsorption
	Electrical
	Clay Minerals
Hydrogeologic Transport Mechanisms	Organic Colloids
	Hydrous Metal Oxides
Diffusion (Fick's Law)	Precipitation
Convection (Darcy's Law)	Replacement
Average Pore Velocity (Dupuit - Forcheimer)	Isomorphic Substitution
Dispersion (Diffusion and Convection)	Coprecipitation
	Nucleation

Table 3-5
Experimental Data Requirements for Groundwater Transport Modelling
(taken from Kilborn and Beak, 1979)

Leachate Chemical Characterization

- pH
- Pollutant Form and Concentration
- Complimentary or Accompanying Ion Concentrations
- Disposal Variations
- Temperature

Soil/Tailings Characterization

- Physical
- Mineralogical
- Chemical

Chemical Interactions of Soil/Tailings Leachate Systems

- Laboratory Studies
  - Equilibrium Techniques
  - Soil/Tailings Column Techniques
- Field Prediction
  - Dynamic Column Analysis
  - Hydrodynamic Dispersion Analyses

Anderson (1992) also drew conclusions concerning uranium concentrations in natural environments:

❶ from 1286 uranium concentrations entered into a database, the range in uranium concentration was one order of magnitude larger in granite groundwaters than those reported for carbonate aquifer groundwaters. This seems appropriate as granitic rock usually contains more uranium-rich minerals;

❷ uranium concentrations in groundwaters at less than 100 m depth ranged from $10^{-11.5}$ to 10^{-4} moles/L, while at greater depth ranged from $10^{-11.5}$ to 10^{-7} moles/L, although these data were limited;

❸ although thermodynamically predicted, a good correlation between CO_3, SO_4, and PO_4 anions to formation of aqueous uranium complexes was not found. Anderson (1992) concluded that this lack of correlation may indicate uranium solubility is controlled by a combination of factors including complexing;

❹ no single "standard" groundwater was found to be representative of all the geochemical conditions observed in the granite groundwater data and, to ensure the accuracy of all work, a "standard" groundwater must be established for each site prior to any modelling and experimental work;

❺ poor correlation was found in a comparison of solubility predictions made by thermodynamic models and relationships found in the natural groundwater environments for uranium concentrations worldwide; and,

❻ the range in concentrations of naturally occurring uranium is greater than in concentrations determined through U_2O_3 leach testing. This may be due to the limited set of conditions present during the test vs. the conditions occurring in a natural system.

The preceding lists of factors and observations to this point illustrate the potential complexity in defining a groundwater system and estimating the effects of various methods of tailings retrieval (Section 3.2.1). However, the preceding work by Anderson (1992) did show the value of general observations in somewhat simplifying the estimation of concentrations. In any case, the following case studies further illustrate the complexity or simplicity, and available approaches, in estimating geochemical effects on groundwater systems.

In a core of stream sediment which had water containing approximately 4 Bq/L radium flowing over it for a few years, Markose et al. (1982) found "significant adsorption of radium in the first few centimetres of the top soil only". They also found the anions such as chloride and sulfate were not retained in the soil, but were more mobile and had reached the underlying groundwater system. During these investigations on sediment from the Juduguda Mine in India, Markose et al. (1982) determined that "activity of the sediment is inversely proportional to size", meaning that finer particle sizes displayed greater geochemical reactivity, and this also carried implications for the fine-grained tailings located at the site.

Snodgrass and Hart (1990) conducted an extensive study of ^{230}Th concentrations in the porewater of the acid-generating Lacnor tailings at Elliot Lake, Ontario. The purpose of the report was to confirm previously measured levels of ^{230}Th which were considered to be high. Through the use of detailed sampling and quality-assurance programs, both precision and accuracy of ^{230}Th measurements was ensured. Laboratory accuracy was also tested by submitting standards of known concentration, and laboratory precision was tested using duplicate-duplicate submission of samples. To assess and predict changes over time ("derived release limit"), a high level of confidence in the data was essential. Detailed recording of techniques and observations, meticulous detail to sampling and sample handling, and assurances that laboratory quality controls were maintained were all necessary.

The range of ^{230}Th found in Lacnor tailings by Snodgrass and Hart (1990) ranged from 200 Bq/L down to less than analytical detection. Snodgrass and Hart (1990) considered this range to be mainly a function of pH range and solubility effects, such as caused by pyrite oxidation which generates acidity in the unsaturated zone and the downward movement of porewater and acidity. The data obtained by Snodgrass and Hart (1990) showed that pH in the Lacnor tailings increased with depth because the acid front, which marks the base of the acidic upper portion of tailings with a porewater pH less than 2, did not extend below approximately 4 m. Depth profiles of ^{230}Th concentrations reflected this pH profile. Related geochemical studies by Constable and Snodgrass (1987) which illustrate potential effects on water chemistry upon tailings retrieval, are discussed in Section 3.1.2.

Snodgrass et al. (1985) reported on studies that indicate the acid front slows as it moves downward through a tailings impoundment, based on modelling of porewater content and chemical kinetic and thermodynamic processes that included oxygen diffusion, adsorption rates, and mineral precipitation/dissolution. The slowing rate of the acid front reflected the slowing of the acid-generation rate and subsequent depressed leaching of soluble radionuclides from tailings. This creates a layer between deeper unoxidized tailings and the atmosphere minimizing further oxidation with depth and creating a pseudo-stable system (Section 3.1). Obviously, the removal of this layer during tailings retrieval can expose unoxidized tailings to the atmosphere and/or oxygenated water, possibly leading to accelerated rates of acid generation and metal leaching affecting groundwater and surface-water quality. In this case, if all tailings are retrieved relatively quickly, no degradation of water quality would be seen. However, the definition of "relatively quickly" is unique to each site and requires predictive testwork such as acid-base accounting and kinetic tests (British Columbia AMD Task Force et al., 1989). Furthermore, if all tailings cannot be retrieved and disposed underground, the initiation of fresh, rapid acid generation in the remaining acid-generating tailings seems inevitable without active controls.

During a study of the acid-generating Waite Amulet tailings in Quebec, St-Arnaud et al. (1989) found pore water concentrations of Fe^{2+} at 1 m below the water table as high as 6000 mg/L decreasing rapidly to 50-100 mg/L at depth. Sulfate concentration were 13,000 mg/L at 2 m below the water table decreasing to 2,000 mg/L at 8.6 m depth. Acidity was 1750 mg/L at 2.96 m and decreased to 70 mg/L at 8.6 m depth. Eh (mV), specific conductivity (μS/cm), calcium, magnesium, potassium and sodium were also measured and generally tended to decrease with depth. As noted in other acid-generating tailings, pH was found to increase with depth, except for one measurement taken at 3.3 m depth which was attributed to water-table fluctuation caused by variations in infiltration. St-Arnaud et al. (1989) also reported depth-specific concentrations of heavy metals in the porewaters from one borehole (Table 3-6).

In addition to in-situ reactions, porewater quality can be affected or determined by the milling processes. Table 3-7 contains data for water collected in 1980 and 1981 at an operating New Mexico uranium mill (Thomson and Heggen, 1982).

3.1.5 Change in geotechnical stability of dams, containment structures and tailings

With many engineered surface-tailings impoundments there is a risk of a dam failure, and the potential for breach of a tailings pond. Also, Hamel and Howieson (1982) noted that as tailings are deposited in impoundments they rise in height and usually have a consistency comparable with natural sands or quicksand deposits. Thus, instability can develop in the tailings over time, which can result in a shifting of the deposit in either a slow gradual process or in an instantaneous movement. Geotechnical stability can be particularly degraded during tailings retrieval due to the additional water sometimes required (Section 3.2.1), the shifting of stress within the impoundment and its structures, and chemical degradation (Sections 3.1.2 and 3.1.4). Ten percent of respondents to this study's questionnaire (Appendix B) raised the issue of stability.

An example of a dam breach occurred on July 16, 1979 at the United Nuclear Corporation's Church Rock site in New Mexico. The resultant spill contained approximately 94 million gallons of tailings liquid and an estimated 1,100 tons of tailings solids. The tailings

Table 3-6
Pore Water Metal Concentrations at Waite Amulet, Noranda, Quebec
(from St. Arnaud et al., 1989)

Location	Piezometer No.	Depth (m)	Al (mg/L)	Cr (mg/L)	Cu (mg/L)	Mn (mg/L)	Ni (mg/L)	Pb (mg/L)	Zn (mg/L)
Waite Amulet, Noranda, Quebec	WA-11-3	2.96	< 2.5	< 0.05	< 0.02	79.30	0.11	0.36	0.19
	WA-11-2	5.15	< 2.5	< 0.05	< 0.02	3.47	< 0.05	< 0.05	< 0.01
	WA-11-1	8.16	< 2.5	< 0.05	< 0.02	0.12	< 0.05	< 0.05	< 0.01

Table 3-7
Selected Constituents from Unfiltered Samples of New Mexico Uranium Tailings Pond Water
(from Thomson and Heggen, 1982)

Constituent	Four Acid Leach Mills (14 Samples)			One Alkaline Leach Mill (5 Samples)		
	Minimum	Median	Maximum	Minimum	Median	Maximum
Gross Radioactivity (pCi/L)	3,200	38,000	73,000	3,400	6,700	10,000
Ra-226 (pCi/L)	15	70	1,800	56	58	90
As (mg/L)	0.18	1.3	5.6	2.1	5	7
Mo (mg/L)	0.20	0.90	29.5	72	98	105
Se (mg/L)	0.006	0.21	6.97	22.1	29.5	51.2
SO_4^{2-} (mg/L)	300	29,700	56,000	5,500	8,400	16,700
U (mg/L)	1.1	15	69	4.2	54	70
V (mg/L)	39	74	107	1.2	14	16
NH_3 (mg N/L)	3.3	400	3,960	1.1	16	335
TDS (mg/L)	17,900	39,800	72,800	17,000	25,400	39,700
pH	0.3	1.05	2.15	9.9	10.1	10.3

were released into an adjacent arroyo, travelled down a so-called "pipeline arroyo", into the north branch of the Rio Puerco arroyo, past the convergence of the north and south Rio Puerco, travelled through the rest of New Mexico, and then approximately 20-25 miles into the state of Arizona (Weimer et al., 1981). The initial site investigation and subsequent cleanup resulted in the chemical analysis of approximately 2,400 samples, primarily to monitor radionuclide levels in the contaminated soil during cleanup (Weimer et al., 1981).

Other examples of tailings impoundment failures include these cases:

❶ in March of 1965, ten of fourteen tailings dams in the El Cobre district of Chile failed. Over 250 people were killed as tailings flowed down an adjacent valley. The diaster resulted from the failure of dam walls and the liquefaction of tailings caused by an earthquake in the area (Down and Stocks, 1976);

❷ in September of 1970, 89 miners died underground at Mufulira, Zambia, when tailings placed over mined-out areas burst through the workings. It was assumed that the tailings had sealed fissures in the hanging wall, but a breach or "sinkhole" (Neller et al.,1973) apparently formed, initiating a mudrush as tailings liquefied under dynamic stress (Down and Stocks, 1976);

❸ in November of 1974, 15 people died at Impala Platinum Mine in the Transvaal when they were engulfed by a mud slide of tailings. It was assumed that a breach developed in a tailings dam retaining wall following a period of heavy rainfall (Down and Stocks, 1976);

❹ in 1972, a refuse pond dam failed at a coal mine in Buffalo Creek, West Virginia (Popovich and Adam, 1985);

❺ Wood (1983) discussed a coal refuse dam failure at Buffalo Creek, West Virginia in 1972. The dam failure resulted in a release of water and sludge that flooded and demolished 1,500 homes and killed 150 people.

In addition to containment structures and the tailings, the stability of other features should be addressed. For example, Peterson et al. (1982) identified tailings impoundment liner failure as another potential environmental problem. They outlined several methods of determining the appropriate liner for a given mine. Popovich and Adam (1985) also identified the potential for contamination of surface waters such as streams, and the potential for unstable or burning embankments (coal refuse or high sulfide content refuse) as problems with the use of surface impoundments.

Tailings placement into an impoundment is not a uniform, homogeneous deposition with respect to particle size (Snodgrass et al., 1982; Bean, 1972) or mineral composition (discussed in Sections 3.1.1 and 3.1.3). As tailings are pumped into the impoundment, the heavier, larger and/or denser particles settle out quickly near the discharge point. This ongoing segregation of particles during episodes of deposition and mill shutdown as well as episodes of higher and lower rates of milling results in the distinct layering of tailings materials and minerals. As a result, retrieval of the tailings (Section 3.2.1) can be more difficult. More importantly, this complex stratification could lead to unexpected movement and migration of tailings solids. This in turn could endanger containment structures, such as through the shifting of tailings against a dam, the unexpected removal of tailings against an upstream-type dam, and the plugging of a gravity-drain (flow-through) dam with fine tailings.

The stability of containment structures are now routinely considered in the closing of a tailings impoundment, and also should be clearly considered during planning of tailings retrieval. The AECB held a workshop with Canadian experts on stability of these structures in October of 1993, and the notes of the workshop provide a good overview of various issues (SNC-Shawinigan Inc. et al., 1995). The longer-term issues will be important in the event that some tailings must remain in the surface impoundment due to the lack of underground storage volume (Section 2.4).

As an example of stability issues that can arise during adjustments of hydrologic and hydrogeologic conditions, the Quirke and Panel uranium tailings basins in Elliot Lake were examined by King and MacPhie (1991) through the "saturated tailings concept". The changes in the character and stability of the tailings basin were discussed and modelled with regards to increased water being pumped to the basins. A water balance model was also prepared.

In a report for the Canadian National Uranium Tailings Program, Steffen, Robertson and Kristen (SRK, 1986a) reviewed three generic uranium tailings sites in Canada: Elliot Lake tailings, Saskatchewan neutral tailings from low/medium grade ores, and Saskatchewan nickel-arsenic-clay tailings from high grade ores. A list of the main long-term concerns was provided regarding stability and the potential for contaminant releases from a tailings impoundment (Table 3-8).

The liquefaction potential of uranium tailings themselves should be assessed when determining the optimum type of retrieval method (Section 3.2.1). Acres International Ltd. (1992) undertook a project to determine the liquefaction potential of uranium tailings at Quirke Mine tailings impoundment in Elliot Lake. Their report described the most recent techniques for modelling of tailings materials. The report also discusses the dynamic behavior of soil grains with respect to the two physical processes of volumetric compaction and pore pressure changes, and the movement generated by earthquakes which can cause "slip at grain to grain contacts in the soil" [tailings]. The movement of particles within dry soils may cause volumetric compaction increasing "inter-grain stresses". In wet soils compaction is resisted by the pore water, and inter-grain stresses are transferred to the pore water increasing porewater pressures.

Liquefaction can occur when there is a "complete loss of shearing resistance in the soil" such as when pore pressures increase to a level equal to the inter-grain stresses (Acres International, 1992). The tendency of liquefaction to occur within a soil is dependent on the cohesion between grains and the possibility of slip, resulting in volumetric compaction.

Table 3-8
Effects Impacting the Long Term Stability of a Tailings Impoundment
(from SRK, 1986b)

Embankments
- sheet erosion can be controlled by a forest cover
- gully erosion; a forest cover may aid in preventing extensive damage
- flood erosion; a forest cover may aid in preventing extensive damage
- wind erosion can be controlled by a forest cover
- leaching
- salt migration can be controlled by a forest cover
- drain clogging from chemical precipitation, root penetration, seepage erosion, frost action, and/or covering of drain outlets
- frost action can cause heave of diversion and drainage structures, frost cycling can induce soil creep on steep slopes, but this can be controlled by a forest cover
- seepage erosion (or piping)
- physical instability such as slip surface failure
- infiltration can be controlled by a forest cover

Diversion works
- floods
- gully erosion
- ice and debris blockage
- sedimentation

Surface water quality
- impoundment surface runoff
- seepage
- acid generation in tailings water

Acres International (1992) suggested that a soil's plasticity index can serve as a general indicator of soil cohesiveness and resistance to liquefaction. For example, silts and clays have a *high degree of plasticity and can "undergo permanent deformation"*. Experiments conducted on tailings containing notable amounts of fine grained micas showed that they were more resistant to liquefaction because of their inherent cohesiveness. These data imply that an assessment of tailings' ability to liquefy must be conducted on a case by case basis and include assessments of physical characteristics, such as grain size and particle shape, and chemical characteristics such as mineral and elemental composition. Acres International (1992) compiled a classification system for various soil dynamic analysis methods (Table 3-9).

The conclusions made by Acres International (1992) with regards to their modelling of Quirke Mine tailings were for only a selective area of the tailings. They noted that other areas of the pond could be quite different due to the heterogeneity of tailings distribution.

Table 3-9
Classification of Soil Dynamics Analysis Methods
(from Acres International Ltd., 1992)

Total Stress Analysis
- Equivalent linear soil method
- Nonlinear soil method

Effective Stress Analysis
- Uncoupled - pore pressures generated separately
  - linear soil method
  - nonlinear soil method
- Coupled - pore pressures generated simultaneously
  - linear soil method
  - nonlinear soil method

There is a geochemical process sometimes operative in tailings that can be viewed as the opposite of liquefaction. Wood (1983) described "lithification" as resulting from the compaction of tailings under their own weight. Lithification can also happen when there is cementation of tailings particles by the deposition of secondary minerals from solution (Section 3.1.1). These processes can increase the in-situ strength of the tailings. The increase in strength will add stability to the impoundment, but in extreme cases could also mean that retrieval procedures would be more difficult and expensive (Wood, 1983).

3.1.6 Degradation of air quality

Although many of the previous subsections discussed the water phase, the degradation of the air phase during tailings retrieval is also a concern and thirteen percent of respondents to this study's questionnaire (Appendix B) agreed. The air phase forms an important pathway for radioactive and non-radioactive contaminants (Section 2.3), both as a direct carrier and as a carrier of solid-phase particles containing contaminants. For example, Kilborn and Beak (1979) wrote that "inhalation of radon gas and radon daughters is probably the most significant airborne pathway to human exposure".

Degradation of air quality in the vicinity of a surface tailings impoundment is a function not only of the physical and chemical ("internal") characteristics of the tailings themselves, but also of external factors such as the surrounding topography, climatic conditions, and the surface wind system (Kilborn and Beak, 1979). Many of these factors are discussed below.

Some physical and chemical factors which can affect the rate of radon migration and emanation include mineralogy, radium concentration, moisture content, air content, porosity, and particle size (Kilborn and Beak, 1979). Radon emanation from tailings begins when radon gas is produced in a tailings mass through radioactive decay of parent isotopes and then enters the interstitial spaces between tailings particles. The radon gas then diffuses or advects through the tailings mass until it either decays or reaches the tailings/atmosphere interface where it escapes into the air (Kilborn and Beak, 1979). Therefore, any factor which affects the open void spaces between tailings particles (Sections 3.1.1 and 3.1.4) affects the pathways for radon gas transport and thus release to the atmosphere.

Additionally, the concentration of parent isotopes affect the emanation of radon. For example, in a report for the AECB, Kilborn and Beak (1979) presented a simple relationship for radon flux:

$$\text{Radon Flux (pCi/m}^2\text{/sec)} = 1.6 * \text{concentration } {}^{226}\text{Ra (pCi/g)} \tag{3-1}$$

All the aforementioned factors internal to the tailings mass affect the rate of diffusion or advection through the mass. For diffusion, the effect of the factors can be combined into a "radon diffusion coefficient" (Table 3-10).

External factors can enhance or depress the aforementioned emanation rates derived from internal factors. Through the use of equations, Kilborn and Beak (1979) found that weather conditions can significantly alter the radon diffusion rate from tailings by reducing the radon flux

Table 3-10
Established Diffusion Coefficients, K
(from Kilborn and Beak, 1979)

Soil Type	Soil Condition	Depth	Bulk Diffusion Co-efficient, K (cm^2/sec)
Unconsolidated Glacial Debris	Moist - Matted Grass 40% Porosity	0 - 28 cm	0.02
Consolidated Sandstone	Mine Tunnel 25% Porosity	0 - 1.6 m	0.03
Alluvium (Yucca Flat)	Dry, Sandy	1 - 3 m	0.036
Alluvium	Very Dry, Powdery Sparse Ground Cover	0 - 30 cm	0.10

when precipitation and/or snow covers the tailings. Changes in temperature can increase advective activity, causing soil gas to move towards the surface. Also, wind movement can draw radon gas from tailings to a depth of 1.8 m. Changes in barometric pressure can affect reaction rates (Lloyd, 1980) and alter the radon flux by 15% higher or lower (Kilborn and Beak, 1979).

Kennedy et al. (1977) found that under controlled experimental conditions only a 10% increase in radon flux at the surface was noted over a 40°K range and that the effect caused by changes in barometric pressure were negligible over a range of 350-660 mm Hg. They found that moisture content of the tailings had the largest effect on radon flux. The specific mineralogy of the tailings can also have an effect on the radon escaping from a uranium tailings impoundment. The example was that of tailings from sandstone ores "where the mineralization generally occurs as a coating on the surface of sand grains, emanating power is essentially independent of grain size, and is about 20%". The thickness and general shape of tailings impoundment can also affect radon emanation, as well as wind velocity across the surface.

Kennedy et al. (1977) also refer to calculations done by the Oak Ridge National Laboratory which showed that the surface flux from tailings reaches about 95% of its maximum at a thickness of 2 m. The increase in surface flux is negligible as tailings thickness exceeds 3 m. With this type of information, optimum designs for tailings-retrieval programs and residual-tailings impoundments can be designed to maximize air quality.

Kilborn and Beak (1979) reported that the average radon emanation rate from tailings in the Elliot Lake area ranged from 6 to 1,905 pCi/m^2/s, whereas for tailings in the United States they reported the range at 10 to 1,400 pCi/m^2/s. Examples of radon emanation rates at various Canadian uranium tailings impoundments are shown in Table 3-11. Kilborn and Beak (1979) concluded that the large range in values can be caused by weather variances between sampling events. When considering the physical, chemical, and climatic factors discussed previously, it would seem logical that the uranium tailings located in Canada would have a lower radon emanation rates due to their higher moisture content and seasonal snow covers, than similar tailings ponds in arid areas of the United States.

Table 3-11
Relationship Between ^{226}Radium Content and ^{222}Radon Emission Rates from Tailings
(from Kilborn & Beak 1979)

Location	Ore Production (tonnes/day)	Ore Assay (% U)	Estimated * Ra-226 Content (pCi/g)	Estimated Potential Radon Emission Rate (pCi/m^2/s)
Denison	-	0.1	335	535
Rio Algom	6000	0.14	470	750
Bancroft	-	0.1	335	535
Beaverlodge	900	0.2	670	1070
Wollaston Lake	1800	0.37	1240	1980

Rather than through direct measurement, computer-based modelling is sometimes used to estimate emanation rates or to create hypothetical, general base cases. During an investigation by Hans et al. (1981), conducted on behalf of the U.S. Environmental Protection Agency (EPA) into the health impacts of inactive uranium mines in the U.S., a list of 1250 surface mines and 2030 underground mines was acquired from the U.S. Department of Energy (DOE). From the information gathered from DOE, models were developed for surface and underground mines to predict the ^{222}Rn emanation rate and subsequent human exposures. The calculated annual release rate of ^{222}Rn was 8.0 Ci/yr from a model surface pit and 7.6 Ci/yr from model underground vent portals.

The International Atomic Energy Agency (1981) also calculated airborne release values of radioactive components. The calculations were based on emissions from a U.S. model mill processing 1,800 tonnes/day of 0.16% U_3O_8 equivalent ore grade, the results are presented in Table 3-12.

Table 3-12
Calculated Airborne Radionuclide Emissions from a U.S. Model Mill Tailings Pile
(from International Atomic Energy Agency, 1981)

Radionuclide	Release Rate[1] ($Ci*a^{-1}$)
^{238}U	0.014
^{234}U	0.014
^{230}Th	0.19
^{226}Ra	0.20
^{210}Pb	0.20
^{210}Po	0.20
^{222}Rn	7000

[1] 1 Ci = $3.7*10^{10}$ Bq

Ambient radon emissions could be expected to rise upon retrieval of tailings from stable or pseudo-stable surface impoundments because higher levels of radon gas would be able to escape from the tailings more easily. Gaseous ^{222}Rn, its daughters, ^{230}Th, ^{226}Ra, and ^{210}Pb, are felt to be the radionuclides of most concern by the International Atomic Energy Agency (1981) when considering health impacts from atmospheric radionuclide releases from tailings impoundments. Based on data represented in Table 3-12 the atmospheric concentration of ^{222}Rn could be 40 to 2000 pCi/m^3 (1 to 54 Bq/m^3) using a "typical dilution factor" of $2x10^{-7}$ s/m^3. The U.S. Nuclear Regulatory Commission proposed a maximum ^{222}Rn emanation limit of 0.74 Bq m^{-2} s^{-1} (20 pCi m^{-2} s^{-1}) from stabilized tailings (International Atomic Energy Agency, 1981).

In addition to direct transport in the air phase, solid particles containing radioactive contaminants can also be transported. For example, wind dispersion of tailings from dry, uncovered tailings impoundments may result in widespread distribution of suspended tailings materials.

Wind transport of uranium tailings dust can also cause radionuclide contamination of surrounding soils and watersheds (International Atomic Energy Agency, 1981). The particle size, density, and shape (in some cases) of the tailings will determine their ability to be transported by wind action. Sieve analysis of windblown and unclassified tailings from the Elliot Lake area are presented in Tables 3-13(a) and 3-13(b). Windblown tailings were predominantly finer than 45 mesh (0.35 mm). Based on these data, as much as 94% of the unclassified tailings sampled at Elliot Lake have the potential for air transport (Table 3-13(b)).

Table 3-13(a)
Sieve Analysis of Wind Blown Tailings from the Idle Stanrock Tailings Site, Elliot Lake, Ontario
(from Kilborn and Beak, 1979)

Mesh Size	Percentage Retained
10	0.021
25	0.029
45	1.47
100	85.3
200	11.7
Passed No. 200	1.44

Table 3-13(b)
Sieve Analysis of Unclassified Tailings from the Elliot Lake Area, Ontario
(taken from Kilborn & Beak, 1979)

Mesh Size	Percentage Finer
48 (~45)	94
65	84
100	70
150	58
200	47

Wind velocity, surface stability, and particle size determines the ease with which cohesionless tailings are transported by wind (Kilborn and Beak, 1979). Actual measurements of suspended particulates originating at uranium tailings ponds in the Ellict Lake area were presented by Kilborn and Beak (1979) (Table 3-14). For the idle Nordic mill in Elliot Lake, Kilborn and Beak (1979) reported a ^{226}Ra concentration of 0.035 pCi/m^3. This number was in agreement with suspended particulate radioactive levels found in the southwestern United States (Table 3-15).

Table 3-14
Survey Summary Of Tailings Suspended Particulates In Air
(from Kilborn and Beak, 1979)

Site		Distance to Dry Tailings (m)	Number of Readings	Concentration of Suspended Particulates In Air ($\mu g/m^3$)			
				Average	Geometric Mean	Highest	Lowest
Stanrock	1 (Inactive)	100	31	115	45	1051	14
	2	100	30	91	55	575	9
	3	250	29	22	19	60	6
Long Lake	1	50	13	44	26	240	4
	2	50	13	15	13	49	6
	3	1000	15	38	19	262	1
Nordic	1 (Inactive)	100	73	94	32	1965	0
	Canmet Lab	200	66	24	20	65	4
Crotch Lake	1 (Inactive)	20	38	54	27	341	4
	2	20	35	43	22	300	4
	3	50	9	15	13	34	5
Panel	1 (Inactive)	10	5	10	7	24	1
	2	20	2	44	30	75	12
Quirke	1	120	48	42	33	147	7
	2	30	28	48	23	441	1
	3	1000	8	67	42	276	16
Sheriff Lake	Control	1000	51	21	18	48	5

The effects of uranium-tailings dust on vegetation and surface water ingested by animals, and overall radiation exposure from dustfall, are all discussed by Kilborn and Beak (1979). They reported a concentration in air, calculated from Stanrock data collected 100 m from the tailings pond, of 0.49 pCi/m^3 for ^{226}Ra (highest suspended particulate concentration of 1051 μg/m^3 multiplied by 470 pCi/g activity level of suspended particulate). The relationship used by Kilborn and Beak (1979) for vegetation contaminant uptake under dustfall conditions was:

Table 3-15
Suspended Particulate Levels in the Southwestern United States
(taken from Kilborn & Beak, 1979)

Radioisotope	Activity (pCi/m^3)	% of MPC*
^{226}Ra	0.04 - 0.07	4 - 7
^{210}Pb	0.01	0.3
^{210}Po	0.061	0.9
^{230}Th	0.0009	1.0

* Maximum Permissible Concentration

Concentration in Vegetation (pCi/kg) = Foliar Deposition + Soil Uptake (pCi/kg) (3-2)

A concentration in vegetation of 2,850 pCi/kg was reported for ^{226}Ra from Nordic mine data.

In a report for Rio Algom Ltd. in Elliot Lake, Senes Consultants (1991b) determined there would be an increase in the radioactivity of surface uranium tailings at the Quirke and Panel Mines resulting from the cycloning of tailings so that the coarse fraction could be used as engineered backfill. The finer-grained tailings returned to the tailings impoundments contained higher levels of radionuclides, which would result in the surface tailings impoundment having an elevated average radiological activity. As a result, air quality, as well as the quality of local waters (Sections 3.1.2 and 3.1.4), could be degraded further than expected from whole-tailings characteristics.

3.2 Technical Aspects of Retrieval

Section 3.1 discussed in detail many of the concerns related to retrieving tailings from stable or pseudo-stable surface impoundments. That section could be thought of as a discussion of "what to watch for" and "what to possibly expect". This section, on the other hand, discusses technical issues and techniques for retrieving tailings, and thus more of a "how to" discussion. Thirteen percent of respondents to this study's questionnaire discussed methods of retrieval (Appendix B).

Goode (1993) explained that poor mill recoveries in past decades and environmental liabilities justify the retrieval of tailings from impoundments and reprocessing to recover economic metals. In 1993, the rate of tailings retrieval was estimated at 40,000 tons a day worldwide. Several mines have conducted retrieval programs (Table 3-16). A particular practical concern expressed over tailings retrieval was the presence of "trash, rubbish, old containers [and drums], and a wide variety of unmentionable articles" as well as any vegetation, timber, and trees. This trash can block the mechanical equipment used for retrieval and any related ditches and pumps. Passing retrieved tailings through screens is recommended for segregating much of the trash.

3.2.1 Methods of retrieval

In choosing a method of tailings retrieval, consideration must be given to subsequent methods of transport and placement in the underground workings (Sections 3.2.3 and 4.2.2). For example, if dredging is the selected method of tailings retrieval, then hydraulic transport and placement can be used (Wood, 1983). On the other hand, dry excavation may require truck or conveyor transport and can be used with pneumatic or mechanical placement. Coordinated planning of retrieval with transport and placement may reduce costs, improve efficiency, and is recommended.

Table 3-16
Examples of Tailings Retrieval
(from Goode, 1993)

Location	Metal	Rate (tons/month)	Method
Chaffers Plant, Kalgoorlie, Australia	Au	60,000	Monitor with 15% solids feed
Daggafontein, South Africa	Au	1,000,000	Monitors
Eastmaque, Kirkland Lake, Ontario, Canada	Au	58,000	Cutter dredge
ERG, Timmins, Ontario, Canada	Au	1,000,000	Monitors
Freegold Consolidated Mines, South Africa	Au, U, H_2SO_4	1,000,000	Monitors
Giant Yellowknife Mine, Northwest Territories, Canada	Au	250,000	Monitors
Lac Minerals, Kirkland Lake, Ontario, Canada	Au	22,000	Cutter dredge
Mount Morgan, Queensland, Australia	Au	250,000	Bucket wheel dredge
Rand Mines, South Africa	Au, pyrite	370,000	Front-end loaders and monitors
Simmergo, South Africa	Au	174,000	Electric shovel and truck
Anaconda's Darwin Project, California, USA	Ag	120,000	Font-end loader
Santa Julia Plant, Real del Monte y Pachuca, Mexico	Ag	200,000	Monitors
Miami Copper Co., Arizona, USA	Cu	324,000	Monitors
Nchanga, Zambia	Cu	1,500,000	Erosion & monitors
Blyvooruitzicht Gold Mine, South Africa	U, Au	100,000	Bucket wheel
Chemwes, South Africa	U, Au, H_2SO_4	290,000	Monitors & bucket wheel excavator
Eldorado, Port Radium, Northwest Territories, Canada	U	5,000	Suction dredge
ERGO, South Africa	U, Au, H_2SO_4	1,700,000	Monitors

3.2.1.1 Excavation

Dry removal of tailings can have very high economic costs due to the high capital expenditures that may be required for equipment and the labour costs associated with usage of this equipment (Sims, 1972). Additionally, excavation of the tailings using tracked or wheeled vehicles or other heavy equipment may be very difficult or even uncertain (Sims, 1972; Bean, 1972). Difficulties arise from the unpredictable nature and physical characteristics of the tailings (Sims, 1972; Goode, 1993), especially after the top layer of the tailings have been removed (Bean, 1972). These points have been discussed in detail in Sections 3.1.3 and 3.1.4.

For example, an area on the surface of the tailings pond may look dry but "the effective use of the bulldozer may only be ten feet before moisture restricts their use", and the equipment must be moved to another area to give the newly exposed tailings time to dry. During a rainy period, no drying occurs and the retrieval process can come to a halt (Sims, 1972).

Conventional shovels and trucks can be used, but clay materials in the tailings may make unloading of the shovels and truck boxes difficult due to their stickiness and may require a hose to dislodge them (Bean, 1972). This could increase water movement through an impoundment (Section 3.1.3). Scrapers, bucket wheels, draglines and front-end loaders can also be used, but may still suffer from similar problems.

With moist tailings, retrieval may also be difficult due to instability and fluidity. Ackman (1982) reported on coal mine operations which used front-end loaders and bulldozers for removal of wet ARD sludge from treatment ponds. Ackman (1982) noted that this was the least used method due to the cost and generally high water content of the sludge which made removal difficult.

Goode (1993) discussed dry excavation as a method of retrieving tailings, particularly where high solids content is necessary for optimum recovery of metals. A peak digging rate of

400 m^3 of tailings an hour was calculated for excavators with 0.25 m^3 buckets. Some examples are listed in Table 3-16.

3.2.1.2 Dredging

Dredging, as referred to in this report, is the use of a barge or floating platform within an impoundment from which submerged tailings are retrieved with various techniques. Sims (1972) and Goode (1993) have discussed the unpredictability of old tailings ponds and the potential difficulty that may be encountered in trying to maintain and control the water necessary for dredging of tailings from an impoundment (Section 3.1.3). Consideration must be given to whether the tailings are too porous to float a dredge (Bean, 1972). If dredging leads to the uncontrolled percolation of water, the hydraulic head in containment structures may increase and reach dangerous levels, possibly resulting in a failure (Section 3.1.5). The failure of a dam may far outweigh any benefits that may be achieved by moving the tailings (Sims, 1972).

Other operating problems reported by Sims (1972) include:

❶ dredges pump a low percentage of solids per water volume, which may dramatically affect operating costs;

❷ labour costs are high; and

❸ maintenance in corrosive conditions may be a problem.

Other factors which can affect the cost and difficulty in dredging tailings are:

❶ physical and chemical characteristics of the tailings;

❷ availability of water required for dredging;

❸ remoteness of the mine and associated cost of bring equipment to the site;

❹ impoundment stability and any modifications required prior to the start of the dredging program;

❺ special considerations in the case of uranium tailings with regards concerns over radioactivity of the tailings;

❻ special handling and/or concerns over acid producing tailings (seepage production, production of excess acid water).

For the Quirke and Panel mines in Elliot Lake, Senes Consultants (1991b) stated that using a dredge to recover their uranium tailings would require 30% solids for retrieval, referring to dredge-retrieval projects in Ontario which called for 30% solids. At Quirke Mine, water-treatment sludge containing barium/radium-sulfate precipitants has successfully been dredged and dewatered for placement underground (CANMET and Kilborn, 1981).

Goode (1993) discussed the use of dredges for the retrieval of tailings. A relatively small dredge platform measuring 8 m by 2.4 m with a 6-inch dredge can retrieve 50,000 tonnes a month. A few case studies of dredging of tailings show success under the appropriate conditions (Table 3-16).

Whiteway (1994) reported retrieval of gold tailings at the Macassa Mine in Kirkland Lake, Ontario, beginning in 1993. Approximately 350,000 tons of tailings will be dredged in 1995, and an additional 200,000 tons a year in 1996 and 1997.

3.2.1.3 Slurrying

The basic concept of hydraulic removal as a slurry involves the use of:

> "slow rotating or oscillating jets of high pressure water working beneath the solids material, causing it to be undercut and to collapse into this high pressure stream and slurried. The slurry flows into a sump from which it is pumped for transportation to the desired location" (Sims, 1972).

Hydraulic slurrying systems have been used at mining operations for china clay, alluvial tin, and copper to retrieve tailings for reprocessing (Kilborn and Beak, 1979; Sims, 1972; Goode, 1993).

If tailings retrieval by slurrying can be implemented, it would likely be the simplest, fastest, and cheapest method, and is thus probably the most desirable method (Bean, 1972). However, if slurrying must work from the top downwards into a tailings mass, rather than undercutting slopes with jetted water, it would be less cost effective. Due to the moist nature of tailings and their commonly low slopes, slurrying from the top would probably be required in many impoundments.

If undercutting of slopes with water jets can be attained, however, the behaviour of relatively steep slopes which have been standing a long period of time may be very unpredictable. Consequently, pumps and jets in close proximity to the slope may be lost if the it slumps or flows (Sims, 1972).

One example of a slurry system is the "MARCONAFLO" system used by the Atlas Consolidated Mining and Development Company to remove copper tailings from emergency storage ponds that were almost full. The system consisted of:

> "a capsule 10 feet in diameter and 30 feet high, which contains the high pressure water lines...which direct water to the tailings for slurrying. This slurry flows back to the grizzly, which is constantly cleaned by the jet itself, and into the sump and slurry pump, all of which are in the capsule, along with drives, hydraulic gates and other necessary operating devices. Because this unit is permanently installed for use whenever needed, it is mounted between four pilings driven into bedrock and connected to the berm of the pond by means of a bridge...for operational access, maintenance, and to support slurry and water lines. In operation, the capsule is lowered into the tailings pond by means of sink jets placed in the bottom. These slurry the material directly beneath the capsule and as it lowers into this material, the resulting slurry is pumped away. Once it is lowered to its desired position, the sink jets are turned off, the capsule secured to the pilings and one or more of the MARCONAJETS are started. Each has a capacity to meet the requirements of 440 tons per hour so that control is exercised over the removal in such a way that the berm will remain intact. As material pumped from the pond, it goes into a drop box for insertion into the tailings disposal pipeline" (Sims, 1972).

The requirements of the MARCONAFLO system included 4 MARCONAJETS, a water flow of 1250 gallons a minute at 450 pounds a square inch, high pressure water pump rated at 460 horsepower, and 50% solids repulping density.

Modifications to the above system included (Sims, 1972):

➊ the use of a "toboggan" which can be pushed around and into the tailings;

➋ the use of bulldozers to move the material to the device containing the jets, pumps, and controls, which is basically excavation (Section 3.2.1.1); and,

➌ the use of a specialized system which removes tailings from beneath a water cover in a slurry form, which is basically dredging (Section 3.2.1.2). This system is reportedly useful for corrosive tailings, unknown materials in the tailings, variable-capacity removal (a hundred to several thousand tons a day), and adverse climatic conditions.

The adverse climatic conditions may be relevant for Canada where relatively long winters with low temperatures can adversely impact the reliability and effectiveness of slurrying through freezing.

Goode (1993) discussed the use of high-pressure water jets (monitors) creating a slurry as the preferred method of retrieval (Table 3-16). With a monitor providing 350 m^3 of water an hour, approximately 350 tonnes of tailings an hour will be slurried. This corresponds to a rate of advance of 2 m an hour for a 20-m-high face. The slurry is typically directed through trenches to a sump, where it is then pumped to its final destination.

3.2.2 Reprocessing

Many ore bodies now being mined with new equipment and technology were once considered to be uneconomical. In the last 100 years, new and improved mining and milling techniques such as chlorination, cyanidation, flotation, and solution mining have redefined what is considered an ore body. It is thus conceivable that within the next 100 years new concepts could resolve the dilemma over tailings (Section 2) by turning them into a valuable mineral resource. In other words, a tailings pile may become ore which is reprocessed or re-milled after retrieval by one or more methods (Section 3.2.1). Goode (1993) estimated that 40,000 tons a day were retrieved for the purpose of reprocessing (Table 3-16). Only 3% of the respondents to this study's questionnaire (Appendix B), however, mentioned this issue.

If reprocessing of tailings was included as part of an underground disposal plan, the reprocessing could pay for at least part of the costs. There are case studies in Canada of reprocessing of tailings, but success is reportedly limited due to problems with retrieval and mineral recovery (Table 3-16). For example, an attempt to retrieve and reprocess gold tailings in Timmins, Ontario, failed and reportedly left the area a visual "mess". At the inactive Norebec-Manitou mine near Val d'Or, Quebec, reprocessing of 700 tonnes of precious-metal tailings a day was underway by mid-1993 (Toronto Star, 24 April 1993; Le Devoir, 28 April 1993). "The multi-step process begins with the removal of sulphates from the residue; it ends with the valuable metals having been extracted and the neutralized residues returned underground or into artificial or natural lakes" (The Northern Miner, 1993). This is a rare example of retrieval of surface tailings, reprocessing for a profit, and subsequent disposal underground. However, the Norebec-Manitou project has apparently encountered unspecified problems and is primarily in the promotion phase at this time.

Whiteway (1994) reported that dredging and reprocessing of gold tailings at the Macassa Mine in Kirkland Lake, Ontario, accounted for almost 20% of mill output of gold. Plans for reprocessing at Macassa included 750,000 tons of tailings in 1995 through 1997.

Down and Stocks (1976) reported that the U.S. Bureau of Mines has researched the viability of reprocessing various tailings materials into commercially saleable products. Some of the work included:

❶ Taconite (iron) into foam building blocks

❷ Copper into dry pressed building bricks and glass

❸ Taconite, copper, lead/zinc into lightweight foamed building products

❹ Gold, copper, lead, zinc into dense calcium silicate bricks and aerated concrete

❺ Aluminum red muds into thermal insulation, concrete additive, stabilizer, absorbent.

There was no mention of any large-scale reprocessing stemming from the research.

Items for considerations, prior to undertaking a reprocessing project, have been outlined by Bean (1972) and include:

❶ a review of old records to establish the amount of material available for reprocessing;

❷ a review of old assay data to identify which minerals are present, and how they might be recovered. However, these records should not be the sole basis for undertaking this venture, and adequate sampling of the material should take place;

❸ an assessment of weathering and alteration of the minerals since tailings deposition. Minerals can become altered over time and when exposed to varying environmental conditions such as physical and chemical weathering (Sections 2.3 and 3.1). For example, pyrite changes chemical form when exposed to the atmosphere. The alteration of minerals can significantly change the expected metallurgy of waste for reprocessing; and,

❹ the association and interaction of minerals in the waste material to be reprocessed which can affect mineral recovery during reprocessing.

All of the above factors should be addressed before developing a reprocessing scheme, and a detailed program of sampling and analysis should be completed in the very early stages to help to ensure a successful project with limited surprises (Bean, 1972). Sampling of a tailing impoundment can be very difficult due to the segregation of slimes, sands and coarse fractions when the tailings were deposited as was discussed in Section 3.1.3 (Bean 1972; Sims, 1972; Snodgrass et al., 1982). However, other problems with sampling can also arise (Bean, 1972):

➊ finding a sampling method which will allow the stratification of the tailings to be preserved;

➋ the physical difficulty of getting equipment such as drill rigs onto and around the impoundment; and

➌ sample preservation to prevent incipient oxidation and precipitation of soluble materials within the pore water of the tailings, and the change or loss of moisture.

After all of this research and sampling, yet more exploratory work should be carried out. The market demand and the costs of retrieval, recovery, and transportation should be compared. If retrieval and reprocessing of tailings is undertaken for environmental reasons, such as to remove acid generating sulphur materials, the project is enhanced if economic value can be extracted from the tailings. For example, some tailings with a relatively small percentage of pyrite may have gold associated with the pyrite which can be recovered to offset costs (Goode, 1993; Golder Associates et al., 1992).

Despite all of this work, unexpected practical problems with reprocessing can arise. Some of these difficulties were discussed by Bean (1972):

➊ hang-ups of feeders caused by the sticky nature of the clay fraction, as well as their usual problems;

➋ ball mills and drums filled with stones can handle clay material quite well with limited difficulties, but screens or trummels and sprays can have problems; and,

➌ the final disposal of waste materials left after reprocessing and reclamation of the tailings area (Section 3.2.4).

Based on this review of literature and case studies, the option of reprocessing tailings for economic benefit has created excitement and received attention. However, there are many factors and problems which should be resolved early, but apparently were not in some cases.

3.2.3 Transportation to underground workings

After tailings have been retrieved (Section 3.2.1) and perhaps reprocessed (Section 3.2.2), they can be transported to the mine for placement. Eight percent of respondents to a questionnaire (Appendix B) raised a concern over this issue.

When discussing underground disposal of coal mine refuse, Gaffney (1983) noted that up to three transport systems may be necessary to take the refuse from the treatment/ pretreatment area to the final disposal location:

❶ surface transport from the waste preparation area or storage area to the underground opening, such as by conveyors, aerial tramways, trucks, mine rail car (Wood, 1983), and hydraulic systems (slurry pipelines) or pneumatic systems;

❷ transport from the opening to the underground level, such as through gravity, mechanical, pneumatic or hydraulic systems; and,

❸ transport underground to the final point of deposition, such as with mechanical, conveyor, mine rail car (Wood, 1983), gravity system (Wood, 1983), hydraulic or pneumatic systems.

Each stage of transport may require a different system to be used. The selection of a particular system will be determined by the local mine operating conditions, such as (Gaffney, 1983):

❶ the topography;

❷ the waste volumes;

❸ the availability of water; and,

❹ the characteristics of the waste material.

For example, a potential problem associated with transporting tailings with a high moisture content by a mechanical means, is "carry-back" (Wood, 1983) or incomplete discharge of the tailings from the transport device.

When choosing the best method for transport of tailings to the point of deposition, attention must be given to any associated methods of tailings retrieval (Section 3.2.1) and placement (Section 4.2.2). For example, if dredging is chosen as the method of tailings retrieval, then hydraulic transport and placement should be used (Wood, 1983). If dry excavation is used for retrieval, then truck or conveyor transport can be used with pneumatic or mechanical placement. The selection of transportation system should also consider site-specific factors such as the layout of the mine, mine operating conditions, existing equipment, required transport capacity, and related economics (Wood, 1983).

An analogy to tailings, Ackman (1982) reported that the ARD sludge from treatment ponds was pumped or trucked to boreholes drilled into underground abandoned deep mines, or inactive operating portions of producing mines. Fuel costs, labour costs, and the initial capital for haul trucks and loading equipment (Ackman, 1982) were important economic considerations when truck haulage was examined as a method for sludge and tailings transport.

If truck haulage is used, increased road traffic may result unless waste is brought back to the mine on the backhaul. There may also be increased dust production on the roadway from hauling dry fine tailings. If tailings are transported saturated, the trucks must be able to carry a load without spillage.

Many underground mines use a rail system, and this system must be reinstated or maintained for transport of tailings back to the mine. For an active mine, the problem of two-way traffic must be addressed if tailings are taken to another area of the mine besides the active mining area.

With pipeline transport of sludge or tailings, the initial capital expenditures for pipe, fittings, pumps, and labour for system design, operation, and maintenance (clog prevention, etc.) are primary economic factors (Ackman, 1982). Also, blockage of pipelines in gravity or pneumatic systems leads to additional costs if the moisture content of the tailings is too high

(Ackman, 1982).

Another method of moving tailings on the surface is with aerial tramways (Wood, 1983). This method has been used in the coal industry to transport coal waste, and to transport ore at other mining operations such as Britannia Mines in British Columbia. This system is particularly valuable in areas of steep grades and cliffs.

In any case, once the tailings are delivered to the mine, a method of unloading and temporarily storing the tailings would be needed. This may involve hoppers, storage ponds or piles, and surge ponds.

An underground mine located in Lee County, Virginia and Harlan County, Kentucky was the site for design and testing of a hydraulic transport system to take coal mine refuse to the underground workings for disposal as backfill. The coal refuse consisted of a mixture of materials ranging in size from 4 inches x 1/4 inch, down to 28 mesh x 100 mesh, and included filter-cake material with 85% of the particles less than 325 mesh in size at 30% moisture content. Vortex-type slurry pumps, a polyvinylchloride pipeline, and underground filter barricades were designed to pump 200 tons an hour of a 55% solids (by weight) slurry at 12.5 feet per second (Popovich and Adams, 1985).

From this work, Popovich and Adam (1985) considered the main factors to evaluate when designing a slurry transport system as:

❶ the selection of pumps. The pump determines the allowable physical parameters and characteristics of the slurry such as the head, the percentage of solids and the maximum particle size of the waste backfill.

❷ the solids concentration, throughput rate, the minimum slurry speed to prevent settling (critical velocity), and friction headloss determine the inside pipe diameter.

❸ the selection of pipe material. In the case of Popovich and Adam (1985), polyvinylchloride (PVC) pipe was four to five times less expensive than lined steel pipe. PVC pipe offered good wear characteristics, but was limited to 200 to 300 psi operating pressure. Thus, for long distances, pumps had to be located along the pipeline instead of being installed in series at the beginning of the pipeline.

❹ the difference in elevation between the location from which the backfill originates, such as a surface processing plant or impoundment, and the mine area to be backfilled.

In this case, the underground mine portal was 1,600 feet from the plant at about the same elevation, the backfill area was 400 feet from the portal also at about the same elevation. Test pumping was to start and stop with water to prevent plugging. At first, low solids concentrations were used and then increased to a "safe" maximum as high as 55% by weight (Popovich and Adam, 1985).

3.2.4 Final cleanup and reclamation

After tailings are retrieved from a surface impoundment, the last technical issue would be cleanup of the former impoundment. If all tailings were retrieved, an impoundment could be dismantled, or at least no further monitoring and maintenance would be needed if there had been no prior environmental effects and pathways established (Sections 2.3 and 3.1). More likely, some amount of tailings would remain and/or some pathways of contaminant migration would have been previously established. Fifteen percent of respondents to a questionnaire (Appendix B) mentioned this issue.

Due to the physical "swelling" of rock as it is processed into tailings and other factors (Section 2.4), one mine's tailings would likely exceed the volume of its underground workings by at least a few tens of percent. As a result, a significant portion of the surface-impounded tailings could remain in the surface impoundment. This would preserve many of the disadvantages and concerns associated with the full impoundment (Section 2.3 and 3.1). Consequently, unless a few key disadvantages and concerns are removed by partial placement of tailings underground, this disposal option optimally requires "all or nothing": if all tailings cannot be moved underground, then there may be little value in moving any tailings.

On the issue of full retrieval, Golder Associates et al. (1992) wrote that high-pressure water could be used to remove the last of the tailings which remain after bulk handling and retrieval. However, some contamination of the natural ground would remain and possibly flush away with precipitation (this issue is addressed further in the next paragraph). The type and level of contamination being flushed will depend on the composition of the original tailings, and the composition and geology/geomorphology of the natural ground (Section 3.1). The impact on the area watersheds from this flushing must be determined and controlled if necessary.

Even if all tailings can be retrieved, monitoring and maintenance of water and air quality may be required in and around a former impoundment for years or decades if contaminant pathways had been established. For example, if local groundwater systems had been contaminated by metal-laden acidic water from an impoundment, removal of the tailings could reverse groundwater flow or accelerate its movement away from the impoundment (Section 3.1.3). In any case, monitoring and perhaps water collection and treatment could be required for decades until the groundwater was remediated and/or all discharge to surface-water courses ceased. Consequently, full retrieval of tailings from a surface impoundment does not immediately remove all concerns over the impoundment area and surrounding environment.

Long-term issues that should be addressed for closure include various aspects of physical, geochemical, biological, and mechanical stability. Tailings-surface erosion and containment-structure stability were the primary concerns of Nelson et al. (1986). The latter concern was the focus of a recent AECB workshop (SNC-Shawinigan Inc et al., 1995). MacLaren Plansearch (1987) used a more general approach to guide the optimization of closure planning. Through a generic Elliot Lake example, MacLaren demonstrated the optimization approach and the methods by which optimization can be attained (Table 3-17).

Table 3-17
Report Outline and Approach for Optimization of Closure Planning
(from MacLaren Plansearch Inc., 1987)

Chapter/Step	Topics
One	Objectives
Two	Characteristics of tailings site Closure options Evaluation criteria Incorporation of time-dependent effects and uncertainty
Three	Optimization procedure Evaluation techniques Contributions from related (nuclear) technologies Comparison of techniques
Four	Cost-impact criteria Health-impact criteria Other impact criteria
Five	Cost/benefit Cost/effectiveness Multi-criteria Decision process

If an impoundment required on-going monitoring, King and MacPhie (1991) listed several items that should be addressed during their study of the Quirke and Panel uranium tailings basins:

➊ monitoring of impoundment seepage (the use of weirs, routes of flow during flood and minimum flow conditions, etc.);

➋ stability of the dam under weather conditions considered normal (i.e., normal rainfall, snowmelt) and extreme (i.e., 100 year flood and associated flood flows into and out of the impoundment, drought conditions especially if tailings reclamation includes an aqueous cover for the tailings);

❸ area hydrology;

❹ area hydrogeology;

❺ area streamflow;

❻ impoundment drainage;

❼ water inflow pathways;

❽ evaporation; and,

❾ seepage losses.

King and MacPhie (1991) address simulations of tailings basins in order to create water-balance predictions.

Reclamation should be directed towards various physical, geochemical, and biological factors. These factors are discussed in Section 2.3 and 3.1 and only a few are reiterated below.

Some items for consideration prior to reclamation of an environmentally disrupted area caused by mining activities include (Johnson, 1979):

❶ simultaneous reclamation of other, non-tailings areas affected by mining operations, such as the reclamation of treatment ponds by using waste-treatment sludges for neutralization of acidic conditions (authors' note: use of such sludges for neutralization should actually be discouraged, because attempts in Canada show the sludges redissolve and release significant amounts of soluble metals); and,

❷ the placement of mine waste in locations with easy access for future recovery or recycling (authors' note: easy access may be discouraged in some cases, such as for uranium tailings).

Investigations at the Waite Amulet tailings site at Noranda, Quebec (St-Arnaud et al., 1989), also illustrated the need for a good knowledge of physical data on:

❶ the tailings - grain size, chemical composition, acid production potential, amount of sulfides, available oxygen, buffering capacity of the tailings;

❷ the impoundment - depth to water table, proximity of discharge point, the amount of water which infiltrates into the tailings, flow system across the dam, horizontal and vertical gradients; and,

❸ climatic conditions to ensure an appropriate reclamation option is chosen.

In May of 1977, the U.S. Nuclear Regulatory Commission developed guidelines for the siting, design, operations and reclamation of tailings impoundments for the uranium industry (Scarano, 1980). These guidelines can be applied to impoundments after full or partial retrieval, and to new holding areas for residual tailings. The guidelines are:

❶ the tailings should be located in an area which is isolated from people, so that the remoteness limits public exposure to a minimum;

❷ the tailings should be located in an area were disturbance and/or dispersion by natural forces is reduced to a minimum;

❸ the design of the impoundment should ensure that contaminant seepage into the surface and groundwater systems be eliminated or reduced to the minimum level achievable;

❹ wind-blown dust should be eliminated from "unrestricted areas" during operation and after closure;

❺ direct radon emanation should not exceed essentially background levels;

❻ radon emanation rate should not be more than twice that of the surrounding natural environment;

❼ long-term reclamation should not require continual monitoring and maintenance;

❽ bonds must be secured to guarantee that sufficient funds are available for complete reclamation of the site.

The portion of reclamation activities pertaining to revegetation often receives strong attention. Hamel and Howieson (1982) confirmed that in Canada the majority of tailings reclamation work has focussed on revegetation and/or concentration of hazardous chemicals. Revegetation of uranium tailings can serve a cosmetic purpose as well as a contaminant-control purpose (Section 3.1.2), although long-term assurances of contaminant control are still under investigation. Many factors that can degrade the success of vegetative reclamation over long periods of time are discussed in Sections 2.3 and 3.1, and only a few points are reiterated here.

Short- and long-term successes of vegetative reclamation may be difficult and costly due to any combination of the problems described by Down and Stocks (1976):

(1) instability;

(2) spontaneous combustion;

(3) steep slopes;

(4) inhibitory water regime;

(5) high levels of toxic elements and varying amounts of process chemicals;

(6) acidity;

(7) compaction and cementation;

(8) inhibitory surface temperature regime;

(9) sand-blasting;

(10) low nutrient status; or,

(11) absence of soil micro-organisms and fauna.

Many of these problems can be enhanced in remnant tailings following retrieval.

Constable and Snodgrass (1987) conducted experiments on both old and fresh tailings. They found that tailings with a vegetation cover of a fescue/red-top grass mixture and layer of sewage sludge for nutrient and organic source became acidic as quickly as the controls for the experiment. The acidic conditions then killed the vegetation. After a seven-year study of vegetation growth on tailings treated with a proprietary solidification process, the grasses had not died, but yields decreased steadily throughout the course of the experiment. The success of long-term vegetation was not known.

In addition to vegetative cover, other types of covers have received attention in order to control radon emanation and seepage of contaminants into the environment. An example of this is a comparison of various covers versus liners that was conducted by Hartley et al (1982). Also, Chilton and Pfuderer (1989) reported on investigations carried out during the U.S. Department of Energy's Uranium Mill Tailings Remedial Action (UMTRA) Project. Descriptions were given of the various types of covers, erosion-resistant binders, soil stabilizers, liners, grouting methods, windscreens to stop wind erosion of surface, thermal stabilization, leaching, neutralization to attenuate the migration of soluble radium that may leach in acidic conditions, and barium and potassium chloride treatment.

Williams (1978) conducted a review, and made recommendations on the usage, of surface-impoundment barriers, both natural and man-made, to control contamination of surface and ground waters from tailings seepage. Rogers et al. (1982) also gave an example of a natural barrier using a moist cover on tailings. The moist cover is desirable to control radon emanation from the tailings into the atmosphere, but the same moist cover can increase percolation of water through the tailings and cause seepage of contaminated water into the groundwater system. Rogers et al. (1982) discussed a model under development, for surface disposal of tailings, which would take into consideration many of the contaminant pathways (Section 2.3) and aid in the development of an interactive reclamation plan. The model will also estimate the cost of covers and the associated reclamation plans.

Hart et al. (1986) discussed modelling of multiple layers of covering materials for uranium tailings. The model deals with radon diffusion through tailings and various cover types both wet and dry. This type of modelling can be used to determine the most appropriate method of reclamation for uranium tailings which will not fit underground, and may be required by mines and regulatory agencies to assure adequate protection of the environment from radon emanation.

Final cleanup and reclamation can sometimes require a much wider focus than the impoundment area. Blanchard et al. (1981) described a case in Grand Junction, Colorado, where uranium tailings were used in the construction of homes resulting in ^{222}Rn decay-product concentrations indoors reaching a level which required a Federal-State remedial program to rectify the situation. Kennedy et al. (1977) also discussed the Grand Junction area where uranium tailings were used as fill around footings and under concrete floor slabs of homes, schools and commercial structures. Kennedy et al. (1977) also mentioned a similar situation at Port Hope, Ontario, near the Eldorado Refinery.

3.3 Costs

Although it is too late for operating and closed mines, the incorporation of tailings-retrieval plans into a design of a surface impoundment can minimize later pitfalls (Section 3.1) and costs, while maximizing retrieval (Section 3.2). Since this type of early-stage planning has apparently not been performed in the past, the costs mentioned in this section reflecting late-stage planning which is typically more expensive. Over one-third (35%) of respondents to this report's questionnaire (Appendix B) indicated cost was an important factor in tailings retrieval and subsequent placement underground.

Senes Consultants (1991b) made reference to a 1983 report by Steffen, Robertson and Kristen in which dredging of 1.1 million tonnes of tailings from the Beaverlodge Mine impoundments in Saskatchewan would cost $1.68 million or $1.51/tonne over a five-month period. Rio Algom Limited (1993) reported a higher cost for this project at $2.46/tonne. Senes Consultants (1991b) and Rio Algom Limited (1993) also estimated a cost of $3-4/tonne to dredge 750 tonnes a day of Kirkland Lake tailings in Ontario. By using only precursory data, Senes

Consultants then estimated a preliminary cost of \$3/tonne for the dredging of Rio Algom's Quirke and Panel Mines tailings near Elliot Lake, Ontario.

Senes Consultants (1991b) calculated that the cost to dredge 30 x 10^6 m^3 of Quirke Mine uranium tailings (roughly 40% of total tailings), of which the coarse fraction of approximately 15 x 10^6 m^3 would be used as carefully engineered backfill underground in the Quirke and Panel Mines, would be \$117,000,000. The calculation is based on: 30 x 10^6 m^3 uranium tailings times 1.3 tonne/m^3 times a cost of \$3/tonne. The project would take 19 to 56 years depending on the placement rate, and would require placement of the fine fraction in a surface impoundment. Additional costs included \$58,500,000 for thickening and backfilling, \$95,000,000 for mine rehabilitation and operation, \$9,500,000 for treatment of mine and tailings-pond water, and \$75,000,000 for hearings, reports, design, and contingency. These estimated additional costs bring the unit cost to \$18/tonne (Table 4-22).

In addition to the dredging/backfill option, Senes Consultants (1991b) also considered the costs for simply dredging the required volume of unsegregated tailings and slurrying it into the underground workings. The maximum dredged volume was estimated at 20.6 x 10^6 m^3 of Quirke and Panel uranium tailings (26% of total tailings) which would cost \$45,100,000. The calculation is based on: 20.6 x 10^6 m^3 uranium tailings times 0.73 tonne/m^3 of slurry times a cost of \$3/tonne. The project would take 3 years depending on the placement rate. Additional costs were estimated at \$8,100,000 for pumping and \$13,300,000 for hearings, reports, and contingency. These estimated additional costs bring the total unit costs to \$4.4/tonne.

Obviously, if underground disposal were to be implemented at Quirk and Panel mines, the slurry approach would be preferred because (1) it is faster, (2) it includes fine fractions which generally contain a higher percentage of radioactive and non-radioactive contaminants (Section 3.1.1), and (3) it is cheaper. However, Senes Consultants (1991b) did not explore underground disposal in Elliot Lake further because (1) at least 60% of the surface-impounded tailings would remain in the impoundments, requiring on-going attention (Section 3.1), and (2) it is "very expensive with minimal to no environmental benefit received for the large expenditure", although the costing of the offsetting benefits and intangible costs (Sections 2.3 and 2.4) as well as additional liabilities (Section 4) were not provided.

In 1985, an underground study mine located in Virginia and Kentucky, was the site for design and testing of a hydraulic transport system to take coal mine refuse back to the underground mine for disposal as backfill. The estimated cost for disposal of the coal refuse underground was $1.60 per ton of backfilled material (approximately 360,000 tons of refuse were disposed of each year) or $0.67 per ton of marketable coal. This cost was felt to be well within the range of 1985 surface disposal costs (Popovich and Adam, 1985).

The reclamation of the Nokomis-Coalton site (Johnson, 1979) in 1976-1977 after closure of the Reliance Mine in Illinois involved mine-waste retrieval and placement. The site consisted of two areas (220,000 m^2 and 57,000 m^2 about 400 m apart) covered by 1-3 m of mine wastes. The waste was excavated down to natural ground surface by dozers and trucked to an abandoned quarry for disposal. The quarry was covered with 1 m topsoil and revegetated. The original site was covered with sludge from a water-treatment plant which consisted of hydrated lime and soda ash by-product material. The site was revegetated and proved to be a successful reclamation project. The total cost of the project was $653,650 or $2.49/$m^2$ (mine refuse removal $1.90/$yd^3$ = $342,000; hydrated-lime and soda-ash addition $7.00/$yd^3$ = $12,250; seeding $1,200/acre = $74,400; mine refuse disposal $1.25/$yd^3$ = $225,000).

Ackman (1982) indicated that dredging was a common but expensive method of removing sludge from coal mine ARD treatment ponds. Costs ranged from approximately $2.00-4.50/$yd^3$ for sludge removal.

Goode (1993) estimated costs for a slurry-retrieval program recovering 4,800,000 t/yr over six months a year. Capital costs were placed at $7,000,000 (Canadian). Operating costs were estimated at $350,000/month, which was a unit operating cost of $0.438/t.

Goode (1993) estimated costs for a dredge-retrieval program recovering 1,800,000 t/yr to a total of 8,000,000 t. Capital costs were placed at $4,000,000. Operating costs were estimated at $103,000/month, which was a unit operating cost of $0.686/t.

When tailings are used as backfill underground, the fine fraction is usually removed and disposed of. The fines are generally returned to a surface impoundment, as was the case at Stekenjokk in northern Sweden and at Saxberget (a zinc and copper mine) in central Sweden (Broman, 1989). The Stekenjokk site contained about 4 million tons of reprocessed fine tailings of which 80% by weight had a particle size $< 80\ \mu m$. The chemical composition of the tailings was 0.19% Cu, 0.64% Zn, 0.11% Pb, 20.0% S, 18.6% Fe, and the calcite content was assessed at about 7%. The reclamation plan for this site included a water cover due to the low permeability of tailings in the subarctic environment and other physical characteristics of the site which made this the most appropriate option (Broman, 1989).

The Saxberget site contained about 4 million tons of reprocessed fine tailings of which 80% by weight have a particle size $< 125\ \mu m$. The chemical composition of the tailings was 0.2% Cu, 0.8% Zn, 0.4% Pb, 2% S, and the calcite content was assessed at about 0.5-1%. The hydraulic conductivity of the tailings was 5×10^{-6} to 3×10^{-7} m/s. The reclamation plan for this site included a dry cover due to the highly porous underlying layer of glacial material. This permeability along with other physical characteristics of the site made this the most appropriate option (Broman, 1989).

The unit reclamation costs at the Stekenjokk and Saxberget sites were $2.6/m² CDN and $12/m² CDN, respectively. This illustrates the economic value of locating tailings, which cannot be placed underground, in areas that will allow the use of a wet cover (Browan, 1989).

The Wismut uranium mines in Ronneburg, Germany (formerly East Germany), face an enormous task of reclaiming their mining areas. There are about 1600 km of tunnels, 140 mine and ventilation shafts, and six underground caverns with some excavations reaching 2 km in depth. Surficial waste includes thousands of waste piles, including one as large as 130 million m^3, and 18 tailings ponds, with the two largest over 250 hectares. The bill for cleanup of the area is estimated at up to 15 billion Deutsch Marks (Toro, 1991).

The levels of radioactivity at the Wismut uranium mining area, Ronneburg, Germany were described by Toro (1991) as being 0.65-0.7 Bq/g for the waste rock piles and 10 Bq/g for the tailings ponds. Metal leaching (Section 3.1) is driving 20 g/L of sulfate and 2 g/L of iron into the groundwater system (Toro, 1991). Radon gas emanation into the mine, and into homes in the region is up to 100,000 Bq/m^3. At these levels of human and environmental exposure and the large extent of underground workings, placement of at least some portion of the tailings and/or rock underground would likely provide some benefit.

4. PLACEMENT OF TAILINGS IN UNDERGROUND WORKINGS

Although the practice of underground placement of tailings in mined-out stopes was, and is still, primarily undertaken for stability purposes, increased environmental concerns over surface-impounded tailings has raised interest in this practice as a potential long-term tailings disposal method. Underground workings as a natural barrier to contaminant migration is a very appealing option, but there are limitations and potential problems, as introduced in Section 2.4 and discussed below. Ninety-three percent of respondents to this study's questionnaire (Appendix B) mentioned some aspect of underground placement.

When determining the viability of underground uranium tailings disposal, the complexities of the site-specific underground environment should be well understood. Fortunately, the extensive international research into underground disposal of high-level radioactive wastes has provided information and case studies on the optimum characterization of the underground environment. In Canada, this research has been directed and conducted by Atomic Energy of Canada Ltd. (AECL), which has focussed on granitic rock. Although most uranium and other tailings are not high-level nuclear waste, the lessons learned by AECL and other organizations are helpful in guiding the less critical task of placing tailings in underground workings.

For example, high-level-waste research was conducted on the thermal conductivity of rock matrices. Thermal conductivity can be important when placing uranium tailings with a high pyrite content underground. If the pyrite oxidizes underground, the generated heat may affect the conditions within placed tailings as well as the surrounding rock mass, such as through fracturing and mineral precipitation/dissolution. At the Brunswick Mine, when waste rock and tailings were placed underground for disposal, a fire started and burned for years (Whelland and Payant, 1991). The heat generated by this fire no doubt affected the underground environment and both the waste material and the host rock in some ways.

To continue this example of heat production, the thermal conductivity of some rock-forming minerals are given in Table 4-1 (Jessop et al., 1979). One point discussed by Jessop et al. (1979) was that variances in thermal conductivities, due to localized preferred mineral orientation, may not have an effect on the regional thermal conductivity. However, samples taken for laboratory experimentation which have small areas of varied preferred mineral orientation can show much scatter about the mean. This concept of regional vs. local sample-size characteristics should be kept in mind whenever dealing with natural systems. Nevertheless, despite the amount of research on heat effects from high-level radioactive wastes, there are still significant uncertainties. For example, the effect of heating (up to 100 °C for 398 days) on the host rock was less than predicted at the abandoned iron mine at Stripa, Sweden (Witherspoon et al., 1981). The rock mass did not act in an uniform linear isotropic manner with constant thermoelastic properties, which may be attributed to fracturing within the rock unit and the presence of unconformities in the rock mass.

By applying some of the heat response data collected by AECL and others, it may be easier to speculate on the effects of an underground fire and any other situation in which heat is generated, over the short and long term. This example on heat production and thermal conductivities provides the first indication in this chapter of the complexity of underground mines.

4.1 Effects on Existing Conditions in Underground Mines

Obvious issues with underground placement are the manner and extent to which existing conditions in underground workings would change, for better or worse. The changes would primarily affect the reactivity of the tailings and wall rock (Section 4.1.1), subsurface water quality (Section 4.1.2), underground water flow (Section 4.1.3), and underground air quality (Section 4.1.4). In fact, these factors along with temperature (discussed above) are interactive,

Table 4-1
Thermal Conductivity of Rock-Forming Minerals
(from Jessop et al., 1979)

Mineral		Conductivity W/(m·K)
Quartz	parallel to axis perpendicular to axis mean	10.7 6.3 7.7
Feldspars	microcline orthoclase albite anorthite	2.4 2.3 2.2 1.7
Micas	muscovite biotite	2.3 2.0
Pyroxenes	enstatite diopside	4.5 5.0
Serpentine		3.0
Amphibole		3.1
Olivines	forsterite fayalite	5.1 3.2

which means that any one factor can affect one or more of the others. The potential change of greatest interest to questionnaire respondents (Appendix B) was water quality (Table 1-1).

4.1.1 Change in reactivity of tailings or wall rock during/after placement

An obvious concern with underground placement of tailings are the degree to which the geochemical reactivity of tailings and wall rock may change, which in turn can affect temperature (discussed above), water quality (Section 4.1.2), rates of groundwater flow (Section 4.1.3), and air quality (Section 4.1.4). Twenty-three percent of respondents to this study's questionnaire (Appendix B) considered this issue important.

Many of the processes which can alter reactivity are discussed in the related section for surface-tailings retrieval (Section 3.1.1), and are therefore not repeated here. However, there can be additional processes due to the proximity of wall rock and its groundwater. For example, Pearson (1987) reviewed evidence for the presence and origin of brines in the Canadian Shield at depth. Interactions of tailings with brine may have to be assessed for some underground disposal environments due to the changes in geochemical reactivity that can be brought on by brines. Brines can contribute to degradation of cement additives, increased mobility of contaminants, and/or creation of unexpected chemical reaction products.

Although geochemical reactivity of tailings is often considered a liability, there are some cases where the benefits reportedly outweigh the liabilities. For example, Patton (1952) reported that sulfide-mineral oxidation enhanced the strength of tailings backfill at Noranda, Quebec, and was thus to be encouraged. This benefit of oxidation was still promoted in the 1980's (Nantel and Lecuyer, 1983), but apparently became a liability by the 1990's. Wheeland and Payant (1991) state, "There is currently a legitimate reluctance on the part of mine operators to place reactive material underground due to the uncertainty of the outcome. For example, Brunswick Mining used some high sulphide slimes as uncemented backfill in the upper section of the mine two decades ago, and the resultant 'fire' is still generating sulphur dioxide which is vented via a surface stack, despite herculean efforts to bulkhead the area 'air tight'".

By looking at the chemical degradation of uranium isotopic components within uranium ores found around the world, we may be better able to estimate the course of degradation of uranium mill tailings placed underground and the movement of radioactive elements within the underground environment. Tailings placed underground are undoubtedly in a very different physical form than the original uranium ore, but most radionuclide constituents of the original ore still remain. These topics are examined further in the following subsections.

4.1.2 Degradation of water quality in placed tailings and surrounding groundwater system

Contamination and geochemical degradation of water quality in a surrounding groundwater system can be a disadvantage of underground placement, and the potential for such degradation should be considered prior to placement. Nearly half (43%) of questionnaire respondents (Appendix B) expressed similar thoughts, which was the highest level of response for any specific factor in surface retrieval and underground placement (Table 1-1). For this reason, additional details and case studies have been included in this subsection. Both radioactive and non-radioactive contaminants can be a concern, and they are both examined here.

Firstly, it is important to note that, while an underground mine is being dewatered, the potential for contamination of the surrounding groundwater system from backfill material is very low, as most water is usually pumped to the surface and treated (Thomson and Heggen, 1982). The long-term effect of backfilling underground stopes with uranium tailings, however, may be quite different as the mine floods.

Weber (1990) noted that the U.S. Nuclear Regulatory Commission considers groundwater pathways as the most important vehicle for radionuclide transport from a low-level radioactive waste site. This concern over the groundwater pathway is echoed by many authors, as demonstrated below. The inter-relationship between an underground tailings management system and the adjacent groundwater system can be extremely complex depending on the geologic and hydrogeologic conditions of the mining area. Interactions between a surface tailings impoundment and groundwater were outlined by Taylor (1980), and are presented in Figure 4-1. These interactions, in some respects, are similar to tailings disposed underground in shallow workings and can thus be used as a general guide for underground assessments. In fact, Taylor (1980) explained that the main purpose of any tailings management system is to control the migration of adverse contaminates into the surrounding ground and surface water systems.

The Nuclear Regulatory Commission requires that an assessment of current and future groundwater and surface resources be undertaken at a low-level radioactive waste site (Weber, 1990). Furthermore, the Commission requires applicants who wish to dispose of low-level radioactive wastes underground to determine the main processes for radionuclide transport in the groundwater system, such as molecular diffusion. The data required to ascertain the dominant transport process are to be gathered from using hydrodynamic, hydrogeochemical, stable-isotope and radioisotopic methods, in association with the chemical and physical characteristics of the host rock and waste material.

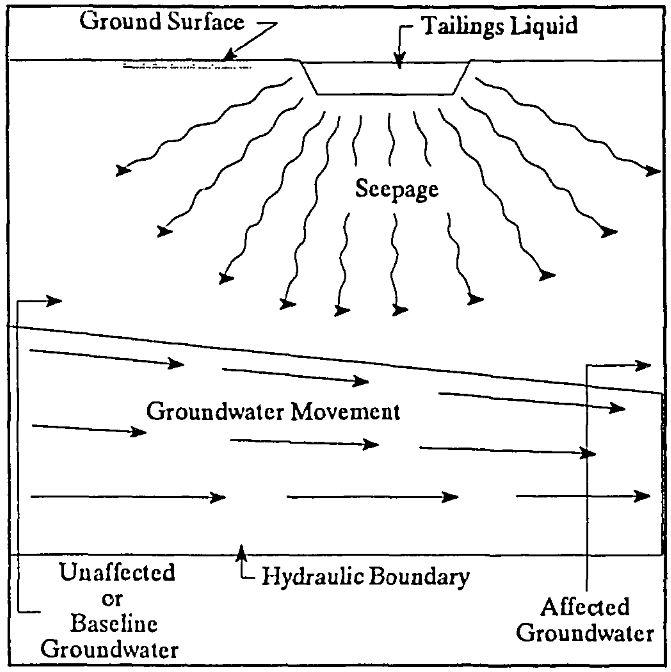

FIGURE 4-1. Contaminant Seepage from Subsurface Tailings (Taylor, 1980).

Public exposure to contaminants from underground placement of waste materials must be carefully considered during the feasibility stage of an assessment into disposal options. Contamination of surrounding drinking water wells or surface springs must be investigated to ensure no adverse impacts to the biosphere or health hazards to human/animals/plants will occur (Schneider, 1988). A full understanding of the regional and local hydrogeologic systems along with the tailings composition and reactivity with host rocks must be compiled to ensure long-term waste confinement in the tailings storage area.

The purpose of placing uranium tailings underground is to provide isolation of the material from surficial natural earth processes and exposure of the biosphere to the waste. West et al. (1991) explained that the main "processes which will affect containment and behavior of the [radioactive] waste are dominated by the behavior of groundwater in the host rock and other rock formations above and below a repository. The movement of groundwater through a repository will result in leaching and mobilization of radionuclides". The impact of radionuclide movement is dependent on the nature of the waste materials and the host rock, as well as the distance to the biosphere, the velocity of movement through the surrounding rock, the geochemistry and physical properties along their path of migration. West et al. (1991) discussed some of the geochemical and biological conditions which can affect the migration of radionuclides, including:

❶ physical sorption such as molecular filtration, ion exclusion and diffusion into "dead-end" pores;

❷ "direct chemical reactions with rock surfaces involving physical adsorption, chemical adsorption or direct incorporation into the rock surface";

❸ "indirect chemical reactions such as precipitation"; and,

❹ microbial mobilization of and uptake of metals.

Similarly, Dowse et al. (1975) indicated that the effect of pollutants and their migration and persistence in a groundwater system are controlled by:

❶ the hydrogeologic conditions such as lithology, aquifer characteristics, hydraulic gradients, porosity, permeability, volume, etc.;

❷ physico-chemical conditions such as viscosity, density, adsorptive, ion-exchange characteristics of the groundwater and contaminants, and molecular diffusion, hydrolysis, oxidation/reduction reactions; and in some circumstances,

❸ biological conditions including aerobic, anaerobic, bacterial food supply availability, and biodegradation processes.

Like many others, McPherson (1984) felt that in most cases groundwater is a primary pathway of radionuclide transport from a natural or artificial source through underground strata. The rate of radionuclide transport was attributed to various factors including release rate, groundwater movement, adsorption/desorption phenomena, diffusion within the rock matrix, radioactive decay, chemical reactions and ion exchange.

Many of the previous references viewed radionuclides as dissolved constituents of gases in groundwater. Vilks et al. (1993) showed that radionuclides can also migrate as colloids and fine particles in groundwater systems.

Although methods of tailings placement are discussed in Section 4.2.2, one important note on placement through boreholes is warranted here. Contamination of shallow groundwater could result from the deposition of wet slurried tailings through boreholes (Kegel, 1975). Cementing and sealing of the borehole casing through all aquifers and into impermeable strata are critical in preventing any hydraulic connections with shallow groundwater. Waste characteristics that must be addressed before injection include: (a) the compatibility with the formation fluids, (b) reactions between the waste and the host formation must form products that are soluble in the wastewater, (c) insoluble solids must be removed prior to injection of the waste stream, (d) any wastes that are exposed to the atmosphere must have any biological growth removed prior to injection (Sethness and Holmes, 1979).

4.1.2.1 Retardation

From the perspective of groundwater-quality degradation, the general mechanism opposing degradation and migration of contaminants is usually referred to as "retardation". "Retardation is the effect of all processes which hinder the mobility or migration of chemicals in solution or in suspension relative to the mobility or migration of water itself. In the context of radioactive waste disposal in geologic environments, consideration of retardation is an essential part of the assessment of the barrier provided by geologic formations" (Intera, 1992). Intera (1992) grouped most retardation processes into four main mechanisms:

❶ Precipitation/Co-precipitation - Dissolution: bulk phase reaction processes including chemical substitution and isotopic exchange reactions.

❷ Sorption: the dominant chemical retardation mechanism with regards to radioactive waste. Sorption can also include physical adsorption, electrostatic adsorption (ion exchange) and specific adsorption.

❸ Diffusion: the dominant physical retardation mechanism. Diffusion can also include pore diffusion (including diffusion into immobile porosity within fractures and within the rock matrix), surface diffusion and solid diffusion.

❹ Filtration: physical retardation processes in which the geologic matrix (medium) preferentially retains waste solute over water held within the pore space (fluid).

The impact of low-level radioactive waste on groundwater chemistry is affected by: 1) the total water available for contact with the waste; 2) the mobility of various components of the waste (i.e. radionuclides, metals, acid products); 3) the physical form of the waste (Fuhrmann and Colombo, 1990). When radioactive materials such as uranium tailings are placed underground, radioactive gases may migrate throughout the surrounding unsaturated zone into contact with ground water (Striegl, 1990). There is then a potential for uptake of available radionuclides by biological organisms through contact and intake of contaminated waters (Striegl, 1990).

Thomson and Heggen (1982) also discuss the interactions of contaminants dissolved in groundwater within the rock/soil matrix, which lead to retardation. "Potential interactions include ion exchange, adsorption, and microbial uptake,...the partitioning between the solid and liquid phases (ie. similar to movement of a solute front in a chromatograph) is described by distribution coefficients which can be used to calculate retardation velocities".

To accurately determine the effect of radionuclide contamination on a groundwater system, Kilborn and Beak (1979) compiled a detailed list of relevant data which should be collected (Table 3-4). To then model the effects of radionuclide contaminant transport in a groundwater system, Kilborn and Beak (1979) assembled a list of requirements (Table 3-5).

4.1.2.2 Natural analogs

Before examining retardation processes further, an examination of water quality in and around uranium-ore deposits, as natural analogs to uranium tailings, provides some indication of very-long-term behaviour. For example, uranium ore discovered in Oklo, Gadon, formed over 2,000 million years ago and has much less ^{235}U relative to ^{238}U than expected (Chapman and McKinley, 1990). Through investigations into the composition of this ore, scientists postulated that the lower levels of ^{235}U were caused by "nuclear fission in a natural chain reaction, accelerating the disappearance of the lighter isotope" (Chapman and McKinley, 1990). They also found than even under temperatures of 600 degrees and within radiation damaged rocks, many radionuclides remained in place or within a short distance of the deposit before decaying to stable isotopes. Other radionuclides such as noble gases, and isotopes of elements which were highly soluble or poorly sorbed onto the surrounding rocks had migrated away.

Another example of uranium ore being able to provide data on the long-term stability of radionuclides is the Cigar Lake deposit in northern Saskatchewan. The deposit is 450 m below the surface and was formed 1,300 million years ago. Although this is a very rich ore deposit containing an average of 14% U_2O_3, the surface water and subsurface groundwater systems did not reflect the presence of the ore body. Even though the rock was estimated to have been saturated with water for approximately 1,000 million years, the uranium had not migrated significantly or degraded surrounding water quality (Chapman and McKinley, 1990).

4.1.2.3 Sorption and ion exchange

There is some disagreement in published literature on whether sorption or mineral precipitation/dissolution (Section 4.1.2.4) is the primary retardation mechanism in groundwater systems surrounding underground mines. In all likelihood, the answer is site-dependent. In any case, relevant case studies and theories on sorption are first presented below, followed by mineral solubility.

Although contaminants may be released from tailings, some portion is retained by the tailings solids and another portion is retained by the surrounding rock. The rock is capable of retaining some contaminants from the groundwater through processes such as mineral precipitation, adsorption, and ion exchange. Also, Birgersson and Neretnieks (1990) felt that the potential uptake or diffusion of radionuclides into a surrounding rock matrix was one of the most important retardation mechanisms for radionuclides carried in groundwater away from a waste disposal repository. Kamineni et al. (1980) investigated the fracture filling material within the rock mass of the Atikokan area in northwestern Ontario. Preliminary investigations into the mineral composition of the fracture filling material can be used to determine groundwater and radionuclide interactions with the host rock mass during underground storage of waste materials.

The greatest attention for controlling and predicting water-quality degradation has been focussed on adsorption. Adsorption is typically quantified through a sorption ratio (R_d) or equilibrium distribution coefficient (K_d) with common non-S.I. units of mL/g:

$$R_d = K_d = \text{Solid-phase amount per unit mass of solids/Aqueous-phase amount per unit volume of water} \qquad (4\text{-}1)$$

There is a great deal of published literature on adsorption coefficients (e.g., Table 4-2), but the most recently compiled and reviewed would probably be contained in a contract report currently in preparation for the Atomic Energy Control Board of Canada.

Bird and Fung (1979) conducted and reported on preliminary investigations into the ion exchange capacity and/or Cs^+ and Sr^{2+} adsorption capabilities of feldspars. Ion exchange is similar to sorption, except sorption addresses only one element whereas exchange addresses the opposing behaviour of two or more elements. Bird and Fung (1979) concluded that "feldspar

Table 4-2
Examples of Adsorption Coefficients for Selected Elements
(as mL/g, adapted from Brookins, 1983)

Element	Devitrified Tuff	Zeolitized Tuff
Barium	adsorption = 430-1500 desorption = 440-1300	adsorption = 15,000-130,000 desorption = 34,000-190,000
Strontium	adsorption = 53-190 desorption = 56-200	adsorption = 1800-20,000 desorption = 2700-20,000
Uranium (air)	adsorption = 1.6-2.2 desorption = 6-13	adsorption = 2.3-5.1 desorption = 15

stability in aqueous systems at low temperature is limited and a number of complex dissolution reactions take place. Dissolution is initially incongruent, but over long periods of time it becomes congruent and is surface-reaction controlled". They wrote that "feldspar minerals are cation exchangers with low total ion-exchange capacities,...but the hydrous oxide and aluminosilicate alteration products of feldspar dissolution are also cation exchangers and have a greater capacity per unit mass than the original feldspars".

The capacity for sorption, where sorption means "any process that removes a trace ion from solution onto the rock or mineral surface such as ion exchange, ion substitution, chemisorption, precipitation, etc.", of primary and secondary igneous rocks and minerals was also investigated by Ticknor et al. (1985). Ticknor et al. (1985) used tracers of radionuclides associated with high level radioactive wastes (^{90}Sr, ^{137}Cs, ^{147}Pm, ^{241}Am, and ^{75}Se) and found that the alteration minerals associated with primary and secondary minerals and rocks played an important roll in the sorption abilities of the sample thin sections. Table 4-3 lists cation exchange capacities taken from Ticknor et al. (1985).

Ticknor et al. (1985) also reported that iron oxides such as magnetite and hematite will, under anaerobic conditions, generally retard migration of the radionuclide tracers better than

Table 4-3
Cation Exchange Capacities for Selected Minerals
(from Ticknor et al., 1985)

Mineral	CEC[1] (meq kg^{-1})	CEC[2] (meq kg^{-1})
Albite	3.7	4.1
Apatite	0.5	
Biotite	17.0	
Calcite	0.2	1.0
Chlorite	50.0	30.0
Epidote	6.0	
Hematite	0.5	
Hornblende	2.5	4.1
Kaolinite	28.0	
Magnetite	0.4	
Microcline	3.7	4.2
Montmorillonite	800.0	
Muscovite	52.0	
Olivine	1.8	3.3
Quartz	0.2	0.7

[1] SKBF Technical Report 83-64 (Particle size 0.44-0.063 mm; pH - 8.0)
[2] Determined at WNRE by sodium saturation (Particle size 0.106-0.180 mm; pH -8.0)

ferromagnesian minerals. The sorption capacities of minerals such as biotite were affected by their orientation and available surface area, ie., along cleavage planes.

Grondin and Drew (1988) conducted larger scale investigations using radionuclide tracers with igneous rock and mineral samples. They also found that ^{137}Cs was retarded by alteration products on fracture surfaces. Iron oxyhydroxides had the most significant impact on radionuclide migration by sorption through ion exchange and chemisorption. The initial studies by Grondin and Drew (1988) dealing with radionuclide sorption were to be followed by large scale testing at the Large Block Radionuclide Migration Facility at Atomic Energy of Canada Ltd.'s Whiteshell facility (Drew et al., 1990). This facility was developed to conduct experiments of radionuclide migration through fractures. The scale of the experiments is between laboratory and field scale, and allows for experimentation with radionuclides under controlled conditions that are not possible in the field due to the possibility of contamination. This facility should be able to supply important information on the interactions between radionuclides and fractured rock matrices.

Through field and laboratory experiments carried out on weathered and unweathered uranium ore in Northern Australia by Airey (1986), some key conclusions were found which facilitated modelling of radionuclide transport in weathered ore. These conclusions can be related or adapted to changes within uranium tailings once placed underground. Airey (1986) found that various zones of leaching and deposition occur due to groundwater movement through, and oxidation within, the ore. Airey (1986) found that uranium and thorium concentrations varied between amorphous iron (ferrihydrate), goethite and clay/quartz components of weathered ore. Although iron comprised only 1-4% of the weathered ore, uranium and thorium were concentrated on the Fe phases. This observation is notable in that the iron phase which immobilized the uranium and thorium only forms under oxidized conditions, which contradicts other studies which called for reducing conditions for immobilization. Radium on the other hand was concentrated in the clay/quartz phases of the weathered ore and its activity exceeded that of the parent thorium.

As a non-tailings illustration of variations in element-specific sorption, Blaylock and Fore (1979) reported on data collected at the Hanford site, Washington. The data showed that ^{90}Sr, ^{137}Cs, and ^{239}Pu had good sorption characteristics and could be successfully removed before reaching groundwater. ^{106}Ru, ^{60}Co, ^{99}Tc, and ^{3}H had poor sorption characteristics and moved through the ground into the groundwater where their concentrations were reduced by radioactive decay, ion exchange, diffusion, and hydrodynamic dispersion.

4.1.2.4 Mineral precipitation/dissolution

Snodgrass et al. (1982) modelled surface uranium tailings impoundments and various close-out options to determine their effectiveness in preventing the release of radionuclides and other contaminants into the biosphere. Snodgrass et al. (1982) estimated that 90% of radionuclides released from tailings are in the form of coprecipitants or precipitates, and are absorbed onto or are present in the leached material (Figure 4-2). Consequently, mineral precipitation/dissolution can sometimes exceed the effects of sorption and ion exchange on water quality (Morin et al., 1988a and 1988b).

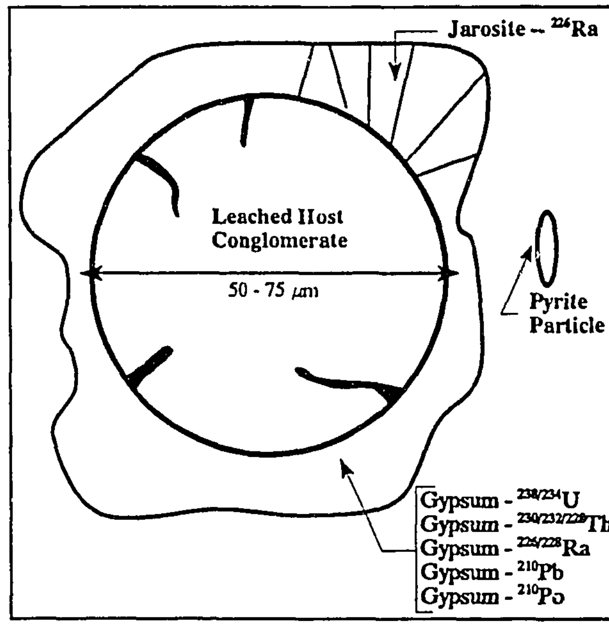

Figure 4-2: Conceptual Sketch of Uranium Tailings Particle (from Snodgrass et al., 1982).

Radium is thought to often coprecipitate with gypsum and other metal sulfides such as jarosite ($KFe_3(SO_4)_2(OH)_6$), while thorium precipitates by neutralization in the milling process leading to it's association with gypsum (Snodgrass et al., 1982). Snodgrass et al. (1982) indicated that the main mechanisms at work in a tailings mass are "precipitation-dissolution reactions, pyrite oxidation, adsorption onto surfaces, solids state diffusion, pore water diffusion, pore water complexation and hydrological transport".

Fleer and Johnston (1986) compiled data for the Atomic Energy of Canada Ltd. on the solubility and dissolution kinetics for minerals in granitic and gabbroic rock. Their report supplied experimental data on mineral-solution interactions, the 57 minerals listed are primary, secondary or alteration constituents within granitic and gabbroic rocks. Fleer and Johnston (1985) wrote that mineral-solution interactions involve both mineral dissolution and precipitation reactions, where precipitation may result in the original solid being reformed, or the precipitation of a secondary solid of lower solubility under existing conditions. Fleer and Johnston (1986) also supplied data on the rate-controlling mechanisms for dissolution in pure water.

Before quality data on mineral solubility can be produced, the basic types of mechanisms acting within a particular reaction sequence should be defined, and experiments to verify the theory and attain representative data then designed. Walton (1982) reported on the design of laboratory experiments to measure radionuclide-geologic material interactions. The focus of Walton's (1982) work was the effect of hydraulic conditions on the resulting reaction data. This type of experimental design is invaluable to other researchers when trying to compare data from various sources, and when trying to ascertain the validity of those data for use in their work.

Instead of experimental testwork, mineral precipitation/dissolution can sometimes be deduced from in-field water chemistry. Johnston (1982) compiled data on groundwaters of granitic and gabbroic rock origins. Johnston (1982) found significant variation in the data due to sampling technique, analytical technique, the suite of elements analyzed, and depth of sampling. Johnston (1982) noted that establishing a "typical chemistry" for groundwaters from granitic and gabbroic origins was very "difficult, if not impossible". Each geologic system is unique, and the fracture system and depth of sampling had a great effect on the concentrations of commonly analyzed elements. Therefore, the assessment of mineral solubility from water chemistry should be based on site-specific data rather than "typical" concentrations.

Garisto and Garisto (1985) indicated that high-level uranium waste material containing significant UO_2 can become unstable in the presence of carbonates and phosphates due the complexing with UO_2 to form uranates. Sulfates and chlorides also act as complexing anions with the UO_2 to form more stable species. Some thermodynamic data complied by Garisto and Garisto (1985) is presented in Table 4-4 to illustrate one complexity involved in assessing mineral solubility and related aqueous complexing.

The dissolution of uranium minerals from tailings solids into the surrounding groundwater system after underground tailings placement can be a significant issue. A concern arises from the fact that some uranium-ore deposits are associated with aquifers, implying groundwater transport and mineral precipitation were important in their formation (Thomson et al., 1986). In these situations, uranium ore deposition is thought to result from the aqueous transport of oxidized, soluble uranium species. The uranium species are transported to chemically reducing (oxygen-depleted) zones in which precipitation due to reduction of uranium minerals occurred (Thomson et al., 1986). Similarly, if oxygenated acid-generating uranium tailings are placed underground, a redox-controlled migration front similar to those found in nature may develop.

> "Water rich in oxygen moved downwards from the surface, oxidizing iron in the volcanic rocks from Fe^{2+} to Fe^{3+}. As the redox front moved the uranium in the rock dissolved under the oxidizing conditions and was precipitated on the reducing side of the front concentrating the uranium into an ore body" (Chapman and McKinley, 1990).

Consequently, oxidizing conditions in placed tailings could dissolve some radioactive components, groundwater movement could transport them away from the mine, and mineral precipitation and other retardation processes could then deposit them elsewhere.

Thomson et al. (1986) felt that "two distinct diagenetic processes" affect the pore water chemistry within backfilled tailings. The two processes are supersaturated mineral precipitation resulting in chemical reduction, followed by the precipitation of constituents which were originally oxidized. Depending on whether the conditions are oxidizing or reducing, precipitation or dissolution of inorganic backfill chemical constituents can result. Mixing of backfill slurry waters with native/natural groundwaters, and backfill compaction are two more

Table 4-4 Thermodynamic Data (from Garisto and Garisto, 1985)			
Solution Species	ΔG°_{f} (kJ mol^{-1})	S° (J·K^{-1} mol^{-1})	$C_{p}{}^{\circ}{}_{298}{}^{473}$ (J·K^{-1} mol^{-1})
$U(OH)_2^{2+}$	-991 ± 10	-69 ± 40	-1
$U(OH)_3^{+}$	-1211 ± 10	19 ± 40	74
$U(OH)_4$	-1424 ± 10	50 ± 40	---
$U(OH)_5^{+}$	1630.6 ± 1.4	94 ± 6	-74
$UO_2(HPO_4)_2^{2-}$	-3236 ± 5	126 ± 200	354
HSO_4^{-}	-755.9	120	30
PO_4^{3-}	-1018.7	-222	-502

processes which are also dependent on chemical conditions in the underground. Thomson et al. (1986) compiled a list of typical dominant species present under each of these conditions in an aqueous system (Table 4-5). From data presented in Table 4-5, they concluded that most transition metal species are insoluble, and therefore immobile under reducing conditions. Maintenance of reducing conditions within a backfill environment, in general, would therefore assist in preventing the dissolution and mobility of many inorganic contaminants (Thomson et al., 1986). However, it is also important to realize that iron and metals normally coprecipitated with, or adsorbed by, ferric-iron precipitants will be mobilized under reducing conditions (Morin et al., 1988a).

AECL has conducted research into uranium ore formation and groundwater flow of the Athabasca Basin in northern Saskatchewan (Cramer, 1986). The basis of the research was to gain some insight into the impacts of placing high-level waste underground and radionuclide escape from subsequent container failures. By using events which occurred in the geologic past during the sandstone-hosted uranium ore deposit formation, it was estimated that a clay layer surrounding the uranium ore deposit buffered contaminant migration within the ground water

Table 4-5
Typical Dominant Species of Inorganic Constituents Under Oxidizing and Reducing Conditions in Aqueous Systems
(from Thomson et al., 1986)

Metal	Oxidizing Condition	Reducing Condition
As	$H_2AsO_4^-$	$As_2S_{3(s)}$, $AsS_{(s)}$
Cd	Cd^{2+}, $CdCO_{3(s)}$	$CdS_{(s)}$
Cr	CrO_4^{2-}	$Cr_2O_{3(s)}$
Cu	Cu^{2+}, $CuO_{(s)}$	$CuS_{(s)}$
Pb	$PbCO_{3(s)}$	$PbS_{(s)}$
Hg	$HgO_{(s)}$	$Hg^0_{(s)}$
Ni	Ni^{2+}	$NiS_{(s)}$
N	NO_3^-	NH_4^+, $N_{2(g)}$
Se	SeO_4^{2-}, $HSeO_3^-$	$Se^0_{(s)}$, $FeSe_{2(s)}$
Ag	$AgCl_{(s)}$	$Ag_{(s)}$, $Ag_2S_{(s)}$
Zn	Zn^{2+}, $ZnO_{(s)}$	$ZnS_{(s)}$

system. Uranium precipitation was estimated to have happened at the interface of an oxidizing uranium-bearing solution and a reducing solution (Cramer, 1986). If the host rock of an underground disposal site for uranium tailings produces a reducing environment for the groundwater to flow through, uranium contaminants which may have dissolved in the groundwater would theoretically precipitate out, thereby limiting their migration from the disposal location.

Cramer (1986) reported on the diamond drill hole groundwater samples which intersected mineralization at the Dawn Lake uranium deposit of Saskatchewan. The range in concentration of various chemical parameters are reproduced in Table 4-6. When these types of data are plotted on a uranium solubility diagram, as Cramer (1986) did, the solubility and potential mobility of uranium can be estimated from the pH of the groundwater, and will show the chemical species which will be present at a given pH.

Slurry water retained in the porewater of the placed tailings and leachate from the tailings act as sources of contaminated water. If the tailings and associated porewater have high levels of sulfate, then precipitation of gypsum can occur. This precipitation can be beneficial through reductions in porosity and hydraulic conductivity of the tailings by a process similar to cementation found in old tailings (Thomson and Heggen, 1982). Coprecipitation of other minerals may also take place concurrently.

Table 4-6
Chemical Composition of Groundwaters from Dawn Lake Uranium Deposit, Saskatchewan
(from Cramer, 1986)

Parameter	Typical Range
Sodium (mg/L)	2.3 - 4.8
Potassium (mg/L)	1.26 - 1.67
Calcium (mg/L)	0.34 - 7.9
Magnesium (mg/L)	0.25 - 4.8
Iron (mg/L)	< 0.06 - 3.23
Nickel (mg/L)	< 0.05
Arsenic (mg/L)	< 0.002 - 1.9
Lead (mg/L)	< 0.06 - 0.08
Cobalt (mg/L)	< 0.1
Vanadium (mg/L)	< 0.01
Fluoride (mg/L)	0.05 - 0.23
Chloride (mg/L)	0.22 - 2.75
Phosphorus (mg/L)	< 0.25
Nitrate (mg/L)	< 0.1
Sulphate (mg/L)	0.60 - 6.41
Bicarbonate (mg/L)	7 - 61
Carbonate (mg/L)	< 2
Uranium (μg/L)	0.15 - 17.03
pH (field)	5.93 - 7.46

If the native groundwater "has a substantially different oxidation potential (Eh) than that of the backfill fluids, ...then under thermodynamic considerations ... many of the contaminants will precipitate under ... reducing conditions" (Thomson and Heggen, 1982). Contaminants that may remain in solution include uranium as a carbonate complex, selenium, and manganese as well as reduced forms of sulfate. One concern over cementation by mineral precipitation is the opportunity for later mineral dissolution and mobilization of contaminants. Mineral precipitation and the conditions for later dissolution can be evaluated with computer programs such as MINTEQ (Allison et al., 1990).

In general, mineral precipitation/dissolution is often recognized as an important mechanism for mobilizing or retarding radioactive and non-radioactive elements. However, as explained above, there is disagreement in the literature on whether oxidized or reduced conditions are better for minimizing contaminant migration through mineral reactions. The disagreement is probably due to variable site-specific conditions.

4.1.2.5 Diffusion

Another process that can affect contaminant migration and groundwater quality, particularly in the absence of water movement through unfractured rock, is diffusion. Diffusion has only minor effects over short periods of time, but becomes significant over longer times. Katsube et al. (1986) indicated that the migration of radionuclides in groundwater through diffusion takes place in a complex micropore network within the host rock, as well as onto fracture surfaces.

Birgersson and Neretnieks (1990) designed and executed a series of experiments over a period of 3 years to simulate the diffusion rate of radionuclides through undisturbed rock mass. This project was initiated to determine the diffusion coefficients for tracer movement in granite and other crystalline rock. Tracers were injected at a constant pressure over various lengths of time, and the experiments were designed to evaluate the rate of diffusion and advection of the tracers through disturbed rock into the surrounding undisturbed rock matrix. The tracers were found to migrate beyond the disturbed area into the presumed undisturbed area with diffusion coefficients and hydraulic conductivity values varying up to an order of magnitude over very short distances in the rock matrix.

When dealing with the release of radionuclides from high-level waste materials after groundwater has breached the storage vault, two radionuclide "escape mechanisms" are involved. These mechanisms were described by Garisto and Garisto (1985) as being:

❶ rapid release into intruding groundwater by fission products such as cesium and iodine from around the grain boundaries; and

❷ dissolution of the waste material which controls most of the release of radionuclides, this process is controlled by the individual solubilities of the radionuclides present.

Both release mechanisms can be regulated by the diffusion coefficients of the radioactive constituents and/or by groundwater velocity and conditions such as pH, temperature, composition, oxidation potential (Garisto and Garisto, 1985).

4.1.2.6 Case studies in Elliot Lake and the Stripa Mine in Sweden

For decommissioning of underground uranium mines in Elliot Lake, the water-quality effects have been considered. For example, Senes Consultants (1991a), in a Decommissioning Proposal for Rio Algom's Panel Mine, indicated that variations in the flow and changes in the quality of surface and ground waters were all potential environmental concerns following mine flooding. The stability of a boundary pillar which the Panel Mine shares with the Denison Mine, was also felt to be a major concern after closure.

The Panel Mine's underground workings are under Quirke Lake, and "under [a] worst case scenario" a release of contaminated water would enter the lake. Water quality in the Panel Mine collected, from January to August 1990, included average contaminant levels of 1443 mBq/L for ^{226}Ra, 1712 mg/L for SO_4, 1350 mg/L acidity, and 408 mg/L for Fe. However, this chemistry is not expected to have a significant impact due to the nature of Quirke Lake (Senes Consultants, 1991a).

Before conducting an assessment on the impact of placing uranium tailings underground into mined-out workings, a knowledge of the background conditions of the area preferably before, during and after cessation of mining operations is needed. Boyd et al. (1982), in their hydrogeologic investigations of the Stanleigh Mine area, Elliot Lake, Ontario reported some typical bedrock groundwater chemistry for the Crotch Lake area near the Stanleigh Mine (Table 4-7).

Table 4-7
Typical Bedrock Groundwater from Crotch Lake Area, Ontario
(from Boyd et al., 1982)

Parameter	Typical Range
pH	6.5 - 7.5
Calcium (mg/L)	20 - 60
Magnesium (mg/L)	3 - 7
Sodium (mg/L)	5 - 20
Potassium (mg/L)	1 - 4
Chloride (mg/L)	1 -20
Sulphate (mg/L)	20 - 100
Silica (mg/L)	3 - 8
Alkalinity as HCO_3 (mg/L)	50 - 150
^{226}Radium (pCi/L)	2 - 12
Iron (mg/L)	0.01 - 0.2
Manganese (mg/L)	0.01 - 0.2
Copper (mg/L)	0.005 - 0.02
Lead (mg/L)	0.005 - 0.02
Zinc (mg/L)	0.02 - 1.0
Nitrate as N (mg/L)	0.2 - 0.6
Ammonia as N (mg/L)	0.4 - 2.0

The Stripa Project in Sweden was the site of extensive investigations into the potential impacts on groundwater of placing high-level uranium waste underground. The research has been ongoing for approximately 20 years, and tends to indicate that acidic oxidizing conditions in the underground "enhance mobility of actinide elements" (Wollenberg and Flexser, 1985).

Stripa geology is such that naturally occurring uranium is found within the fracture fillings of the regional pluton (Wollenberg and Flexser, 1985). The association of uranium with fracture filling minerals instead of the more usual presence in "non-opaque accessory" minerals such as sphene, zircon, monazite and apatite seems to imply uranium mobility.

Uranium concentrations in rock at the Stripa underground site show a depletion in uranium concentration at the surface (mean 27 ppm vs. mean 37 ppm underground), and a slight but steady increase with depth after 410 m (Wollenberg and Flexser, 1985). Uranium concentrations in groundwater showed a significant increase at 150-200 m, decreasing steadily with depth to levels consistent with those above 150 m.

Eh and pH data at Stripa suggest that uranium "may be removed from nearby surface rock where slightly acidic and oxidizing conditions prevail, and transported in fractures by groundwater to deeper zones where more reducing conditions favor its concentration in fracture-filling material" (Wollenberg and Flexser, 1985). "In contrast to the relatively oxidizing-acidic nature of the groundwater at depths to approximately 120 m, more reducing and alkaline conditions exist below approximately 310 m. The gradational increase in U with depth in the rock below 410 m then might result from redistribution and concentration in fracture filling material by the groundwater system" (Wollenberg and Flexser, 1985).

"The strong association of uranium with chloritic fracture filling material suggests that it is the principal source of ^{222}Rn observed in water of some Stripa boreholes. In hydrologically saturated crystalline rock masses, where the groundwater systems are confined to the joints and fractures, the presence of the uranium daughter ^{222}Rn in the water may serve as a natural tracer to locate fractures along which significant flow is occurring and to measure their flow rates. At Stripa, Rn activity concentrations in the water from some fractures reach 1 μCi/L" (Wollenberg and Flexser, 1985).

Witherspoon et al. (1981), during experimentation at the Stripa minesite, stated that the there are three basic properties of underground host rock that affect the movement of radionuclide contaminants from an underground disposal site, these are: (1) permeability or hydraulic behaviour of the host rock; (2) the effective porosity of the host rock; and, (3) the host rock's capacity for sorption of the radionuclide contaminants.

4.1.2.7 Modelling

Based on the previous information and related studies around the world, various types of models have been developed or adapted to assist in the prediction of underground water chemistry. Although geochemical models are the focus of this section, most models combine geochemistry and water movement to simulate contaminant migration. Consequently, both aspects are discussed below, avoiding repetition in Section 4.1.3.6.

To determine the migration characteristics of radionuclides in an underground environment, Laul et al. (1988) looked at the movement and presence of naturally occurring radionuclides over geologic time scales not possible to duplicate in laboratory or in-situ testing. Laul et al. (1988) examined the solution transport of radionuclides within a geologic environment basing the investigations on the disequilibria of radioisotope pairs, or the concept of parent to daughter relationships. By knowing the decay rates of various radionuclides and their solubilities in an aqueous medium, the ratios of parent to daughter were used to determine the rate of radionuclide movement. Parent to daughter radionuclide ratios close to "unity" can indicate that migration has taken place especially if one of the isotopes is more soluble than the other. One problem with this method is the analytical limitations in accurately determining the concentration of specific radionuclides.

If precise measurements could be made, Laul et al. (1988) felt that the model they used could supply information on past movements of naturally occurring radionuclides over a geologic time scale (thousands of years to tens of millions of years). These data could then be used to characterize adsorption/desorption (Section 4.1.2.3) between the strata and its natural waters for natural radionuclides, and thereby allow the calculation of chemical migration for these radionuclides within the geologic environment.

McKeon and Nelson (1984) developed a model based on analytical equations for estimating "consequences" of groundwater-borne contaminant migration. The model simulates movement of groundwater and contaminants from an underground stope backfilled with uranium tailings to a downgradient water-supply well.

Tang et al. (1980), in describing the numerical modelling of contaminant migration in a fractured groundwater system, recognized the following processes as key factors in the transport of contaminants and groundwater in the fractured system:

❶ "advective transport along the fractures;

❷ longitudinal mechanical dispersion in the fractures which is a combined effect of parabolic velocity due to mixing in the direction of the fracture axis and roughness of the fracture walls;

❸ molecular diffusion within the fractures in the direction of the fracture axis;

❹ molecular diffusion from the fracture into the matrix;

❺ adsorption onto the face of the matrix;

❻ adsorption within the matrix; and,

❼ radioactive decay".

This type of modelling allowed for the calculation of the rate of migration of radioactive contaminants and the extent of their dispersion within the underground fractured groundwater system.

Wuschke et al. (1981) modelled the potential interaction and migration of radionuclides from high-level radioactive waste materials to be stored in granitic rock of the Canadian Shield. The report discussed the movement of radionuclides into the rock matrix, groundwater, and buffer material surrounding the waste, by diffusion and advection mechanisms; the subsequent retardation of the radionuclides by the geosphere; and migration into the biosphere. Their approach to modelling, although simplified, may be applicable to the disposal of uranium tailings in underground workings.

Hajas and Heinrich (1987) reported on the results of running the SYVAC2 model developed to predict the movement of radionuclides through the geosphere from a waste repository. Although developed for nuclear wastes, the model could potentially be used or altered to predict the migration of radionuclides from uranium tailings in the geosphere. The model predicts the concentration of a radionuclide at a defined distance from a repository in a specified rock matrix. This model was compared to results from other models and was found to perform satisfactorily. The model SYVAC2 (Hajas and Heinrich, 1987) is an update to the original stochastic SYVAC geosphere model described by Heinrich in 1984, which was comprised of a geosphere, biosphere and vault submodel. SYVAC3 allowed for the development and use of probability density functions, the necessary parameters of which were outlined by Stephens et al. (1989).

Tabori and Wilkinson (1984) reported on a radionuclide/groundwater-transport model called WASTE, which simulated radionuclide transport from a high-level radioactive-waste vault into a three-dimensional groundwater flow system. This system was modelled as a network of one-dimensional flowpaths.

Lyon and Rosinger (1980) reported on the initial phases of modelling the migration of radionuclides from a high-level waste repository. The overall assessment included the use of three different pathway analysis models: geosphere, biosphere and potentially-disruptive-phenomena analysis. This approach can be relevant to the underground disposal of uranium tailings. The geosphere analysis conducted by Lyon and Rosinger (1980) included hydrological modelling, which consisted of regional, local and disposal-site conditions (hydrogeologic system within the backfill material), as well as geochemical modelling which accounted for the effects of variations in redox potential, temperature, interactions among dissolved species, and non-equilibrium reactions. The biosphere model estimated the movement of radionuclides within the surface or near surface environment to final contact with man (ie. food chain, water and air exposure). The potentially disruptive phenomena analysis evaluated the potential for disturbance and/or dispersion of the wastes by: (1) a man-made event such as an accidental or intentional intrusion; (2) a natural event such as earthquakes, meteorite impact, volcanoes, glaciation, and erosion; or, (3) fracture growth due to stress change with time in the geologic underground environment. Fracture parameters considered in the model were frequency, orientation, length, width, and interconnection (Lyon and Rosinger, 1980).

Halbert et al. (1982) developed a model to determine the effect of placing uranium tailings from the Elliot Lake area of Ontario underwater into nearby Quirke Lake. The model addressed many chemical, physical and biological effects on the water column and on effects on the adjacent Serpent River Basin watershed.

4.1.3 Change in groundwater flow through/from mines

The placement of tailings which form a porous medium in underground workings can be expected to change the physical characteristics of flow through or from the mine. In turn, these changes can alter the quality of water (Section 4.1.2) and air (Section 4.1.4) in the underground environment. The effects are dependent on whether the mine is drained during placement, filling or filled with water after placement, or flooded at the time of placement. Eight percent of respondents to this study's questionnaire expressed a concern over this issue.

In spite of the condition of the mine during and after placement, changes in water flow through or from the mine depend primarily on the physical hydrogeologic conditions of the surrounding rock. The two most important factors are: hydraulic gradient and hydraulic conductivity (Freeze and Cherry, 1979). One compilation of international hydrogeologic data for underground mines is presented by Geologic Testing Consultants (1986) on behalf of AECB.

Hydraulic gradient is the change in hydraulic head between two points in a groundwater system divided by the distance between the points, which characterizes the "driving force" behind the groundwater. Values of gradients typically range over roughly four orders of magnitude (0.0001 to 1.0) and reflect site-specific conditions including recharge and discharge areas and variations in hydraulic conductivity. The gradients can be, and should be, obtained through direct measurements made with piezometers or related equipment.

Hydraulic conductivity (K) reflects the water-carrying capacity of a unit cross-sectional area of rock (1 m^2) under a hydraulic gradient of 1.0. Conductivity typically ranges over 10 orders of magnitude (roughly 10^{-15} to 10^{-2} m/s) and thus can proportionally play a greater role in regulating groundwater flow than hydraulic gradient.

Underground mines can be divided into several basic types based on rock types and their hydrogeologic and mechanical characteristics. One classification from the perspective of nuclear repositories is: salt, granite, shale, basalt, and tuffaceous rock (Brookins, 1983). In addition to differences due to origin, these basic classes differ in characteristics including hydraulic conductivity and geochemical retention (Table 4-8). However, several characteristics can vary greatly with the degree of fracturing. Granites, tuffaceous rock, and basalt are of greatest interest here because Canadian uranium mines are typically associated with these rock types.

In granites, tuffaceous rock, and basalts ("hard" crystalline rock that characterizes Canadian uranium mines), bulk hydraulic conductivity is a composite of the intrinsic conductivity of the rock, which is typically low, and the fracture conductivity, which is typically higher. Thus, bulk conductivity and its control over groundwater flow are often determined by fracture conductivity. In turn, fracture conductivity is determined by several factors such as fracture aperture, smoothness, and interconnectivity to other fractures. The proper delineation of all these factors along all fracture surfaces in the rock mass surrounding an underground mine are obviously beyond current technology and capabilities, as indicated by intensive international studies for high-level radioactive waste repositories. For example, predicted changes in hydraulic conductivity during mining at the Canadian Underground Research Laboratory (URL) in Manitoba, based on computer modelling, were opposite to the observed trends (Section 4.1.3.4). This led to the conclusion that "... the models and/or the codes used do not correctly simulate the physical processes that occur in a fracture subjected to excavation-induced displacements" (Lang, 1989). No less uncertainty could be expected with fractures in operating and filled mines.

Table 4-8
Example of Rock Classification for Underground Mines and Some Generalized Characteristics
(from Brookins, 1983)

SALT DEPOSITS	low water content, low porosity, low hydraulic conductivity, self-healing fractures, radioactive contaminant migration limited physically, low mechanical strength, high thermal conductivity
UNWEATHERED GRANITES	low water content, low porosity, low hydraulic conductivity, non-healing fractures, radioactive contaminant migration limited physically and geochemically, high mechanical strength, relatively heat resistant
SHALES	high water content, high porosity, low hydraulic conductivity, non-healing fractures, radioactive contaminant migration limited physically and geochemically, low mechanical strength, low heat resistance
BASALTS	negligible water content, low porosity, low hydraulic conductivity, non-healing fractures, relatively heat resistant
TUFFACEOUS ROCK	contaminant migration limited physically and geochemically

Consequently, a thorough characterization of groundwater flow at an underground mine now and after placement of tailings cannot be reliably made, and this increases the risk and uncertainty in the estimation of impacts on groundwater flow and chemistry. Such risk and uncertainty should be discussed in a proposal for underground disposal of tailings. Perhaps a more empirical approach could be implemented if a mine has monitored flow since the initiation of mining. In this case, certain empirical relationships could be drawn and extended to closure of the mine. For example, empirical monitoring of a base-metal mine excavated into a valley wall indicated the rock in the outer 50-100 meters of each level experienced a factor of 10 increase in conductivity, which affected closure planning that initially included sealing and flooding (Northwest Geochem, 1992).

In any case, the literature contains numerous sources pertaining to, or adaptable to, changes in groundwater flow during tailings placement. Several sources addressing general concepts, theory, case studies, and modelling are summarized below.

A general example of a study which found no significant impacts from backfilling a mine with tailings was cited by Thomson and Heggen (1982). Despite underground mining activity, they wrote that the flow of groundwater in the Grants Mineral Belt would not be significantly affected by backfilling of underground stopes.

Underground disposal of wastes with a high water content (ie. deep well injection) should not take place in highly fractured or faulted formations, or in formations with nearby outcrops (Sethness and Holmes, 1979). Accordingly, a good disposal formation should:

❶ have sufficient porosity;

❷ have sufficient permeability;

❸ contain an adequate volume to handle waste being injected;

❹ be below all fresh water aquifers; and

❺ be confined vertically to prevent migration through adjacent shale, slate, slay, gypsum or marl.

While these characteristics are good for deep-well injection and important for rapid draining of water used to place tailings underground, the first two points may not be desirable if draining water is of poor quality.

When a mine closes and the underground workings are abandoned, the underground in most cases will flood. Mines above the regional water table, such as those excavated into valley walls, may not flood.

Flooding is due to the decay of the water-table cone of depression as active mining and draining cease (Thomson and Heggen, 1982). The regional groundwater system which adjusted to mining activities will therefore change and approach pre-mining levels (Morin, 1994). The placement of tailings underground can affect both the flow and chemistry of groundwater during the change in water levels. Groundwater could begin flowing through and saturate initially dewatered tailings which have been placed underground (Thomson and Heggen, 1982). The physical and chemical characteristics of the tailings could then affect the quality of the natural groundwater. This affected water can then pass into the host rock and its fracture systems. This evolution of water movement following mining is discussed in more detail in following subsections.

In discussions concerning disposal of high-level waste underground, McPherson (1984) considered a change in groundwater due to the heat given off by decaying radioactive wastes. As radioactive waste material decays or sulfide minerals oxidize, heat is generated and the heat is then transferred to the surrounding rock matrix. If the heat is sufficient, regional thermally induced uplift and fracturing may occur, as well as expansion of the rock and closing of fractures. These events could significantly change groundwater flow. However, the required amount of heat probably could not be generated by most uranium tailings. Nevertheless, even mild heat generation within the underground can reduce groundwater density and generate density-driven movement. For example, groundwater travelling horizontally may begin moving vertically when it reaches the heated disposal area. "Such buoyancy effects may result in slow moving but large-scale convection currents of groundwater" (McPherson, 1984).

Evans (1970) discussed the effect of increased fluid pressure on underground systems from deep well injections. For example, earthquakes in the Denver, Colorado, area in 1962 corresponded to injection of liquid wastes into a 12,040 ft deep disposal well. Evans (1970) also raised the issue of groundwater movement in the future due to mining activities or the removal of petroleum, gas, fresh water or mineral-rich brines. As a result, groundwater could possibly be drawn away from the underground disposal area at an increased rate or altered direction causing unexpected contamination.

Changes in groundwater flow can sometimes be delineated with tracers or through age dating of waters. To determine the age of groundwater entering a given area of an underground mine ^{14}C was used at the Asse Salt Mine in Germany (Batsche et al., 1979). Such dating of groundwater can provide information on changes in the hydraulic connection between surface and ground waters.

The following subsections discuss some of the issues related to groundwater flow in and around underground mines. Case studies illustrate the complexity and current primitive understanding of groundwater movement around mines.

4.1.3.1 Stages of mine flooding

Potential changes in groundwater flow in and around an underground mine during tailings placement depend on the state of the mine, namely, (1) dewatered during placement, (2) filling or filled with water after placement, and (3) filled prior to placement. The implications of these three states will be briefly summarized before presenting case studies from the literature.

In a mine that is actively drained, water enters through fractures in the walls and through interconnected pores in the rock and flows through the workings. Placement of tailings would thus slow water movement, eventually resulting in (1) ponding of water behind the tailings if there is open space, (2) increasing hydraulic heads in the groundwater system behind the walls, and (3) increased saturation of the tailings mass if placed by a dry method (Section 4.2.2). As a result, the patterns and rates of water movement would be expected to change. Flow would predominantly bypass the tailings mass and enter the open, drained workings. However, some water movement would still occur through the tailings, creating a potential for geochemical effects on the tailings solids (Section 4.1.1), water-chemistry impacts within the mine (Section 4.1.2), and air-quality effects due to pore-gas displacement (Section 4.1.4). Furthermore, this water movement and changes in the degree of saturation can affect geotechnical stability of the tailings mass (Section 4.2.1.5) and requirements for ventilation in the mine (Section 4.2.1.7).

In a mine that is filling with water as tailings are placed, the patterns and rates of water movement are already transient and changing with time. In general, water moving into the mine from the walls will, like a drained mine, result in (1) ponding of water behind the tailings if there is open space, (2) increasing hydraulic heads in the groundwater system behind the walls, and (3) increased saturation of the tailings mass if placed by a dry method (Section 4.2.2).

A filling mine will eventually create sufficient hydraulic heads so that an underground level no longer acts as a sink for water and groundwater simply passes through the workings. At that point, because that area of the mine is no longer operative, the major concern would be water-chemistry effects in the surrounding groundwater system (Section 4.1.2). In many cases, the geochemical reactivity of the tailings mass would be expected to decrease through time (Section 4.1.1) and the low hydraulic conductivity of the tailings relative to the open, flooded workings may cause most groundwater flowpaths to divert around rather than through the tailings. However, other contributors of chemistry such as the wall fractures may become important (Morin and Hutt, 1995; Morin, 1994).

Placement of tailings in a filled mine poses few technical concerns that can be addressed (Section 4.2.1) since the mine is already abandoned and inaccessible. The most important technical issue would probably be maximization of the volume to be placed (Section 4.2.1.2) because of the low slope angles formed by submerged tailings and the difficulty of accessing all portions of the workings. However, once the tailings are placed and the groundwater system has stabilized, the effects on patterns and rates of groundwater movement would probably resemble those in the previous scenarios. Specifically, the tailings will have a lower hydraulic conductivity than the previously open workings and thus groundwater will be preferentially focussed around the tailings. Nevertheless, there may still be sufficient flow through the tailings pores to affect groundwater chemistry (Section 4.1.2). Chemistry could be further affected negatively by rapid removal of any metals and non-metals that accumulated on the tailings solids prior to placement as well as positively by any changes in the reactivity of the tailings after submergence (Section 4.1.1).

4.1.3.2 Fractures

Because most Canadian uranium mines are located in the types of "hard rock" discussed above, fractures within the rock mass provide the primary pathways for groundwater movement and contaminant migration. Therefore, the assessment and simulation of fracture networks during tailings placement and mine flooding are critical in estimating changes in groundwater flow. However, Williams (1978) noted that groundwater flow within fractured rock, such as that associated with mining activities, is an extremely difficult medium to thoroughly and reliably investigate and assess. Disposal of contaminated material underground may be very difficult to monitor because groundwater flow and retardation of contaminants can be difficult to determine. For example, if significant subsidence occurs, as with a sudden collapse, new fractures may open and supply pathways for surface and ground waters, possibly forming hydraulic connections between aquifers and/or surface waters (Thomson et al., 1986).

Even before tailings placement and mine flooding, the permeability of adjacent rock masses are changed during active mining. Blasting, ore removal, and stress redistribution can cause fracturing throughout the rock unit and groundwater flow path, direction and volume. Wei and Hudson (1990) described the modelling of the stress induced permeability and subsequent water flow into underground workings, while taking into consideration both the permeability of the rock mass and joints within the rock mass. The depth at which the excavation takes place and the quality of the rock matrix can influence the type and extent of the permeability change in the underground. Wei and Hudson (1990) noted that in some cases deep excavation in good quality rock can cause reduced permeability, especially if the rock exhibits elastic or plastic characteristics. When an underground opening is backfilled there will again be a change in the permeability and flow paths of the groundwater towards, into, or away from the underground mining area. This change in flow through the underground will be one factor in determining if and how contaminants will migrate from the underground disposal site. A descriptive account of excavation impact on granitic rock at AECL's Underground Research Laboratory was presented by Everitt et al. (1990) and is discussed in Section 4.1.3.4.

Aydan and Kawamoto (1990) discussed the effect of discontinuities (basically fractures and faults) within a rock mass on the mechanical and structural stability of the rock. The impact of mining operations will be different for a rock mass with a large number, or variety of, discontinuities in its matrix. Similarly, Raven and Gale (1986) noted that the degree of groundwater seepage into a mine is dependent on the type, character and depth of discontinuities within a rock mass. Raven and Gale (1986) also noted, during investigations into mined openings from various underground mining operations in Canada, that groundwater seepage seemed to decrease with depth. They postulate that this may be caused by:

➊ a decrease in permeability of the structural break;
➋ a decrease in rock mass porosity and permeability;
➌ interception of recharge waters by upper levels of the mine;
➍ effect of stress concentration around mined openings; and,
➎ increased fracture permeability at mined openings.

However, as a mine floods, changes of hydraulic pressures and gradients within the mine and groundwater system may cause lubrication of fault zones under stresses, a change in fracture size and number, and a change of groundwater flow through the underground system.

The degree to which a rock mass is fractured will influence the permeability of the rock mass, its strength, and the rate and ease of groundwater flow in the rock. Atomic Energy of Canada Ltd. engaged in a program of fracture testing in granitic rocks ranging in scales from 45 mm to 2 km (Lang, 1986). The program was set up to evaluate coupled mechanical, thermal, hydraulic, and geochemical properties within a natural fracture system.

A detailed hydrogeologic study of shallow fractured gneiss (200 x 150 x 50 m deep) including visual borehole logging of fractures and pump tests is reported by Raven (1986). Bulk hydraulic conductivity and calculated effective aperture were lognormally distributed with geometric means of $2x10^{-9}$ m/s and 11.8 μm, respectively. The scale below which the rock could be considered a relatively homogeneous and isotropic porous medium was approximately 10 m. Anisotropy ratios of conductivity were typically between 1 and 10.

Wilkins et al. (1984) investigated the slow micro-fracture growth within plutonic rock. Fractures opened and closed due to changes in the geologic environment of an area. In the case of an underground mine used for the disposal of uranium tailings, the growth of microfractures could cause an increase in the rock matrix permeability and an increased flow of groundwater which could in turn produce an increased rate of contaminant migration from the disposal site. Under the stress of mining operations, fracturing of the rock body can increase due to the changes in stress. As stresses continue to change within the rock mass due to continued mining, flooding of the mine, or backfilling of stopes, fractures may continue to grow at a slower rate. The rate and/or probability of fracture propagation can be determine from strength/stress ratios for various rock types and stress situations (Wilkins et al., 1984). One of the stresses modelled by Heinrich and Walker (1987) was the expansion of water causing microfracturing in rock. This process is relevant in underground disposal of nuclear waste due to the heat generated during radioactive decay. The heat generated by uranium tailings through sulfide oxidation or radioactive decay may not reach those of nuclear wastes, but the concept of heat effects leading to some microfracturing in rock adjacent to backfilled tailings may still be relevant.

Because groundwater flow in and around an underground mine is often determined by fractures, it is worthwhile at this point to summarize some detailed fracture research arising from potential high-level-waste repositories. This research reflects the current state-of-the-art in monitoring and characterizing fractures. Similar studies for underground disposal of tailings would likely be less intensive and would probably be less informative due to the impressive amount of time and effort needed to conduct these studies.

Fractures exposed in underground mine walls can be grouped into two basic types at endpoints in a spectrum: long, continuous, pre-existing fractures and short, mining-induced fractures. The first type of "large-scale fractures" is the result of long-term geological processes whereas the second type is caused by mining activity and is often labelled "excavation response" or "excavation damage zone" in waste-repository studies. The excavation-response fractures are derived from immediate fracturing of rock upon detonation of explosives as well as creation and expansion of fracture surfaces through time due to stress relief (Everitt et al., 1989). Excavation-response fractures can sometimes be distinguished by their lack of mineral infilling or alteration and by their symmetry to blast or room geometry.

The characteristics and patterns of large-scale fractures reflect numerous geological processes that have operated over some period of time, including the present, since formation of the rock. Because of the variety of processes and their effects on fracturing, fracture patterns on small and large scales often appear almost random in nature (e.g., Figure 4-4).

Some of the most recent, detailed studies of fracturing from excavation response come from studies of nuclear-waste repositories. These studies are often divided on the basis of three basic rock types: clay, salt, and crystalline. The general view of "clay" repositories appears to be one of great difficulty because of the "very wide range of numerous characteristics" and the difficulty in safe, careful mining of the clay (Heremans, 1989). Results of one repository are not thought to be easily extrapolated to other sites, although the self-healing of fractures in clay is typical. Nevertheless, salt and "granite" are thought to be more consistent worldwide in their physical characteristics (Heremans, 1989).

Researchers in salt repositories do not seem as confident as Heremans (1989) believes (Matalucci, 1989). However, the researchers appear more comfortable and successful with their understanding and modelling of processes which affect fractures, but more work is required. Fracturing in crystalline rock is discussed in the following three sections.

The flow of groundwater in, and conductivity of, fractures is regulated by the third power of the width of the aperture (summarized in Brookins, 1983):

$$Q = 2/3 \{[(P_i\text{-}P_o)d_a{}^3 d_w]/[vd_l]\} \tag{4-2}$$

where Q = flow (m^3/s)

P_i = pressure at inlet of fracture (N/m^2)

P_o = pressure at outlet of fracture (N/m^2)

d_a = half-width of fracture aperture (m)

d_w = length of fracture, perpendicular to flow (m)

d_l = length of fracture parallel to flow from inlet to outlet (m)

v = viscosity of fluid ($N{\cdot}s/m^2 = kg/(m{\cdot}s)$)

The average velocity of groundwater flow is two-thirds the centerline velocity where velocity across the aperture is given by (corrected from Brookins, 1983, but still not dimensionally correct):

$$V_x = \{(P_i\text{-}P_o)/(2vd_l) \cdot [1\text{-}(d_x{}^2/d_a{}^2)]\} \tag{4-3}$$

where d_x = distance from the centerline of the aperture (m)

V_x = velocity at distance d_x from the centerline of the aperture (m/s)

4.1.3.3 Studies at the Stripa Mine in Sweden

The Stripa Mine located in central Sweden has been used as a research facility in determining the responses of a granitic body to excavation operations. Research into geochemical, hydrogeologic, and geomechanic responses as well as the development of monitoring and testing equipment were all goals of the project. This project was a cooperative international effort among a several participants (Carlsson, 1981).

Nordstrom et al. (1989), while reporting on the hydrogeology of the Stripa Mine in Sweden, stated that the total porosity within a fractured medium can be calculated from:

$$\theta_T = \theta_K + \theta_D + \theta_R \tag{4-4}$$

where θ_T = the total porosity in a fractured medium;

θ_K = the effective flow porosity or kinematic porosity which represents dominant fluid flow through the fractures;

θ_D = the diffusion porosity which represents limited flow through the fractures; and,

θ_R = the residual porosity which also represents limited fluid flow through the fractures.

Kinematic porosity regulates groundwater flow through main fractures; diffusion porosity regulates groundwater flow through smaller pores connected to the kinematic fracture openings; and residual porosity is that associated with closed pores (Figure 4-3; Nordstrom et al., 1989).

The Stripa project in Sweden involved detailed studies of fracture hydrology in and around Precambrian quartz monzonite at depths of hundreds of meters (Witherspoon et al., 1981). Fracture mapping, even on a small scale, was found to be complex (Figure 4-4) and the conclusion reached was "[a]t present, it is impractical to model such ubiquitous joints as they actually exist; techniques are being developed to represent them stochastically...".

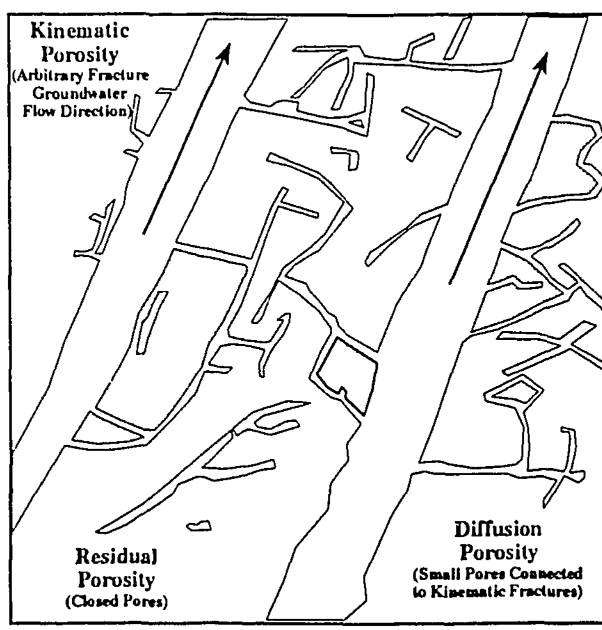

FIGURE 4-3. Schematic Example of Kinematic, Diffusion, and Residual Porosities (Nordstrom et.al., 1989).

To further examine large-scale hydrogeology of the rock at Stripa, 30-meter-long boreholes producing minor flow into one tunnel were sealed with multiple packers and the resulting increases in piezometric pressure were monitored over discrete intervals (Figure 4-5). The relatively low pressure increases were attributed to years of loss of pressure and water to adjacent free-draining workings. When the borehole with the dominant flow (R01) was packered later, additional increases in pressure after eight days were noted in some intervals along all boreholes around the tunnel (Figure 4-5). The additional increases likely reflected the degree of interconnectedness of discrete fracture zones distributed three-dimensionally around the tunnel.

After all flowing boreholes were packered during this test at Stripa, air circulation was adjusted to evaporate all inflowing groundwater. Based on air flow and temperature, total inflow was 50 mL a minute and, based on measured hydraulic gradients, average hydraulic conductivity of the rock was calculated to be 10^{-11} m/s (Witherspoon et al., 1981).

4.1.3.4 Studies at AECL's Underground Research Laboratory

High-level-waste repositories in crystalline or "hard" rock are of great interest in Canada. This can be seen in the extensive physical, chemical, thermal, and mechanical studies performed in granite at the Canada Underground Research Laboratory (URL) at

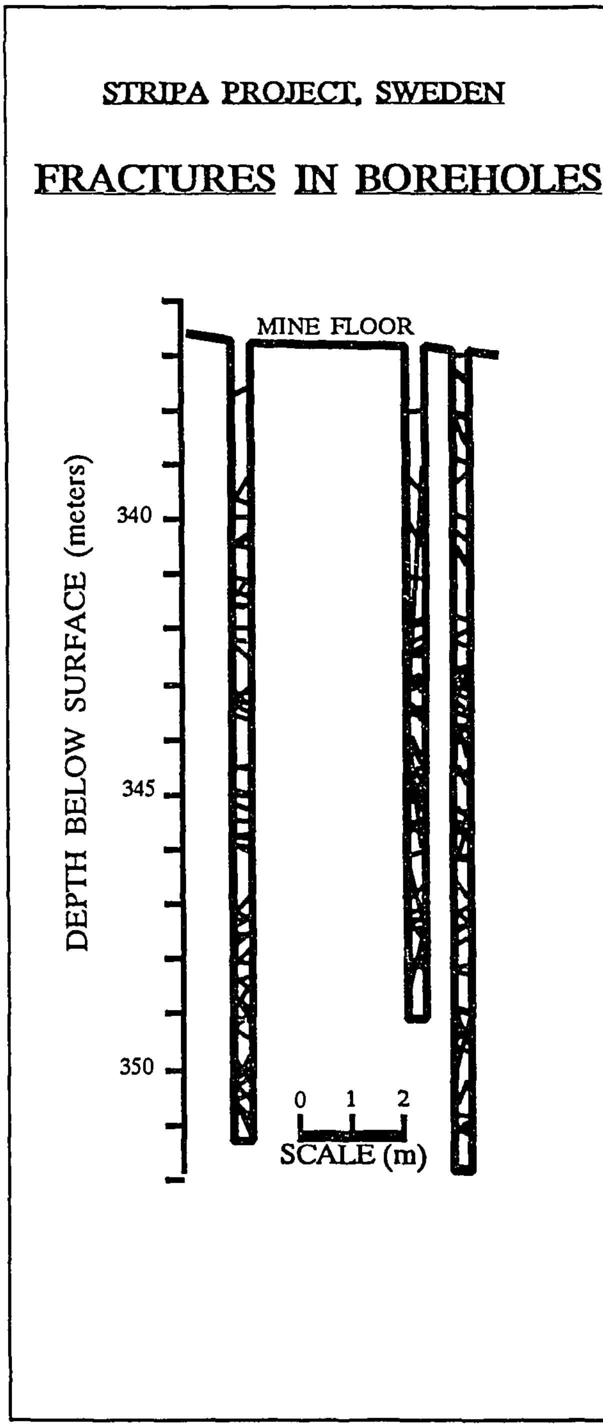

FIGURE 4-4. Fracture Patterns in Closely Spaced Boreholes at the Stripa Project (from Witherspoon et al., 1981).

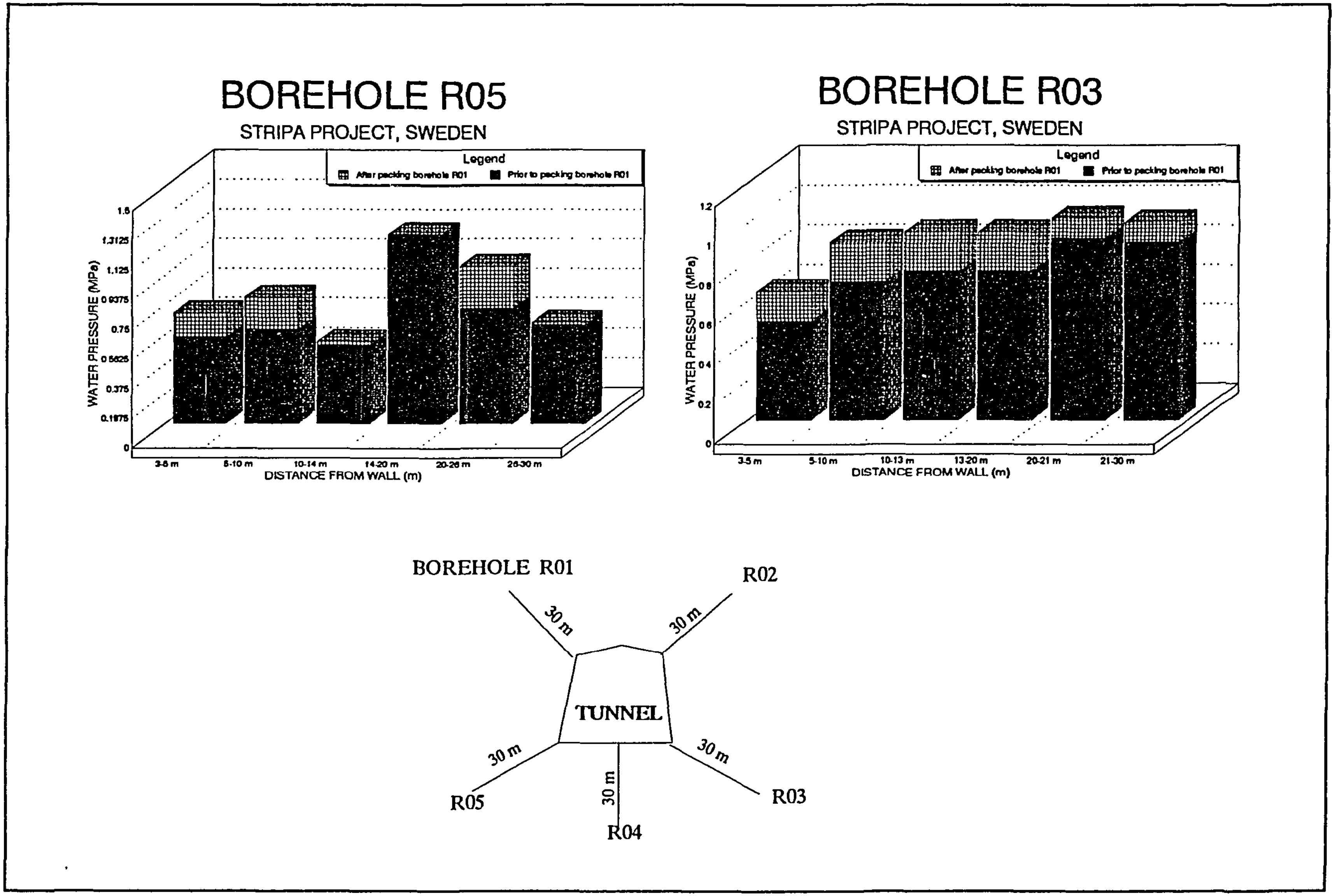

FIGURE 4-5. Changes in Water Pressure within Selected Fractures at the Stripa Project, Sweden (adapted from Witherspoon et al., 1981).

AECL's facility at Pinawa, Manitoba. The URL studies appear to be particularly focussed towards physical aspects of fractures and are well received by international researchers (Barbreau, 1989). In general, excavation-response fractures were found to extend to about one meter from the mine wall and were not long-term concerns due to the existing ability to seal them.

In a carefully planned, executed, and monitored study, AECL mined through a distinct fracture zone in the URL at a depth of 237 meters in granite (Lang, 1989). Through prior drilling, the "Room 209 Fracture Zone" was known to have a maximum thickness of 0.4 meters, consisting of one to six en echelon fractures. It intersected the proposed tunnel nearly vertically, perpendicular to the length of the tunnel, and was associated with a low-permeability "band" which represented about one-half of the zone to be excavated by the tunnel. The zone extended at least 30 meters laterally beyond the sides of the intended tunnel, only a few meters below the floor, and roughly 30 meters above the floor where it was hydraulically connected to a shear zone. The remainder of the rock to be exposed was "essentially unfractured" with an unconfined compressive strength of 182 $\pm$ 10 (standard deviation) MPa, a tensile strength (Brazilian) of 9.1 $\pm$ 0.4 MPa, Young's modulus of 69.1 $\pm$ 1.7 GPa, a Poisson's ratio of 0.24 $\pm$ 0.02, and a coefficient of linear thermal expansion at 25°C of (2.5 $\pm$ 0.7) x 10^{-6} $(°C)^{-1}$. Normal stiffness of the zone was measured at 500 MPa/mm.

Excavation of the tunnel towards the fracture zone, from a distance of approximately 12 meters to 4 meters (not reaching the zone), caused the equivalent single-fracture aperture to decrease from about 59 to 53 μm within a few days. Subsequent mining through the fracture zone occurred as a "pilot tunnel" of 2.5 meters width, followed about three weeks later by "slashing" which widened the tunnel to 3.85 meters. Modelling showed that stress changes and displacements in intact rock could be reasonably predicted, but the hydrogeologic response of the fracture zone could not. When the pilot tunnel passed from 2.5 m behind the zone to 2.5 m past the zone in one day, measured flow rose from 0 to 300 mL/min relative to the predicted flow of 2077 mL/min. At the same time, equivalent single-fracture apertures within 2 meters of the walls lost about 10 μm of aperture where total aperture was 60 to 100 μm and about 5 μm from the portion with a total of 20 μm. Because conductivity is typically considered a function of the third power of aperture (Equation 4-2), this loss represented a significant change in conductivity. However, most of the lost aperture was recovered within a few days. Also, a monitoring point at 12.8 meters inside the wall showed no loss of aperture. Piezometric heads were predicted to decrease 20-50 meters during pilot excavation, but head losses no greater than 5 m were measured (Figure 4-6).

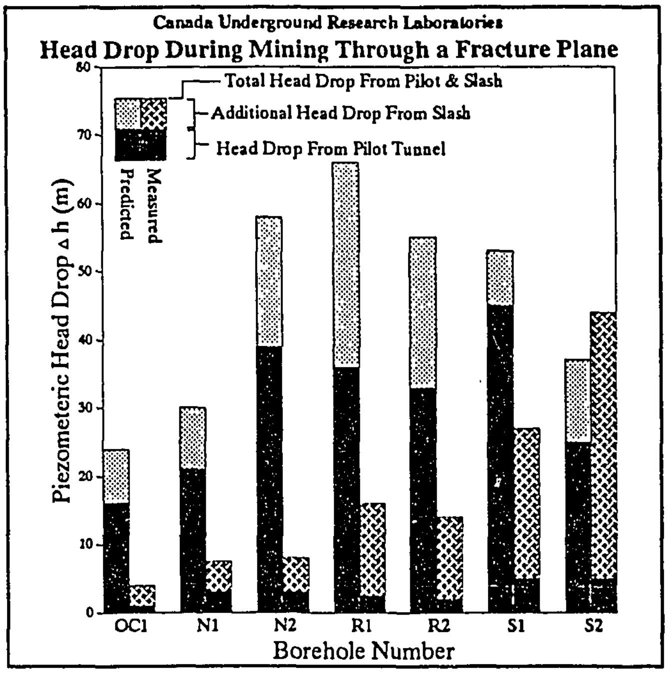

FIGURE 4-6. Predicted and Measured Head Losses During Mining Through A Fracture Plane at URL.

The later slashing of the tunnel raised the rate of inflow to 450 mL/min relative to a prediction of 3565 mL/min. Like the pilot tunnel, the portion of the zone with 20 μm aperture lost roughly 5 μm and recovered the loss within a few days. However, the 60 and 100 μm portions lost 20-30 μm and did not recover. Loss of piezometric head was expected to be less than during pilot excavation, around 7-30 meters (Figure 4-6). Although measured head losses were about one-third of predictions, except at two stations, the absolute values of head loss were greater than those of the pilot excavation.

Visual inspection and monitoring after excavation indicated the walls and roof showed no visible fracturing beyond 0.2 meters. However, blasting of the floor involved less controlled methods and thus blasting-induced fracturing extended at least to one meter below the floor with the shallowest 0.3 meters having a high conductivity. In general, Lang (1989) concluded that "the models used for predicting the hydrogeologic response did poorly" and improvements are underway.

A similar project of carefully mining through a section of welded tuff was conducted by Sandia National Laboratories (Zimmerman et al., 1988). Careful monitoring provided critical information for predictive models.

In another URL study, the depth of excavation-response fracturing around a carefully blasted, circular shaft extension was limited to approximately 0.3 meters (Everritt et al., 1989). This fracture depth was less than the measured 1.5 meters in the rectangular access shaft and an average of 0.5 meters (maximum of 10 meters) in access tunnels. Jakubick et al. (1989) indicated that the maximum induced depth of fracturing was 10 meters, but such fracturing is not apparent at shallow depths (15 meters) where lithostatic pressure is minimal. For the excavation-response fractures in the circular shaft, Everritt et al. (1989) created two categories. "Microfractures" had with exposed trace lengths on the wall of less than 0.05 meters, were parallel to the maximum principal compressive strength in a horizontal plane, and were believed to be extensional in origin. "Mesoscopic" fractures had exposed trace lengths of 0.29-1.5 meters, were perpendicular to the microfractures, and were of uncertain origin. The mesoscopic fractures consistently had apertures less than 0.5 mm, were described as "undulating and rough", and had roughness ratings of 4 to 5.

The permeability of a discrete fracture can sometimes be determined with a vacuum-based technique (Jakubick et al., 1989). Because the technique can disturb the fracture plane up to a distance of a few meters, the full bulk conductivity of most excavation-response fractures can be determined. In a summary of fracturing and vacuum-based permeabilities at three sites, Jakubick et al. (1989) reported for the first site that excavation-response fracturing in a 60-meter-deep tunnel in horizontally bedded limestones with shaley interbeds was 0.9 m in the sidewall and 0.4 m in the roof. At a depth of only 15 meters at the URL access shaft (second site), no excavation-response fracturing was found, apparently due to the relative lack of lithostatic stress at this shallow depth. However, pre-existing discrete fractures were identified and tested (Figure 4-7). At a depth of 100 meters at the third site, a tunnel in Precambrian medium-to-coarse-grained granitic gneiss with lenses of pegmatite, biotite schist, and quartz showed excavation-response fracturing extended to depths of 0.5 to 1.15 meters with other, permeable zones further behind the walls (Figure 4-8). Based on this work, Jakubick et al. concluded that the excavation-response zones should not be conceptualized as homogeneous anisotropic porous media with monotonically decreasing permeability with distance from the wall. However, sealing of the zone and deeper fractures is possible with clay-based and/or cement-based compounds.

4.1.3.5 Studies at Elliot Lake, Ontario

The issue of altered groundwater flows upon mine closure has been examined in Elliot Lake. Golder Associates Ltd. (1991b) conducted a study on behalf of Rio Algom Limited regarding the impacts of flooding on Quirke and Panel mine's hydrogeology. Panel Mine workings are located primarily under Quirke Lake, while the Quirke Mine workings are situated such that they could affect both Quirke Lake and the Serpent River. Golder Associates (1991b) assessed the impacts of flooding these two mines on the operating conditions within nearby mines, the regional groundwater flow system, and the potential for acidic mine water to enter surface waters. Golder Associates (1991b) developed a "Groundwater Flow Model" to simulate the groundwater conditions during the periods of pre-mining, mined out but not flooded, and mined out and flooded.

Golder Associates Ltd. (1991b) determined that for the Quirke and Panel mining areas, groundwater circulation occurs mainly within the uppermost portion of the bedrock and overburden, with infiltration occurring at higher elevations discharging to major surface-water

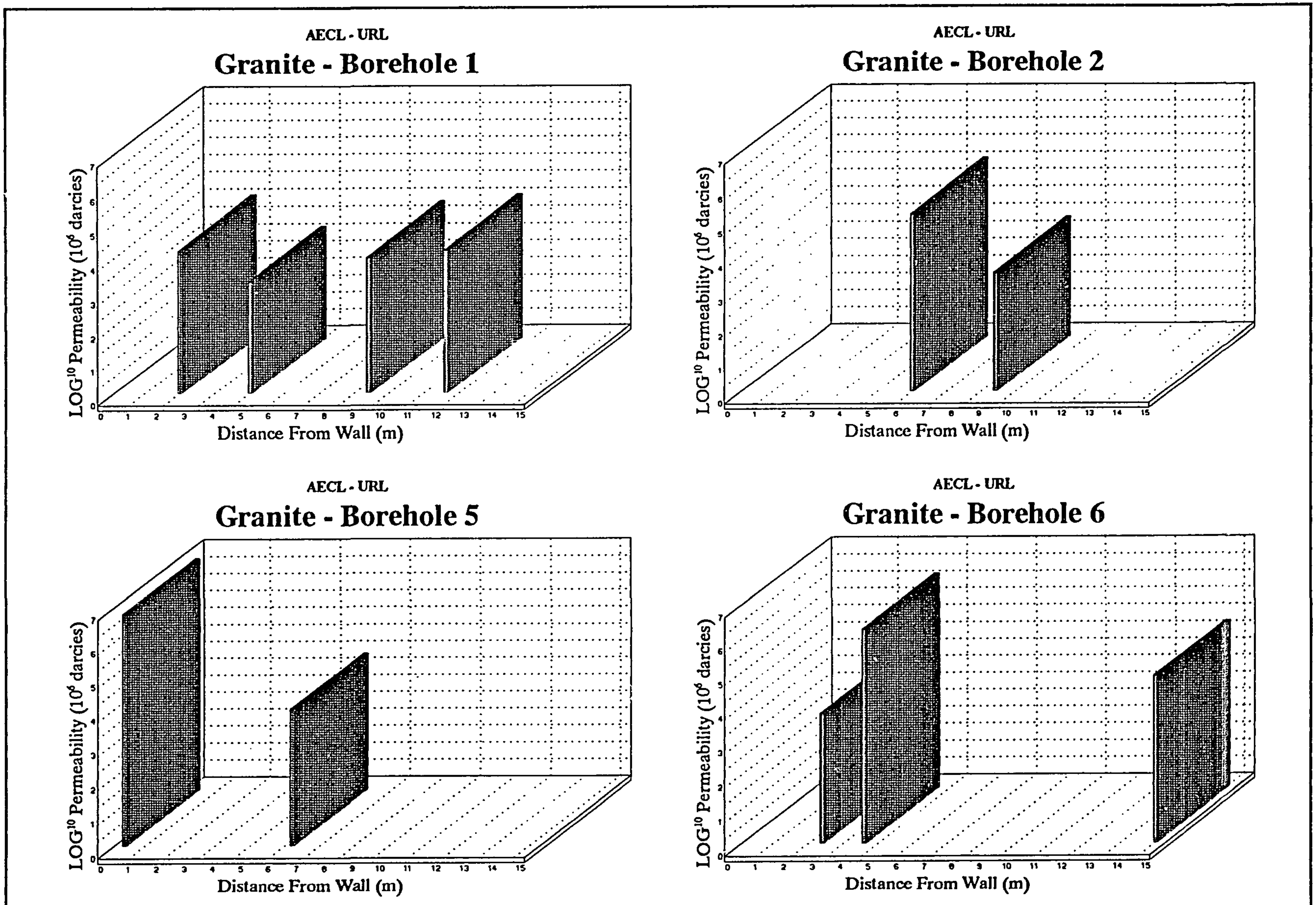

FIGURE 4-7. Discrete Fracture Permeabilities at URL (15 m depth at shaft collar).

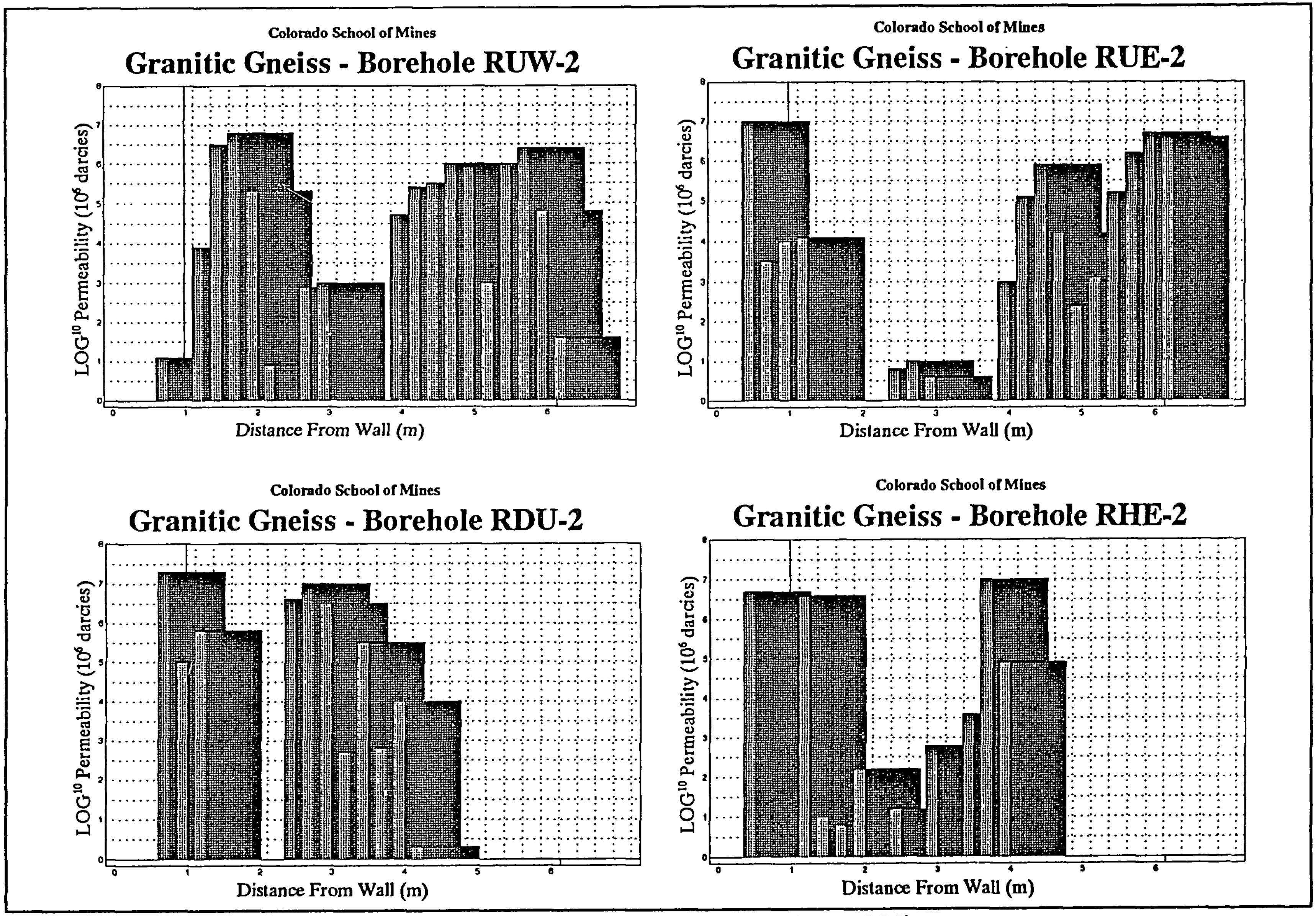

FIGURE 4-8. Fracture Permeabilities at Colorado School of Mines' Experimental Mine.

courses within local valleys. Hydraulic conductivities of 1×10^{-6} to 1×10^{-7} m/s were reported for the upper 200 ft (60 m) of fault fractured bedrock. The hydraulic conductivities were found to decrease with depth to less than 1×10^{-8} m/s, with the fractured rock exhibiting hydraulic conductivities one order of magnitude higher than the unfractured rock (Golder, 1991b). For the Quirke Lake and Serpent River overburdens, which vary from a well graded silty sand till to gravelly sand, Golder Associates (1991b) reported hydraulic conductivities of 1×10^{-6} m/s for tills, 1×10^{-7} m/s for more sandy tills, less than 1×10^{-8} m/s for more silty tills, and 1×10^{-6} m/s for silty sand to 1×10^{-3} m/s for sands and gravels.

From their investigations, Golder Associates (1991b) calculated an infiltration rate into unfaulted, fractured, low-hydraulic-conductivity rock of 4×10^{-3} inches/yr (0.1 mm/y) for the Serpent River area near Rio Algom's Quirke and Panel Mines in Elliot Lake, Ontario. This was based on an assumed hydraulic conductivity of 1×10^{-10} m/s and a hydraulic gradient of 0.025 towards lower levels of the Quirke Mine.

In their final conclusions on the flooding impacts of Quirke and Panel Mines, Golder Associates (1991b) wrote that the Panel Mine would not flood to a high enough level so that a "significant groundwater flux to the [Quirke] lake would occur". Water will "flow through the [Quirke II] mine workings and ultimately discharge to the Serpent River after flooding". "Since the water will be passing through the ore zone, some mineralization of this water can be initially expected. This may result, in short term, in water chemistry which is different from the current conditions in the Serpent River". Regional effects of mining on the Serpent River area resulted in the groundwater flow system being affected for at least a 6 mile radius, causing the Serpent River to become a groundwater recharge zone, as opposed to the discharge zone existing before mining commenced (Golder Associates, 1991b). "Closure and flooding of the [Quirke and Panel] mines is expected to result in the re-establishment of a regional flow system that approaches that of the pre-mining case. Minor variations in the immediate vicinity of the mines will remain though, even after flooding" (Golder Associates, 1991b).

Hydrogeologic investigations into the effect of surface tailings impoundment on the groundwater system on the Crotch Lake Basin near Elliot Lake was described by Boyd et al. (1982). The investigations included examination of fracture zones and the hydraulic conductivities of the rock around the Stanleigh Mine proposed tailings impoundment. The hydraulic conductivities determined from hydrogeologic modelling of the area, based on field data collected during the investigation, are reproduced in Table 4-9.

Table 4-9
Hydraulic Conductivity Ranges from the Stanleigh Mine Area, Elliot Lake
(from Boyd et.al., 1982)

Rock Type	Hydraulic Conductivities (m/s)
Unstructured Metasedimentary Rock	
Near surface zone, at depths < 12 m	$1x10^{-5}$ to $5x10^{-7}$
Deep zone, depths 12 m - 90 m	$2x10^{-6}$ to $8x10^{-8}$
Diabase Sills	
K_{max} (E-W)	$5x10^{-6}$ to $3x10^{-8}$
K_{min} (N-S)	$5x10^{-7}$ to $3x10^{-9}$
Major Fault Zones	
K_{max} (parallel to fault)	$1x10^{-5}$ to $8x10^{-7}$
K_{min} (perpendicular to fault)	$8x10^{-6}$ to $5x10^{-7}$
Minor Fault Zones & Diabase Dykes	
K_{max} (parallel to fault)	$1x10^{-5}$ to $8x10^{-8}$
K_{min} (perpendicular to fault)	$8x10^{-6}$ to $5x10^{-8}$

4.1.3.6 Modelling

In light of published work such as presented above, various models have been developed to address groundwater movement in and around underground mines, often combined with other features such contaminant transport. Most of these models were discussed in Section 4.1.2.7 due to the often integrated nature of models simulating flow and chemistry. Only a few models focussing on water movement are discussed here.

Prior to initiating modelling projects AECL organized a review of available data on pluton hydrogeology to determine the parameters of significance and the relevant equations associated with the parameters for groundwater modelling (Burgess, 1979). Atomic Energy of

Canada Ltd. has organized several projects which involved the modelling of local and regional groundwater flows prior to (Guvanasen et al., 1985; Guvanasen, 1984; Weitzman, 1985; Intera, 1985), during (Intera, 1985; Weitzman, 1985), and after (Weitzman, 1985) the excavation of underground mined areas. New data which could now be included are the borehole experiments conducted to determine existing conditions prior, during and after excavation of underground workings (Bower et al., 1986; Lang, 1986).

Pickens and Grisak (1979) took a regional approach to model groundwater flow in a fractured system. They used a continuum approach for simulating liquid flow, heat transport, and solute transport in a groundwater flow system using a finite-element method. Mathers and Hembroff (1986) used a Hele-Shaw method of predicting groundwater flow through an idealized fracture network. The dynamic pressures calculated by their model showed reasonable agreement with measured values obtained through laboratory experimentation.

Despite all of the foregoing research, the impact of short-duration catastrophic events on groundwater flow have often been largely ignored by modellers. However, Marine (1980) summarized the categories and simulations of potential effects of earthquakes on flow in and around underground mines. More details, through a literature review and analysis, are provided by Pratt et al. (1978).

4.1.4 Degradation of air quality in mines

When radioactive materials such as uranium tailings are placed underground, radioactive gases may migrate through underground fractures and any surrounding unsaturated zone (Striegl, 1990) into the atmosphere within and outside of the mine. Non-radioactive constituents in tailings can also affect air quality, such as sulfur-bearing gases from pyrite oxidation. Eight percent of respondents to the questionnaire for this study (Appendix B) mentioned this issue.

Related topics and case studies have been presented in Section 3.1.6, so only topics particularly relevant to placed tailings in underground mines are discussed here.

If the mine and placed tailings are in full states of saturation (Section 4.1.3.1), there is no air space and thus there is no concern over air quality in the mine. However, there can be problems with migration of gaseous contaminants along aqueous pathways until they volatilize into air space elsewhere. This issue was examined in Section 4.1.2.

Many authors, including Kilborn and Beak (1979) and Khan and Raghavayya (1989), feel that the inhalation of radon gas and radon daughters is probably the most significant airborne pathway to human exposure from uranium tailings, although many others also state that the water pathway is found to be more important in most case studies (e.g., Hladka et al., 1991). Radon gas arises from the radioactive decay of radionuclides within the uranium tailings and diffuses from its point of origin into the interstitial spaces of the tailings. The radon moves through the tailings by diffusion eventually reaching the tailings/atmosphere interface (Kilborn and Beak, 1979). Kilborn and Beak (1979) explained that the rate of radon diffusion is a function of partial pressures and the properties of the tailings which include mineralogy, radium concentration, porosity, moisture content, air content, particle size. A list of some established radon diffusion coefficients (K), can be found in Table 4-10.

Table 4-10
Established Diffusion Coefficients, K
(from Kilborn and Beak, 1979)

Soil Type	Soil Condition	Depth	Bulk Diffusion Coefficient, K (cm^2/sec)
Unconsolidated Glacial Debris	Moist - Matted Grass 40% Porosity	0 - 28 cm	0.02
Consolidated Sandstone	Mine Tunnel 25% Porosity	0 - 1.6 m	0.03
Alluvium (Yucca Flat)	Dry, Sandy	1 - 3 m	0.036
Alluvium	Very Dry, Powdery Sparse Ground Cover	0 - 30 cm	0.10

After placement of uranium tailings underground, but prior to flooding of the mine, Striegl (1990) indicated that the "movement of the gases away from the waste may be dominated either by diffusion, or by coupled advection plus diffusion, depending on the rate and volume of gas production and on local boundary conditions". He also noted that "where partial pressures of gases produced are small relative to total barometric pressure, it generally is assumed that total gas pressures are equal at any given altitude and that transport of trace quantities of waste-produced gases is controlled by ordinary diffusion along partial-pressure gradients.... Molecular diffusion of the trace gases through the surrounding unsaturated zone is affected by pore-size distributions, air- and water-filled pores, and by chemical and physical interactions that occur between gases, water, and solids."

On the other hand, placement of tailings could improve air quality. Franklin et al. (1982a) reported a brief period of higher radon levels in an underground mine during coarse-tailings placement, followed by a long-term reduction of 52-84% in radon levels. Poorer water quality, however, was noted around the backfilled tailings. In a related study, Franklin et al. (1982b) reported an emanation rate of 0.55 pCi/cm^2/s from unsorted, cemented, backfilled tailings in two dewatered stopes in an Elliot Lake mine. This rate was approximately 100 times greater than the average flux in the mine. Kauffman et al. (1987) noticed a small decrease in radon release upon backfilling. Clausen and Archibald (1983) noted that radon production decreased proportional to the amount of backfill. These variations likely reflect the site-specific nature of radon-emanation rates, as discussed further below.

Radon diffusion from tailings can be calculated through the use of equations which employ the physical/chemical characteristics of uranium tailings. This subject was discussed by Kilborn and Beak (1979) in a report for the Atomic Energy Control Board. By using existing equations, the effects of weather conditions and complete coverage of the tailings by water was calculated and were usually found to significantly alter the radon diffusion rate from tailings. Some relationships between radon emanation and climatic conditions affecting the tailings include

(Kilborn and Beak, 1979):

❶ precipitation and snow cover can reduce the porosity of the tailings reducing the flux rate;

❷ changes in temperature can increase convection activity causing pore gas to migrate;

❸ wind action can cause migration of radon gas from 0-1.8 m depth;

❹ a 15 cm frozen ground can reduce radon flux rate by 40%;

❺ changes in barometric pressure can alter the radon flux by 15% up or down.

Points 2, 3, and 5 are relevant for placement of tailings in unflooded underground mines. An analogy to Point 1 appears to be the most important in a flooded mine.

Archibald and Nantel (1984) wrote that "the rate of radon emanation is proportional to the rate at which radon is produced within the host material (a function of the ore grade of U_3O_8), to the radon concentration gradient in the host pores and to the diffusion properties of the material". The source of radon gas is calculated from the rate of ^{226}Ra decay and the fraction of radon which escapes from the source, or emanating coefficient, into and through the pore space. Archibald and Nantel (1984) reported that an average emanating coefficient for porous materials such as backfill is 0.2, and "the rate of diffusion is proportional to a bulk diffusion coefficient of radon in fluid", sample porosity, moisture content, temperature and fluid composition can all have an effect on this coefficient. Kilborn and Beak (1979) wrote that the "rate of emanation of radon gas (the primary radiological hazard) would not necessarily increase since permeability and moisture content of the fill is inversely proportional to the size distribution."

Archibald and Nantel (1984) supplied the following equation to calculate the concentration of radon in backfill or an orebody at any depth:

$$C = Co\ (1-e^{-z\sqrt{\frac{\lambda P}{D}}}) \qquad (4\text{-}5)$$

where: Co is the pore radon concentration,

z is the depth,

λ is the radon decay constant (2.1×10^{-6} d/s),

P is the porosity (void fraction), and

D is the bulk diffusion coefficient (cm^2/s).

These researchers also noted that radon migration through the pore system of the backfill is dependent on the effective diffusion coefficient, D/P, "which is function of the radon source material and pore fluid conditions".

Archibald and Nantel (1984) described as series of tests which should be performed on backfill material to determine its radioactive properties:

❶ determination of (β) parameters;

❷ analysis of ^{226}Ra content;

❸ determination of emanation rate parameters (Js) either in the laboratory or in the field; and,

❹ a determination of radon pore concentrations to specified depths (2 m) in the laboratory and field (using a hollow steel probe).

The results from emanation power testing conducted on a uranium tailings sample and a sample of backfill prepared from these tailings are shown in Table 4-11 (Archibald and Nantel, 1984). As can be seen from Table 4-11, the finer particles have a higher radioactivity.

Table 4-11
Typical Emanating Powers of Sample Canadian Classified and Unclassified Uranium Tailings
(from Archibald and Nantel, 1984)

Description	Emanating Power ($Ci/cm^3/s \times 10^{-18}$)
Unclassified Tailings	400
Backfill Materials	
D_{10} = 200 mesh	190
D_{10} = 270 mesh	225
D_{10} = 400 mesh	250
Fines	
Fraction - 200 mesh	530
Fraction - 270 mesh	720
Fraction - 400 mesh	850

Archibald and Nantel (1984) found that the effect of adding cement to uranium tailings for radon emanation reduction, as well as additional strength and stability, was noticeable. During a 2 m tube-pour test on uranium tailings from a Canadian uranium mine, Archibald and Nantel (1984) noted a very steep increase in emanation rate over the first 10 days after pouring the backfill. They found that the emanating rate of cemented backfill levelled out at approximately 10 $Ci/cm^2/s \times 10^{-18}$ after approximately 10 days, while the emanating rate for non-cemented backfill levelled out at approximately 17 $Ci/cm^2/s \times 10^{-18}$. After gathering more data from the laboratory experiments on ore, wastes, and tailings materials, Archibald and Nantel (1984) calculated the radon diffusion lengths over time, which are shown in Table 4-12.

Table 4-12
Diffusion Length Versus Time
(from Archibald and Nantel, 1984)

	Diffusion Length Lm (cm)	
Time After Pour (days)	**1.5m Tube Cemented**	**1.5m Tube Non-Cemented**
1	2	18
2	6	22
4	10	34
6	15	40
8	16	47
10	16	46
14	18	47
18	17	49
22	18	45
28	18	48

In another experiment on uranium tailings samples from a Canadian uranium mine, where backfill/cement mixtures by weight were 15/1, 10/1, and 7/1, a placed slurry with 65% solids by weight when showed a lower emanating power for the mixtures containing a higher amount of cement (Archibald and Nantel, 1984). Since the emanating power of the cemented backfilled is noticeably less than that of the non-cemented backfill material, this would seem to be an effective method of lowering the rate of radon in escape into the underground environment for tailings backfill. Archibald and Nantel (1984) also reported that results similar to those discussed above have been found on the top of an aged, in-situ test pour of identical non-cemented backfill.

The preceding types of tests and models can be used to determine the ventilation requirements in a mine which is using uranium tailings as backfill material. An example of radon production in a mine using uranium tailings as backfill material is presented in Tables 4-13(a) and 4-13(b) (Archibald and Nantel, 1984). Once the concentration of radon and its emanating power are determined, the volume of air exchange required to maintain safe working concentrations in the underground mine can be calculated, and the ventilation requirements determined.

Table 4-13(a)
Physical Parameters Of Mining Zone
(from Archibald and Nantel, 1984)

Total Mining Area	4.81×10^4 m^2
Total Pillar Area	1.55×10^4 m^2
Average Height of Zone	6.1 m
Exposed Surface Area of Floor	2.40×10^4 m^2
Exposed Surface Area of Back	3.26×10^4 m^2
Volume of Air	147×10^6 L
Exposed Surface Area of Walls	3.57×10^4 m^2
Exposed Perimeter of Pillars	8505 m
Exposed Surface Area of Pillars Walls	5.19×10^4 m^2
Exposed Backfill Surface Area	6.74×10^4 m^2

Hart et al. (1986) reported that the molecular diffusion coefficient for radon at 20°C at one atmosphere pressure is 10^{-5} m^2/s in air and 10^{-9} m^2/s in water. This difference shows how moisture can affect the emanation of ^{222}Rn through a porous medium.

Table 4-13(b)
Radon Gas Produced In Mining Zone, 25% Mined Out
(from Archibald & Nantel, 1984)

Source	Radon Produced (cCi/min. x 10^6)	% of Total
Floor and Back	24.50	4.6
Broken Ore	1.26	0.2
Old Layer Ore on Floor	19.20	3.6
Walls of Mining Zone	14.99	2.8
Pillar Walls	258.46	48.4
Backfilling Operations		
-Backfill in Place	212.30	39.8
-During Pouring	0.10	0.0
-Build-up in Transit	1.10	0.2
-Fresh Pour	2.10	0.4
TOTAL	534.01	100.0

During an investigation into the health impacts of inactive uranium mines in the U.S. conducted on behalf of the U.S. Environmental Protection Agency (EPA), a list of 1250 surface mines and 2030 underground mines was acquired from the U.S. Department of Energy (DOE) by Hans et al. (1981). Based on the information obtained from DOE, models for surface and underground mines were developed to predict the ^{222}Rn emanation rate and subsequent human exposures. The calculated annual release rate of ^{222}Rn from a model surface pit was 8.0 Ci/yr, while ^{222}Rn releases from model underground vent portals were 7.6 Ci/yr.

The inhalation-hazard indices in the Konrad iron ore mine were found to be 0.6 for ^{230}Th and 4.0 for ^{228}Ra as calculated from (Brewitz and Löschhorn, 1980):

$$HazardIndex(HI) = \frac{A}{MPC}(m^3) \tag{4-6}$$

where: A is the activity of the radionuclide and

MPC is the Maximum Permissible Concentration by inhalation or ingestion.

4.2 Technical Aspects of Placement

4.2.1 Technical Issues

4.2.1.1 Preparation and maintenance of underground workings

To carefully backfill uranium tailings into abandoned underground workings, inspection of the workings would have to be conducted first to ensure they were safe for human entry. If the workings were not deemed safe, maintenance would have to be performed along with the installation of ventilation, the equipment for tailings placement (Section 4.2.2), and other services necessary for disposal operations. The volume of tailings to be relocated into underground workings can determine the amount of maintenance or preparatory work required to make this transfer possible. For example, at Denison Mines in Elliot Lake, Golder Associates et al. (1992) estimated that placement of 0 - 5 million short tons of tailings would require no work underground, from 5 - 14 million short tons would make placement difficult with some work required underground, and finally 14 - 24 million tons would make placement extremely difficult with extensive work needed underground.

Once the underground workings are entered, many of the features and parameters discussed in Section 4.1 should be assessed and documented. At that point, any work needed to improve the stability and isolation of the workings can be undertaken.

If the workings are already flooded and inaccessible (Section 4.1.3.1), then much of the discussion in this section may be irrelevant. Perhaps drilling from the surface into and around the workings may provide some information pertinent to tailings placement and subsequent effects, although such drilling into deep mines may not be highly successful due to inaccuracies in drilling and mine plans (Aljoe and Hawkins, 1991). For this reason, underground disposal of tailings into inaccessible flooded mines is fraught with more uncertainties.

To help offset the cost of preparing and/or maintaining underground workings so that uranium tailings may be backfilled, underground heap leaching may be a possibility. Also, washing of the underground walls with water may result in water containing recoverable amounts of uranium. These practices may help to pay for underground maintenance costs, but in any case would lessen the inventory of potentially economic contaminants placed underground upon closure (see also Section 4.2.1.4).

Underground leaching of abandoned workings has been done for many years, as early as 1964. MacGregor (1966) reported on leaching of abandoned underground workings to recover U_3O_8 at the Stanrock Uranium Mine, Elliot Lake, Ontario. McCready (1986) reported on experimental underground bioleaching of uranium ore by flooding mined-out workings with water carrying nutrients and the bacteria *Thiobacillus ferrooxidans*. This experimentation took place at the Denison Mine, Elliot Lake, Ontario and started in 1984.

4.2.1.2 Maximization of tailings volume

One of the main problems, if not the primary problem, in using underground mined-out workings as a final disposal site for uranium or other tailings is available space. If milling or concentrating of ore only removes a small portion of the original weight, not all the tailings from one mine will usually fit back into that mine's underground workings because of the "swelling" factor (Section 2.4). One-third of respondents to this study's questionnaire (Appendix B) mentioned this issue.

Kilborn and Beak (1979) reported to the Atomic Energy Control Board that complete underground placement of tailings is not possible due to the bulking of the extracted ore during milling processes. Kilborn and Beak (1979) further explained that, when rock is mined, the volume of that rock increases by 60 - 70%. After milling, the volume still remains at about 67% greater than the original rock prior to mining even though (1) milling produces particles of a smaller and more uniform size and (2) valuable minerals (often comprising less than 1% of the total ore) are mostly removed (Kilborn and Beak, 1979). If some portion of surface-impounded tailings must remain in an impoundment, the value of underground disposal decreases and may only be worthwhile if one or more key concerns over surface impoundments (Sections 2.3 and 3.1) are significantly lessened.

Senes Consultants (1991b) estimated that underground placement would only remove 26 - 35% of the surface-impounded tailings. The majority of the tailings would have to remain in surface impoundments and the overall benefit to the environment would thus be limited. Table 4-14 is a comparison of available underground void space and relocation potential for uranium tailings at the Quirke and Panel Mines at Elliot Lake, Ontario.

Since the finer "slimes" fractions of mill tailings do not dewater effectively and are more difficult to handle, they can be a significant concern when considering disposal underground. If the slimes are separated from the tailings by cycloning, for example, a method of preparation

Table 4-14 Mine Tailings Backfill Data (from Senes Consultants, 1991b)	
Total available space in Quirke and Panel Mines	
conventional engineered fill	15.3×10^6 m^3
thickened slurry	20.6×10^6 m^3
Percentage of tailings potentially relocated to backfill	
conventional engineered fill	35%
slurry	26%

will be necessary prior to their final disposal. One innovative technique for dewatering and stabilizing slimes is electrokinetic densification, which uses direct-current electricity to draw water out of the tailings mass (Sprute and Kelsh, 1976; Anonymous, 1977). The process involves passing 100 - 400 V of direct current through the slimes, using wire fencing and metal pipes as electrodes. The electrical current causes suspended particles to move towards one of the electrodes and clarified water towards the other electrode. Initial large-scale tests using 18 yd^3 containers required 25-30 kilowatt-hours a yd^3, which was approximately \$0.12-0.15 a yd^3 in 1976. In five field-scale tests, up to 41 yd^3 of slimes and slime-bearing tailings were satisfactory stabilized within a few days usually with a total of 7.4 kilowatt-hours a yd^3 (\$0.04 a yd^3 in 1976). One older, large slime deposit required 35 kilowatt-hours a yd^3.

Kilborn and Beak (1979) also acknowledged that the use of solidification or electrokinetic densification methods may allow an increased volume of the slimes to be use in backfill underground. A solidification method was developed by Canadian Waste Technology Inc. to covert industrial waste sludges and slimes into solids, while two American base metal mines were using electrokinetic densification of slimes and partially classified tailings fill with success (Kilborn and Beak, 1979). Electrokinetic densification can be used to solidify fine tailings to achieve an acceptable strength but the costs associated with the required electrical energy needed for the process may make it not economically feasible (Kilborn and Beak, 1979). Besides economics, safety hazards are associated with the method such as (1) running of electrical lines underground and potential electrocution and (2) the hydrogen gas generated by electrolysis may potentially cause underground explosions if appropriate safeguards (additional ventilation and scrubbing equipment) are not installed and maintained. "One serious problem of dewatering fine tailings would be the risk of recharge seepage water. In saturated state, tailings would remain fluid and it may be assumed that they would require confinement behind reinforced concrete bulkheads designed to withstand the maximum anticipated hydrostatic pressure" (Kilborn and Beak, 1979).

If a decision is made to maximize the volume of fine tailings and slimes placed underground, and return the coarser tailings which contain less radionuclides to a surface impoundment, another impoundment may be needed. The coarse tailings may have to be, at least temporarily, placed in a holding area until sufficient space is available in the original pond for replacement (Kilborn and Beak, 1979). It would be very difficult, especially using a wet system of removal (Section 3.2.1), to keep the original tailings and the newly separated coarse tailings apart in one impoundment (Kilborn and Beak, 1979).

4.2.1.3 Coordination with active mining operation

Underground disposal of tailings does not have to await shutdown of a mine, but can be carried out in concert with mining activity. However, this presents additional problems, primarily in minimizing danger and disruption to the mining. The general lack of concern (5%) over this issue by respondents to this study's questionnaire probably reflects preconceptions that disposal would normally be carried out only after active mining.

In New Mexico, underground workings are backfilled with the classified (sorted) sand fraction of uranium tailings. The whole tailings contains approximately 30% by weight of "slimes" (<200 mesh) which inhibit dewatering and stabilization and increase the potential for bulkhead failure which could be a hazard to workers (Thomson et al., 1986). Removal of the "slimes" fraction of the tailings allows quick gravity draining of the backfill.

Wood (1983) discussed some of the disadvantages associated with the placement of mine-waste materials in an active underground mine:

❶ congestion underground;

❷ potential dust problems associated with pneumatic stowing (Section 4.2.2) and additional ventilation requirements (Section 4.2.1.7); and,

❸ increased humidity underground from hydraulic stowing of wastes (Section 4.2.2) and additional ventilation requirements (Section 4.2.1.7).

Nevertheless, Sassos (1986) reported successful ongoing placement of tailings using conveyors (Section 4.2.2.1) during active mining at the Brunswick Mining and Smelting No. 12 Mine near Bathhurst, New Brunswick. Approximately 5,000 metric tonnes a day were carried underground.

4.2.1.4 Detoxification of tailings

One option to accompany retrieval of surface-impounded tailings and subsequent placement underground is the "detoxification" of the tailings by removal of radioactive and/or non-radioactive contaminants. This basically involves some form of reprocessing and can be done in combination with preparation of tailings (Section 3.2.2). Only 3% of respondents to the questionnaire (Appendix B) mentioned this option.

Raicevic (1980) felt that there are three critical components to uranium tailings:

❶ the sulfide minerals;

❷ remaining radionuclides (mainly ^{226}Ra); and

❸ the fine silica and radioactive dust which can be blown from the tailings impoundment.

These components can be mostly removed with available technology, although costs might be high and problems with storage and disposal would create additional problems. When detoxifying uranium tailings with regard to radionuclides, essentially all isotopes should be removed such as those of uranium, thorium, and radium; otherwise decay will eventually generate new levels of daughter isotopes (Kilborn and Beak, 1979).

Despite costs, the removal of sulfide minerals and radionuclides to yield chemically stable tailings can have both economic and environmental advantages, as described by Raicevic (1980). In his paper, he discussed two potential methods for segregation of pyritic and radionuclide materials:

➊ removal of sulfides and radionuclides by flotation, using a xanthate collector for sulfides and a commercial reagent called "Single Distilled Oleic Acid" as a radionuclide collector;

➋ removal of radionuclides and sulfides by preconcentration, using beneficiation methods prior to extraction by leaching of Elliot Lake ores.

Dreesen et al. (1982) discussed thermal stabilization of uranium tailings to inhibit release of radionuclides. Uranium tailings were heated to temperatures of approximately 1200°C in order to alter the structure and change the mineralogy of the tailings, thereby immobilizing radionuclides, reducing ^{222}Rn emanation power by more than 95%, and decreasing toxic-metal leachability. Dreesen et al. (1982) found that uranium tailings, stabilized or sintered at 1200°C, became slightly fused and experienced a surface-area reduction from 15-17 m^2/g to less than 0.1 m^2/g. At these temperatures there was an increase in amorphous material, which sealed and joined mineral grains, accompanying the large reductions in surface areas. The cost of this type of detoxification, estimated by Dreesen et al. (1982), ranged from \$17.50-32.00/t operating either 450 or 900 t/d facilities. Most of the cost was attributed to the energy requirements of the system. It should be noted that thermal stabilization typically results in a fused mass of tailings, which can no longer be placed (Section 4.2.2) and thus would have to be carried out underground.

4.2.1.5 Stabilization of underground workings and placed tailings

The underground workings and the tailings placed into them may have to be stabilized for various reasons including handling, safety, and maximization of underground volume (discussed in other subsections of Section 4.2.1), as well as subsidence (Section 2.4). This stabilization was a concern for 15% of respondents to this study's questionnaire (Appendix B)

Investigations into the placement of high level waste into salt repositories at Gorleben and Asse II research sites, resulted in the development of series of "safety criteria" dealing with the stability of underground openings (Langer et al., 1988). Due to the generality of the criteria (Table 4-15), they may be adjusted and applied to the placement of uranium tailings into underground workings.

Gaffney (1983) discussed, in his article on underground disposal of coal mine refuse, that pretreatment of the waste materials may be necessary prior to placement of the materials underground. The characteristics of the waste materials and the method of placement will determine the type of pretreatment required such as screening, crushing, desliming, dewatering, centrifuging and/or blending.

Wood (1983) discussed the use of Atterberg tests to determine the effects of moisture on tailings backfill stability after the mine floods, or in a mine with 100% relative humidity. Wood (1983) found that waste backfill containing more than approximately 12% of <200 mesh material will not compact well. However, waste backfill with very little <200 mesh material generally has a low unconfined shear strength (Wood, 1983).

Yamaguchi and Yamatoni (1983), in a paper on backfilling of underground workings for stability, described the use of a 3% cement material consisting of mill tailings sand, volcanic ash

Table 4-15
Safety Criteria Concerning the Stability of Underground Openings
(from Langer et al., 1988)

Safety Criteria	Natural Influences	Technical Influences	Measures
Deformations	Geological Conditions	Cavity Geometry	Geological Exploration
Stresses	Tectonics	Building Processes	Geotechnical Investigation
Failure Mode	Primary Stress	Method of Utilization	Static Design
Bearing Capacity	Mechanical Rock Characteristics	Conditions of Operation	Control Test
Brine Incursion	Gas and Brine Deposits	Temperatures	Mining Measures

and 3% portland cement which was commonly used in Japan's Kuroko ore mines. The physical characteristics of this backfill mixture, and two types of host rock found in the underground workings are presented in Table 4-16.

Table 4-16
Mechanical Properties of Two Rock Types and One Backfill Material
(taken from Yamaguchi and Yamatoni, 1983)

Property	Basalt	Clayey Ore	Backfill Material
Specific Gravity	2.22	3.05	1.91
Young's Modulus (MPa)	1.10×10^4	6.10×10^3	1.72 x 10
Poisson's Ratio	0.154	0.112	0.313
Compressive Strength (MPa)	39.3	3.95	0.17
Tensile Strength (MPa)	4.41	0.35	0.0077

In New Mexico, underground workings were backfilled with the classified (sorted) sand fraction of uranium tailings. The unclassified fraction contained approximately 30% by weight of slimes (<200 mesh), which inhibited dewatering and increased the potential for bulkhead failure which could be a hazard to workers (Thomson et al., 1986). Removing the "slimes" fraction from the tailings increased stability and allowed faster gravity draining of the backfill (Thomson et al., 1986).

The presence of slimes can cause the tailings mass to retain moisture and weight, and can decrease its angle of repose. For these reasons, slimes are often removed from the tailings mass, by cycloning or other methods, before sending tailings underground as backfill. However, for general disposal of tailings, the removal and return of slimes to surface impoundments may or may not be not preferable. The slimes can have greater geochemical impacts than the coarse fraction, but their enhanced moisture retention can minimize oxygen-based geochemical reactions that affect water chemistry (Section 4.1.1). In any case, the slimes create a lower hydraulic conductivity in placed tailings, which can minimize or redirect flow through the mine (Section 4.1.3). However, for actively operating mines (Section 4.2.1.3), stability of slimes-bearing tailings would usually be an issue (Section 4.2.1.5).

Once tailings backfill has been pumped underground, dewatering of the slurry may prove to be a problem especially if the tailings have a high clay or slimes content. Dewatering of clay tailings in surface impoundments has been a problem for the phosphate industry for many years. That segment of the mining industry has been researching methods to increase the rate at which clay tailings dewatering takes place. On the other hand, sand tailings (25-30% solids) formed during the phosphate milling process have been pumped and used without difficulty as backfill for mined-out strip cuts, in dam construction, and/or for land reclamation (McFarlin et al., 1989).

Kilborn and Beak (1979) explained that for underground disposal of uranium tailings to be environmentally advantageous, the finer fraction should be deposited first, as it contains the highest levels of contaminant radionuclides. The problem, they note, is that this finer fraction does not have the type of hydraulic properties required for successful, stable placement underground. Classified tailings have been used by the Canadian uranium industry for many years at various sites, including Beaverlodge and Madawaska which used tailings during cut-and-fill mining operations (Kilborn and Beak, 1979). The strength of backfill comes from the addition of cement to the tailings, and is dependent on: (1) the cement:tailings ratio; (2) particle size distribution; and (3) moisture content of the mixture, which in turn, is control by the initial density of the backfill slurry, the cement:tailings ratio, and the water percolation ratio. The percolation ratio is largely dependent on the slimes content (Kilborn and Beak, 1979). Table 4-17 is an example of the particle size distribution found in Elliot Lake backfill mixtures.

Table 4-17
Particle Size Distribution of Tailings/Cement Backfill from the Elliot Lake Area, Ontario
(from Kilborn and Beak, 1979)

% By Weight @ Tyler Mesh Size	Unclassified Tailings	Classified Tailings		Classified Tailings	
		Tailings 65%	Backfill 35%	Tailings 46%	Backfill 54%
+48	6	0	18	5	8
-48/+65	10	2	28	7	15
-65/+100	14	8	30	11	20
-100/+150	12	14	9	10	16
-150/+200	11	14	4	9	16
-200	47	62	11	58	25
TOTAL	100	100	100	100	100

The interparticle distance in a slurry, or backfill material is determined by the particle size, particle shape and the percentage of solid phase in the slurry (Marcus and Sangrey, 1982). The higher the volume of liquid phase in the slurried backfill, the more additives that have to be added to fill interparticle void spaces. Marcus and Sangrey (1982) felt that a good understanding of the chemistry of the tailings, tailings liquor, and additives would provide some indication as to the type of chemical processes which take place during tailings stabilization. They described four types of chemical mechanisms which are active during tailings stabilization processes:

➊ initial-stage neutralization of free acidity;

❷ precipitation reactions take place with increased pH;

❸ hydrolysis reactions of calcium silicates and aluminosilicates; and,

❹ hydration-crystallization reactions.

"The crystals will grow in time while water is consumed in the above reactions inducing the precipitation of hydroxide and other reaction products. The result will be the filling of void volume between the particles, and finally the bridging between solid particles. At this final stage, the material will be stabilized and may have sufficient strength and stability".

This technology may be applicable to the underground disposal of uranium tailings, by helping to reduce the production of acidic seepage, limiting migration of metals and radionuclides (Section 4.1.2), and increasing the strength of the backfill on curing. Investigations into the compatibility of mine uranium tailings and other alkaline materials should be assessed on a site-specific basis.

Kilborn and Beak (1979) discussed inconclusive backfill mixture tests run by Canada Cement Lafarge Ltd. in which they used flocculants to increase the slimes percentage without jeopardizing the backfill strength. However, stabilization through flocculants or alkaline addition may not even be warranted in situations where disposal requires fluid, destabilized tailings to migrate through underground workings (Section 4.2.2).

Wilkins and Rigby (1991) reported on a model, called MCDIRC, which was developed to estimate the strain around an excavated underground rock opening/stope/shaft after excavation is complete. This type of model may be used to ascertain existing conditions within underground openings prior to backfilling with uranium tailings, and to better predict the impact of the backfill on the underground system.

4.2.1.6 Handling of water

Although underground mines that are not free draining have dewatering systems, the placement of tailings in a moist or saturated manner (Section 4.2.2) could produce sufficient additional water to exceed dewatering capabilities. Consequently, the technical aspects of handling of additional water should be explicitly considered for dewatered mines (Section 4.1.3.1). Thirteen percent of respondents to a questionnaire (Appendix B) agreed.

Thomson et al. (1986), reporting on uranium tailings backfilling operations in New Mexico, felt that the impact of additional water in the underground workings resulting from backfill drainage was not a significant environmental problem. They felt that the drainage water mixed with mine water and became diluted enough not to adversely contaminate the mine water which was pumped to the surface, treated, and released. However, the additional volume of water resulting from gravity drainage of the backfill was significant enough to impact on the pumping and treating facilities.

4.2.1.7 Air-phase requirements

Most of the concerns and issues with degradation of air quality were discussed in Section 4.1.4. This section only discusses a few issues specific to radon emanation and backfilled tailings.

To control the radon concentrations in underground mines, a comprehensive array of ventilation systems is usually required. A computer model was developed and discussed by Kauffman et al. (1987) to simulate the radon release levels underground while backfilling activities are taking place. The model was calibrated with measured data taken prior, and during underground backfilling operations. The physical properties of the original ore and the tailings backfill are listed in Table 4-18.

The levels of radon gas measured at 10 locations during their experiment provided Kauffman et al. (1987) with the range of data presented in Table 4-19. They explained that the range is typical of the day-to-day fluctuations encountered during mining operations.

Table 4-18
Physical Properties of Ore and Backfill
(from Kauffman et al., 1987)

Parameter	Ore	Backfill
Bulk density (kg/m^3)	3,000	2,640
Porosity	0.25	0.41
Emanating coefficient	0.10	0.10
Radium concentration (pCi/kg)	variable	50,000
Effective diffusivity (m^2/s)	2.1 x 10^{-6}	3.1 x 10^{-6}
Radon decay constant (/s)	2.1 x 10^{-6}	2.1 x 10^{-6}

Table 4-19
Range of Radon Concentrations Measured Among Sampling Locations
(from Kauffman et al., 1987)

	Air Inlets (pCi/L)	Air Outlets (pCi/L)
Premining		
Mean	102 - 403	102 - 1573
Standard Deviation	9 - 81	9 - 290
High	121 - 481	121 - 1847
Low	94 - 261	94 - 1400
Mining		
Mean	96 - 347	96 - 1480
Standard Deviation	33 - 62	33 - 220
High	122 - 493	122 - 1547
Low	78 - 294	78 - 1321
Mining and Backfilling		
Mean	115 - 347	115 - 1239
Standard Deviation	6 - 25	6 - 254
High	121 - 365	121 - 1440
Low	107 - 323	107 - 1175

The data used in calibrating the model were taken with hand-held instruments on periodic inspections, usually three times a week. Measurements were made at five air inlet points and six air outlet points, with all other airflow into and out of the test area being blocked by solid barriers. Kauffman et al. (1987) identified two variables which affect the day-to-day fluctuations of radon: barometric pressure, which at lower pressures enhanced bulk convection and added to diffusive flow, and variations in radon emanation rates as mining operations expose more surface wall area and the geometry of the mine changes.

4.2.1.8 Sterilization of ore

"One obvious disadvantage of filling an abandoned mine is the absolute finality implied. In many instances throughout the mining industry, changes in market conditions or advances in mining techniques have permitted previously abandoned mines to be re-opened as profitable ventures, thus making more efficient use of the mineral reserves in those properties. The filling of such mines with unconsolidated slurries would effectively preclude any further extraction of ore" (Kilborn and Beak, 1979). In fact, this "sterilization" of potential ore is regulated or forbidden by legislation in some Canadian provinces. Therefore, careful consideration should be given to the "finality" and regulatory agencies should be consulted before implementing underground disposal of tailings.

Golder Associates et al. (1992), in a report on possible closure options for the Denison Mines in Elliot Lake, reasoned that a major drawback of underground disposal would be that currently uneconomical ore reserves would be "tied up" and would therefore hinder any future plans of extracting these mineral resources from the mine. Apparently, such concerns are not applicable to the low-grade iron ore deposits at Konrad and the salt deposits at Asse, because the areas that would be sterilized by the placement of low and intermediate radioactive wastes would be small compared to the total area of the deposits (Brewitz, 1986).

In any case, the sterilization of ore by tailings disposal is not a technical "finality". In the event that tailings have already been or will be placed underground, a potential method of reprocessing placed tailings and any remaining ore is with a form of in-situ leaching. The uranium industry has already used the in-situ leach method to recover low-grade ore, such as at the Clay West site in Texas (Larson, 1980).

4.2.1.9 Length of time to complete placement

An inordinate length of time to complete the underground placement of tailings would be a disadvantage due to operating and maintenance costs in a dewatered mine (Section 4.1.3.1). However, this does not seem to be a widespread concern because only 5% of respondents to this study's questionnaire (Appendix B) mentioned the issue, and it should not be a concern for a flooded inactive mine where boreholes are used for tailings placement (Section 4.2.2).

In Senes Consultants (1991b) report for Rio Algom Limited, the estimated time for backfilling uranium tailings into two nonoperating underground mines at Elliot Lake was 19-56 years with an engineered backfill method, and 3 years for a slurry backfill method. Therefore, the choice of method for placement (Section 4.2.2) can play a major role in determining the length of time to completion.

4.2.2 Methods of placement

The method of underground tailings placement is dependent on the physical and chemical nature of the tailings (Sections 3.1 and 4.1), and the subsequent ease and effectiveness of transport (Section 3.2.3) and placement methods (Wood, 1983). Both physical and chemical characteristics can determine the types of pumping and/or transportation and equipment that will be used. Wood (1983) felt that the choice of placement method was also dependent on the layout and operations of the mine and geological conditions. However, only 5% of respondents to a questionnaire for this report (Appendix B) mentioned methodology as an issue in underground disposal.

The most important physical and chemical characteristics of the tailings to consider include (Atkins et al., 1987):

➊ particle size distribution;

➋ Atterberg limits;

➌ moisture content;

➍ volatile matter;

➎ ash percentage;

➏ fixed carbon content;

➐ sulfur percentage;

➑ void ratio of the tailings; and,

➒ mineral matter calculated using Parr formula: $M = 1.08\ A + 0.55\ S$

where M = Mineral Matter, A = Ash Percentage, S = Sulfur Percentage.

Gaffney (1983) summarized the underground mine-waste disposal methods existing in 1983 (Table 4-20). Each method is composed of assorted methods of pretreatment (Section 4.2.1.4), surface transport (Section 3.2.3), surface-underground transport (Section 3.2.3), in-mine transport (following subsections), and placement techniques (following subsections), all with their own specific requirements.

Belyaev and Posazhennikova (1983) expressed a preference for wet-placement techniques within inclined workings and dry-placement techniques within relatively horizontal workings. In Poland, wet methods of placement were employed 2.3:1 over dry methods due to lack of efficient equipment and relatively high costs (Grela and Kutyla, 1978).

In this section, methods, requirements, and associated problems encountered with surface-to-underground and underground stowage and transport will be discussed. A summary of the technical feasibility for three disposal systems, namely mechanical, hydraulic, and pneumatic, was produced by Gaffney (1983) and has been reproduced below in Table 4-21.

Table 4-20
Existing Underwater Mine Waste Disposal Methods
(from Gaffney, 1983)

Principles of Stowing and Transport	Normal Pretreatment Requirements For Command Course and Fine Refuse	Surface Transport	Surface-Underground Transport	In-Mine Methods	Backfilling Methods	Disposal Systems
Gravity	Dewatering Blending	Not Applicable	Shaft	Not Applicable	Free Fall	w/Mechanical Surface Transport w/Mechanical In-Mine Transport
Mechanical	Screening and Crushing Dewatering Blending	Belt Conveyor Aerial Tramway Trucks	Spiral Conveyors Skip Cars Mine Cars	Railroad Belt Conveyor	Manual Scrapers/ Slushers Surgers	Complete Mechanical w/Gravity Surface To Underground Transport
Hydraulic	Screening and Crushing Blending	Positive Displacement Pumps Centrifugal Pumps	Pipeline	Pipeline	Blind Flushing Controlled Flushing	Direct Hydraulic w/Mechanical Transport and Natural Head w/Mechanical Transport and Artificial Head (Pump)
Pneumatic	Screening and Crushing Dewatering Blending	Cyclic Stowers Continuous Stowers	Pipeline	Pipeline	Buttock Stowing Lateral Discharge	w/Mechanical Surface Transport w/Mechanical Surface and Underground Transport Hydropneumatic

Table 4-21
Technical Feasibility Summary for Various Disposal Systems
(from Gaffney, 1983)

Disposal System Design	Major Component Types	Alternation Or Disruption Of Mine Operation	Manpower Requirements	Production Personnel Health and Safety	Health and Safety Of Disposal Personnel	Benefits and Disadvantages	Flexibility
Mechanical	Trucks Conveyors Vertical Pipe Wheeled Mine Vehicles	Open Panel Partial Extraction No Interference	More Than Surface Disposal	No Change	Normal Hazards Of Underground Work Minor Temporary Risk Of Spontaneous Combustion	Little Or No Pretreatment Necessary	Good
Hydraulic	Piston Pumps Centrifugal Pumps, Pipes,Blending Agitators Water Supply	Open Panel Partial Extraction No Interference	More Than Surface Disposal	No Change	Normal Hazards Of Underground Work No Dust, Fire or Noise Problems Potential For Minor Water Problems	High Density Fill For Good Subsidence Control	Good
Pneumatic	Trucks, Mine Railroad Cars, Air Compressor Stower, Pipe	Open Panel Partial Extraction No Interference	More Than Surface Disposal	No Change	Normal Hazards Of Underground Work High Level Of Noise And Respirable Dust	High Density Fill High Backfilling Rate Very Labor Intensive	Relatively Inflexible

The original method of underground coal refuse stowage in Europe and later in North America, which is no longer used on a large scale, was manual or hand stowing (Gaffney, 1983; Wood, 1983). Due to intense labour costs, slow productivity, worker safety concerns, and advancements in technology, this method is now outmoded.

Delivery of tailings into an underground mine would be relatively simple if the tailings could be simply dumped down a shaft. But differential settling of the coarser particles would result in plugging of workings near the shaft. This behaviour precludes a simple dump-from-the-surface approach for larger volumes of tailings.

4.2.2.1 Dry placement by hauling

Mechanical stowing systems utilize trucks, rail cars or conveyors to transport the tailings to the disposal area (Section 3.2.3). Many active underground mines already have a truck or rail system for carrying ore to the surface. In an active mine, however, coordinating and scheduling mining with disposal activity would be difficult (Section 4.2.1.3), and a separate system of trucks or rail cars may have to be used (Wood, 1983). In inactive mines, potentially costly upgrading or reopening of workings and/or rail lines may be required if tailings are to be transported underground.

Conveyor systems can move material along belts and are able to handle materials which are not easily dewatered (Bloomfield, 1984). Some other aspects of conveyor systems are that it (Wood, 1983):

❶ can be used in steeper grade areas (±30%) with out loss of efficiency;
❷ requires a fairly uniform power supply;
❸ is easy to control; and,
❹ is low in overall cost with preventative maintenance.

Conveyor systems usually only run in one direction. Therefore, this approach cannot be easily combined with ore-withdrawal conveyors (Wood, 1983).

In addition to conveyors and trucks, tailings can be moved to the disposal location by the use of scraper buckets pulled by wire ropes (Roberts, 1981). Then blade-type packing equipment can be used to ram the waste material into the void subsequently giving some degree of compaction (Wood, 1983). A disadvantage of the scraper method is that the fill is not well packed. Another method is to dump the backfill material onto a short, high speed, conveyor belt, or mechanical centrifugal impeller, which throws it into the void being filled (e.g., Figure 4-9). With the "thrown" method of backfilling, equipment wears out quickly due to abrasion and high impact (Roberts, 1981).

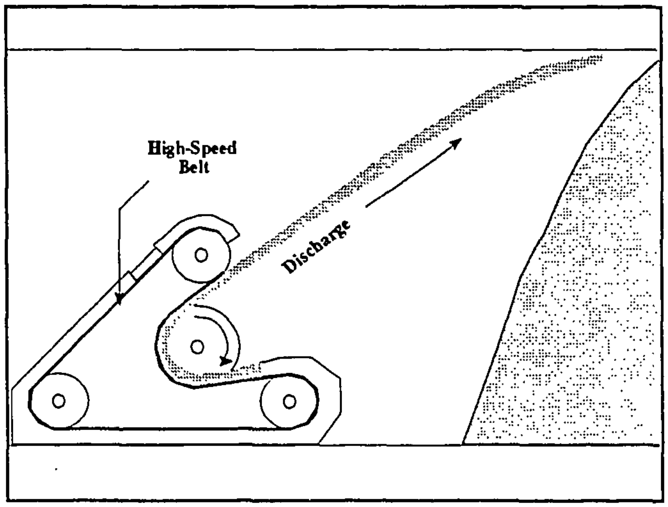

FIGURE 4-9. Mechanical Stowing by High-Speed Belt (from Roberts, 1981).

Dry placement using heavy equipment is labour intensive. In Gaffney's (1983) paper, he states that mechanical stowing was very seldom used due to the "low throughput capacity of machines, incomplete filling of gob, and insufficient density of the stowed mass".

Although an advantage of mechanical stowing is that it does not require water, it can create a dust problem (Kilborn and Beak, 1979). The method could be used with dry cycloning, screening, or other classifying techniques to reduce the volume needing underground disposal (Kilborn and Beak, 1979), and to remove potentially unstable fines. However, any tailings not delivered underground would require long-term maintenance on the surface (Section 3).

Kegel (1975) felt that the expense, physical hazard to workers, congestion underground, dust, and noise far outweigh any benefit obtained from the movement of coal mine gob from the surface in many cases. Kegel did note, however, that backfilling of anthracite coal mines in Pennsylvania was possible because of the specific conditions surrounding the mines such as large areas of steep sloping voids which in many cases were easily accessible from the surface. Kegel also noted that backfilling in European mines is performed because the dense population in some areas does not allow for surface disposal of mine refuse, and thus underground deposition is justified.

The Penobsquis salt mine northeast of Sussex, New Brunswick, utilizes the underground caverns and stopes as storage areas for tailings. The tailings are reduced to 6% moisture by centrifuge, then conveyed to a service shaft and delivered by gravity to conveyers underground (Branch, 1983). A scoop tram takes the tailings from the conveyor end to the disposal areas. The dewatering and disposal of waste tailings underground is considered to be an environmentally safe system for this mine (Branch, 1983).

Rio Algom's potash mine in Sussex, New Brunswick, has been placing tailings underground for over ten years. Although the mine is considered to be a small operation, producing only 640,000 tonnes/year of potash, the result of backfilling activities has meant that "all" of the tailings produced are returned underground with no corrosive salt brine effluent problem on the surface (Whiteway, 1991). "Tailings are dewatered in centrifuges to 8% moisture, conveyed to the shaft, dropped down a 150-mm diameter pipe onto an apron feeder and conveyed to cut-and-fill stopes where it is distributed by diesel load-haul-dump machines equipped with 7-cu-yd ejector buckets" (Whiteway, 1991). This closed system is due to mining of Windsor Group halite (salt) as well as potash from this deposit. The mining of the halite in the footwall produces openings 20 m high, 20 m wide and 275 m long to be used as repositories for the fine fraction of the potash mill tailings. "Slimes settle out of the slurry placed in the salt stopes and the remaining clarified brine is recirculated to the mill as feed stock for the evaporator. The mine currently [1991] has storage capacity underground for 265 million litres of slimes slurry. Salt stopes are laid out with ramp access to avoid the expense of building retention dams" (Whiteway, 1991).

4.2.2.2 Dry placement by pneumatic transport

Pneumatic backfill uses air at high pressure and high-volume throughput to move tailings backfill through pipes to the disposal locations (Bloomfield, 1984; Roberts and Masullo, 1985). An example of a pneumatic system described by Roberts (1981) used a 150 mm diameter feed-pipe which delivered into a hopper, then to a rotary valve which passed the material into a chamber pressurized by compressed air. The material became a fluidized stream in the discharge pipe and was blown into the void to be filled, with a deflector on the end of the discharge pipe controlling the direction of the stream. Roberts (1981) indicated that the air velocity in a 150 mm pipe, discharging fluidized material of specific gravity 2.6 and 75 mm size, would be 24 m/s, resulting in an exit velocity of material at 30 m/s. The radius of bends in the pipe were not less than 1 m, but ideally 2 m to reduce energy losses within the system.

A pneumatic pump with a capacity of 10-20 m^3 an hour was designed for tailings placement in South African mines (Anonymous, 1983). Tailings were pumped consistently at 76-78% solids over 180 m in 100 mm pipelines.

Major advantages of a pneumatic system for large scale waste handling include:

❶ operation is relatively easy (Wood, 1983);

❷ the required space needed in a shaft for operation of the system is small when compared to mechanical methods (Wood, 1983); and,

❸ the backfill material is relatively dry so there is no water to be pumped to the surface for treatment or disposal.

Major disadvantages of pneumatic systems are:

❶ equipment wear from abrasion and impact (Wood, 1983);

❷ dust generation;

❸ noise from equipment (Wood, 1983);

❹ the possibility of an explosion from the buildup of static electrical charges on solid particles as tailings are blown into the void (Wood, 1983);

❺ the potential for spontaneous combustion of backfill materials through oxidation during stowing (Wood, 1983);

❻ the buildup of heat in the transfer pipeline due to the high pressures, friction and impact of particles (Wood, 1983); and,

❼ clogging of the systems pipeline if the moisture content of the backfill is too high.

Gaffney (1983) described two basic methods of pneumatic backfilling used in the placement of coal mine refuse behind a longwall face during mining. "These systems are the conventional method in which the backfill material is discharged at the end of the pipe parallel to the face, and the lateral discharge method in which the backfill material is discharged from the side of the stowing pipe". Wood (1983) characterized pneumatic placement into three categories: (1) dilute-phase conveying; (2) dense-phase conveying; and (3) high-density low-velocity conveying which incorporates compact-phase, slug-phase and pulse-phase flow.

Loss of pressure can be a major problem in pneumatic systems, leading to clogging. Loss of pressure is related to particle acceleration, static head, horizontal transfer, frictional factors and particle size (Wood, 1983). Wood (1983) stated that as the particle size of the backfill material decreased, the corresponding pressure loss increased. The shape of the particles within the backfill also affect how the tailings flow. For example, cubic particles with three axes approximately equidimensional have better flow characteristics than flat or elongated particles, requiring less energy to transport (Wood, 1983).

The relative density of tailings is important as it can determine the fall characteristics in air during placement as well as the abrasiveness of the particles by influencing the impact velocity. Mineralogy of the tailings can also affect the abrasiveness of tailings on equipment. For example, quartz has relative density of approximately 2.65 and hardness of 7 (Moh's scale), whereas pyrite has a relative density of approximately 5.0 and a hardness of 6.

Dry pneumatic placement of mine waste can cause major dust problems, although Litjer (1985) reported this method preferable due to its "cleanliness and dust free conditions". In any case, leaks and/or breaks in pipes and hoses can be a significant hazard with regards to air quality in the underground mine (Section 4.1.4) and could lead to citations, fines and shutdown by government agencies if conditions cannot be controlled by additional ventilation systems (Kegel, 1975). The additional expense of increased ventilation requirements for dry placement of mine waste may not be possible for smaller operations. Kegel (1975) cautions that even though dry placement of mine waste has been used in some mines, the actual type of mining, such as the longwall system in Europe and retreating longwall in the USA for coal mining, can determine the success of the placement.

4.2.2.3 Wet placement as slurry

In addition to the dry-placement methods (Sections 4.2.2.1 and 4.2.2.2.), another general method employs water and is sometimes referred to as "hydraulic" or "slurry" placement. As the following pages demonstrate, this method has received more attention than the others, presumably indicating a general preference for it.

With this method, water and tailings are moved under (Gaffney, 1983):

❶ an artificial head, created by slurry pumps; or

❷ a gravity head, created by a difference in elevation;

❸ a combination of the two heads.

The method with pumps is discussed first, followed by a discussion of gravity drainage.

Wood (1983) described the mechanics of flow within a slurry system:
"the flow of liquids and suspensions results from shear stress and requires the shearing of one layer upon another within the fluid. With simple fluids there is a linear relationship between the rate of movement (shear rate) of the liquid and the stress applied to it. Such a fluid is described as Newtonian. Most slurries however, including coal waste slurries do not exhibit these properties. Such slurries exhibit viscous properties such that the shear stress and shear rate do not have a simple linear relationship."

The causes for non-Newtonian flow, as discussed by Wood (1983), are:

❶ solids concentration;

❷ particle size of the solids. For example, slurries containing fine particles will flow faster and exhibit turbulent, homogeneous flow, while slurries composed of coarser particles will have a slower flow rate and will display laminar, heterogeneous flow with the coarser particles concentrating towards the bottom of the flow; and

❸ particle distribution in the slurry. Coarser particles can be carried by finer particles rather than water.

Nevertheless, some advantages of slurry placement discussed by various authors include:

❶ hydraulic placement is suitable for use in deep mines, in multiple-lift mines with gentle or steep slopes, with spontaneously combustible materials, in conjunction with longwall or room and pillar mining (Gaffney, 1983);

❷ when the mine's mill closes, slurry handling equipment should be available for use in the backfilling operation (Kilborn and Beak, 1979);

❸ tailings can be sorted in hydrocyclones and the finer fraction directed to the underground voids (Kilborn and Beak, 1979);

❹ no dust problem (Kilborn and Beak, 1979; Wood, 1983);

❺ only one transportation medium is involved from retrieval (Section 3.2) to placement

(Kilborn and Beak, 1979; Wood, 1983);

❻ ease of operation when compared to dry methods of stowage (Wood, 1983);

❼ use over long or short distances (Wood, 1983);

❽ use of relatively small space compared to dry methods of placement (Wood, 1983);

❾ handling of a variety of waste material compositions, particle sizes, particle shapes, and densities (Wood, 1983); and,

❿ more tightly compacted fill than dry placement so more material can be disposed underground (Wood, 1983).

Some disadvantages of a slurry method of stowing waste underground include:

❶ potential for floods if the barriers used to hold the tailings fail (Wood, 1983);

❷ seepage of water into the mine wall, floor and ceilings and through fractures causing structural weakness (Wood, 1983);

❸ increased humidity in the mine (Wood, 1983);

❹ requirement of large volumes of water even if recycled (Wood, 1983);

❺ freezing of pipelines during winter (Wood, 1983);

❻ groundwater contamination if tailings are acid generating (Wood, 1983); and,

❼ potential for air pockets in the discharge/transport pipe, as well as selective pipe abrasion and wear (Roberts, 1981).

Although a higher content of fines in a slurry can increase the viscosity and promote homogeneous flow conditions, if the fines are in too high a concentration they can inhibit drainage within the backfill. Inhibited drainage may then increase the hydrostatic pressure behind barriers, and thereby increase the potential for a barrier failure (Section 4.2.1.5; Wood, 1983). Wood (1983) noted that in transportation, particles may degrade through abrasion. Particle abrasion can result in an increase in fines content before the material reaches it final disposal location. The degradation characteristics of the material during transport should therefore be determined before placement.

When using a pump to move the slurried backfill, the pumping pressures required for transportation are determined by slurry flow rate, slurry concentration, slurry relative density (specific density), solids particle-size distribution, pipe size, transportation distance, and friction losses (Wood, 1983). Wood (1983) felt that the blockage of a hydraulic system and/or maintenance of the adequate flow rate to sustain particles in the slurry are the greatest concern when designing a slurry system. Another consideration is the amount of water used to carry the solids in a slurry form, because mines located in relatively dry climates may have an insufficient flow for slurrying. Wood (1983) noted that, by increasing the solids concentration, less water is required and subsequently handled, which in turn can reduce operation costs. However, a problem with increasing the solids concentration is wear and abrasion to equipment and pipes. Abrasion to a hydraulic system is also caused by the slurry mineral concentration, mineral hardness, particle shape, particle size and presence of acidic conditions within the slurry (Wood, 1983).

Atkins et al. (1987) stated that the most important factors affecting pumpability of coal mine tailings are:

❶ particle size distribution;

❷ moisture content; and,

❸ solids concentration.

In the wet placement of tailings, Atkins et al. (1987) indicated that fine particles are important as they "act as heavy media fluid that effectively suspends the larger particles during transportation". Fine particles can also affect the cohesive strength of the slurry mixture. Since fine tailings are less abrasive than the coarser fraction, the type of equipment used can be influenced by the ratio of fines vs. coarse material (Atkins et al., 1987).

Atkins et al. (1987) felt that the moisture content, or degree of saturation, is one of the most important factors governing the pumpability of mine tailings. Variations in moisture content control the quantity of material that can be pumped in relation to the pre-determined compressive strength of the material. However, Atkins et al. (1987) noted that the chemical properties of the tailings are also important for stabilization. The ease of pumping the tailings is directly related to void ratios and inversely related to solids concentration.

Considerations for choosing an optimum slurry mixture, pumping and transport systems are (Atkins et al., 1987):

❶ predetermination of compressive strength and liquid conditions of any slurry mixture or packing, so that throughout the disposal process, from the source of the slurry to the final point of discharge, the slurry should remain in suspension within the pipeline;

❷ determination of the equipment/pipeline layout and design requirements (ie. choosing an appropriate pump), so as to eliminate all potential (and any existing) pipeline blockages, and to ensure an adequate capacity and head are maintained;

❸ the overall wear on all equipment and pipes must be minimized; and

❹ there must be careful monitoring of changes in slurry characteristics (ie. moisture content, pulp density), and automatic changes to the disposal system to accommodate those changes must be possibie.

Popovich and Adam (1985) discussed the need for preliminary testing on the feasibility of returning waste underground. Samples should be taken during and after the initial backfilling test prior to transport, at the discharge point, and from the backfilled area of the mine to ensure a optimum mixture which will allow proper drainage. The samples should be tested for size consistency before, during and after transport and placement, chemical composition, float and sink analyses, permeability of deposited material, specific gravity, maximum and minimum densities, moisture content, repose angle, and undergo compression tests and shear tests. Visual inspection of the placed material to determine stratification of the waste and efficiency of backfilling (average distance to roof) should also be performed.

To optimize slurry placement of tailings, the workings should be dewatered. If the mine is flooded when the slurry is being placed (section 4.1.3.1), the tailings will only travel approximately 100 m and then plug the workings so that further placement of tailings would not be possible (Senes Consultants, 1991a). If the mine is dewatered, the tailings slurry will flow as a fluid and fill all available void spaces with the exception of areas where no air can escape to allow displacement by the slurry.

When slurry is pumped into underground workings through boreholes drilled from the surface, an increased hydrodynamic pressure keeps the solids in solution longer as the slurry moves away from the disposal borehole (Wood, 1983). As long as the velocity of the backfill is maintained by the pumps, preferential channels will develop and carry the slurry out to the boundaries of the fill along paths of least resistance (Wood, 1983). This concept is illustrated in Figure 4-10.

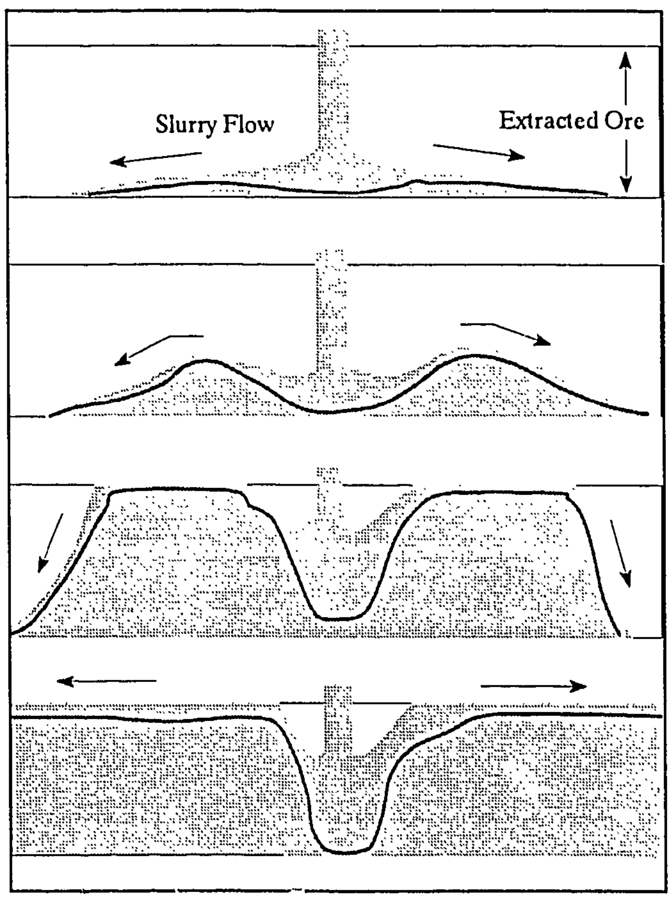

FIGURE 4-10. Stages of Mine Filling by Pumped-Slurry Injection (Wood, 1983).

Wood (1983) found that some studies reported that more backfill could actually be placed under flooded conditions, using a slurry which contained finer materials, and with greater pumping pressures and higher velocities. This may also require less boreholes and a shorter time to complete the disposal program. However, with higher pumping pressures comes an increase in equipment wear and maintenance. One problem with this method and its variations is the general inability to determine when the underground voids are full (Wood, 1983).

An example of a hydraulic placement system, presented by Roberts (1981), handled backfill material fed from a bunker to a mixing cone, which was added at a solid:water ratio by volume of about 1:1 to 1:3. The mixture was fed into a discharge pipe, from 75 to 150 mm diameter depending on the size of the backfill material, and transported underground by gravity flow when the minimum gradient was approximately 0.1, or otherwise by pumps. At the underground discharge point, cavity sock cloths held by wire mesh were fixed to the walls behind which the slurry was pumped. The barrier held the slurry solids allowing the water to drain, and the water was collected in a sump and pumped back to the surface. Once the water had drained, a compacted solid mass formed. Void spaces above this mass could then be filled with more slurry material (Roberts, 1981).

An underground study mine located in Lee County, Virginia, and Harlan County, Kentucky, was the site for design and testing of a hydraulic transport and backfill system to take coal mine refuse back to the underground mine for disposal. Within the backfill area, filter barricades were designed to retain solids while permitting water to drain through to collection sumps. The design of the filter barricades is shown in Figure 4-11 (Popovich and Adam, 1985).

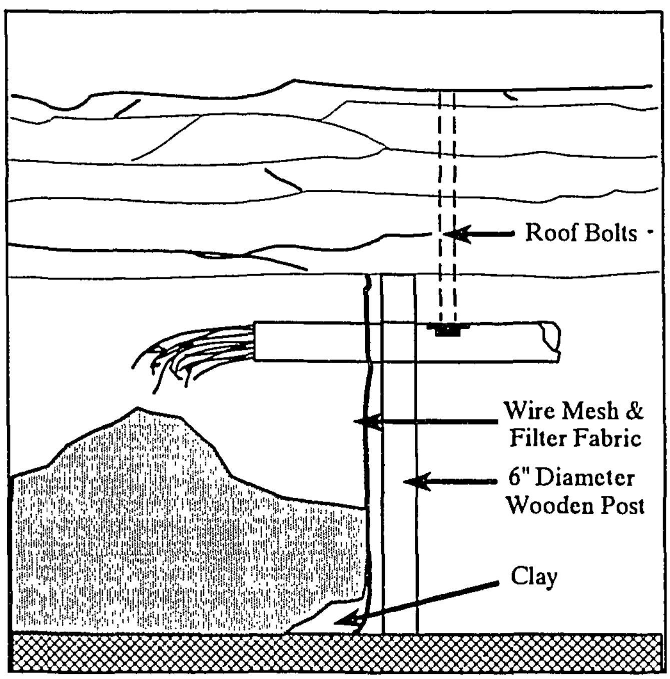

FIGURE 4-11. Filter Barricade (from Popovich and Adam, 1985)

Falconbridge Nickel Mines in Ontario have used computer controlled devices to manage the tailings/cement ratio of their hydraulic tailings backfill program (Petterson, 1975). Falconbridge went into production in 1968, and in 1975 was using a million tons of backfill a year. The tailings were cycloned to draw off the coarse tailings which were placed in storage, while the fines were pumped to a tailings pond. The coarse tailings, when needed, were drawn from storage and mixed with cement at a ratio of 32 parts tailings to 1 part cement for normal filling, and 8 to 1 when working floors were poured. These ratios were appropriate for the type of coarse tailings generated during the milling process. The use of computer controlled equipment ensured that correct ratios were being blended to prevent too weak a mixture, or too strong and expensive a mixture. The mixture of tailings, cement and water was checked to ensure proper density and flow rate before being poured down 4-inch-diameter diamond-drill holes at a rate of 200 tons per hour at 65% solids (Petterson, 1975).

Thomson et al. (1986) reported on backfilling operations in New Mexico, where the segregated sand fraction of uranium tailings was pumped underground as a slurry consisting of equal amounts of sand and treated mine water. Ackman (1982) reported that the disposal of ARD sludge from treatment ponds was commonly "either pumped or trucked to boreholes drilled into underground abandoned deep mines or inactive operating portions of producing mines".

Goedde (1980) discussed a large-scale research project undertaken by Bergbauforschung GmbH in Essen-Kray, Germany. The "slurry" transport tests were undertaken to measure physical parameters such as forces, impulses, densities, stresses, pressures, torques, and temperatures of the system. On completion of the testing, Goedde (1980) stated that the hydraulic movement of materials had a potential for lower accidents, easy automation, and good economic feasibility.

Thomson and Heggen (1982) reported that slurry backfilling of 70% solids by weight (50% by volume) containing the sand portion of uranium tailings had been done in the Grants Mineral Belt of New Mexico. The slimes fraction were not backfilled with the sand fraction, as improper draining caused structurally unstable mud, whereas solely using the sand fraction resulted in quick dewatering. When the backfill was kept drained, the material was found to take on a rock-like form due to interparticle cementation.

Kegel (1975) cautioned that although wet placement of mine waste has been used in some mines, the type of mining (i.e. longwall system in Europe vs retreating longwall in the USA for coal mining) has a significant effect on the success of the practice. Kegel (1975) indicated that wet placement of refuse is practical in abandoned and worked-out mines and probably works the best, but was not practical in active mines where placement in one area requires measures to restrict it in another area.

The potential slurry placement of low and intermediate level waste materials from the nuclear industry was discussed by Kühn (1983). The waste would be mixed with cement and pumped underground through a 50 mm diameter pipe, and would solidify in-situ. This disposal method was proposed for placement of low and intermediate nuclear waste into salt formations in Germany.

In the case of the Quirke and Panel Mines in Elliot Lake, Ontario, Senes Consultants (1991b) indicated that basically all of the mine (95%) could be filled with slurry, with the practical limit for pumping the slurry and maintaining fluid properties being 50% solids. Underground filling of these mines would take 3 years to complete (Senes Consultants, 1991b). To optimize available storage space, underground dams and drains would have to be constructed at critical locations to prevent plugging of the shaft by tailings. Mine dewatering would allow more slurry to be pumped underground filling additional voids. Although this may seem like an attractive method of underground disposal of tailings, problems such as dam and dyke failure, plugging of the shaft and workings, and plugging of the dewatering pumps with solids would all have to be considered before initiating the placement program (Senes Consultants, 1991a).

The MacLellan Mine near Lynn Lake, Manitoba used a cut and fill method of mining its gold and silver deposits. Stopes were hydraulically backfilled with uncemented sand screened to minus 1.27 cm (Voisey and Spencer 1990).

A slurry-backfill system was designed for the No. 4 Mine at the Wolf Creek plant at Pilgrim, Kentucky (Ketron Inc., 1982). The system included pumping 150 tons an hour of 45% solids by weight at roughly 1000 gallons a minute. Slurry velocity was 10 feet a second in a 6-inch pipe, and dual delivery pipes were included for contingency. Bulkheads were designed to hold and settle the slurry, but the major problem was expected to be deterioration of the mine floor by water draining from the backfill.

As explained at the beginning of this subsection, a slurry can also be moved underground through gravity drainage. However, gravity filling of stopes is usually restricted to workings with very steep slopes. Gaffney (1983) and Wood (1983) both cited a 40-50% gradient requirement without pipes, and down to 30% with pipes. The use of vertical or inclined chutes can also be used in addition to boreholes drilled into the final deposition location (Wood, 1983). Wood (1983) reported that material has been moved by gravity transport to depths of 600 m in the former Czechoslavakia.

Gravity transport requires that a removal system be in place underground to prevent clogging of the pipe or chute if the area being filled extends horizontally beyond the pipe/chute discharge point (Wood, 1983). Gravity placement was reported by Wood (1983) to be a relatively inexpensive method of backfilling if underground conditions are appropriate. He also noted that high placement capacities of up to 200 m^3/h have been achieved.

If gravity-driven placement is accomplished through boreholes, the continual drilling of new boreholes and the construction and maintenance of roads would make this an expensive and laborious method (Kegel, 1975). Also, there is the issue of accurately drilling boreholes into selected underground workings. In many cases, due to the limited precision of mapping of underground workings and the difficulty of drilling to an exact location in rock, this may not be a practical method (Kegel, 1975; Aljoe and Hawkins, 1991).

Contamination of groundwater could result from the deposition of wet slurried tailings by either borehole disposal or other slurry methods (Kegel, 1975). If borehole disposal is used, careful cementing and sealing of the casing should be completed through all aquifers and into an impermeable stratum to prevent any hydraulic connections.

Wood (1983) noted that both sand and mine waste have been hydraulically introduced into flooded and non-flooded underground mines by hydraulic "flushing" of a 30-50% solids material in water. The hydraulic "flushing" method utilizes boreholes drilled from the surface into the underground void, where the slurry is injected by either pumping or by gravity placement (Falkie et al., 1974; National Academy of Science, 1975; Wood, 1983). "Controlled flushing" involves having workers underground to place distribution pipes and build barriers if necessary. This method provides for better filling as there is more supervision of where and how the waste is placed (Wood, 1983). Thomson (1989) provided an example of controlled flushing at a coal mine in Britain.

"Blind flushing" is used when it is not possible to safely position workers and equipment underground due to caving, flooding, etc. (Wood, 1983). This method of placement involves the use of vertical boreholes drilled into the workings from the surface, but there is no active control underground of placement (Wood, 1983). The results of this method (Figure 4-12) include gravity-fed slurry building up into a conical pile beneath the disposal opening until the cone builds up to the mine roof and no more fill will enter (Wood, 1983). More backfill must then be placed through other boreholes or shafts.

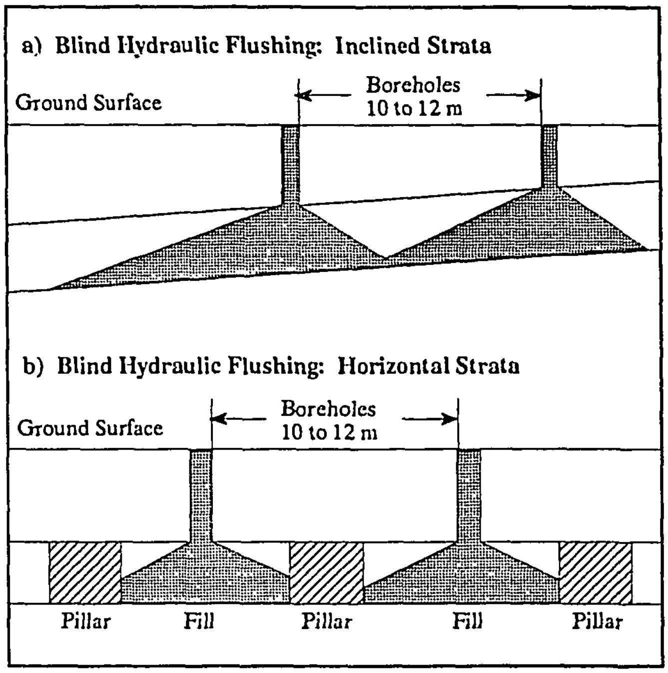

FIGURE 4-12. Blind Hydraulic Flushing: a) Inclined Strata; b) Horizontal Strata (from Wood, 1983).

Through blind flushing, the maximum amount of placed tailings and the optimum distance between holes are determined by (1) characteristics of the tailings, such as grain size, water content, and its ability to flow away from the deposition borehole, and (2) characteristics of the underground workings, such as the dip of workings, extent of caving and/or flooding, and sizes and shapes of voids (Wood, 1983). Wood noted that boreholes usually need to be closely spaced, and up to hundreds of holes may be necessary to accommodate the volume of wastes usually requiring placement underground (Figure 4-12). He also noted that volumes of 50 - 760 m^3/borehole have been achieved, and that usually only about one third of available space can be filled this way, making this an expensive less-than-optimum disposal method.

At Cominco's Polaris Mine on Little Cornwallis Island in the Canadian High Arctic, underground mining takes place within permafrost (Keen, 1992; Scales, 1982). A frozen plug is put in place at the uppercut stope access before commencing with backfill operations. Backfilling of stopes is accomplished by two methods depending on the season, summer or winter. In the winter, backfill material, silty limestone and shale mixed with mine waste is dumped from the surface down a raised borehole into the stope. Stopes at the Polaris Mine are 15 m wide and 100-150 m long. A D-8 dozer mixes the backfill material with water to make a tight-filling backfill. In the summer, the backfill material has enough free water to be dumped down into the stope and used directly. Backfilling at Polaris is a year-round activity and, in

1991, 315,000 m^3 of quarried material and 45,000 m^3 of mine waste was used (Keen, 1992).

4.2.2.4 Wet placement as paste

Golder Associates et al. (1992) wrote that traditional backfill placement was generally by hydraulic backfill of the coarse fraction of the tailings (Section 4.2.2.3). However, Golder Associates also reported on recent developments in the placement of total tailings as a dense paste (Aref et al., 1989), which for example can be placed by concrete-style pumps. Cement is added for stability to prevent liquefaction or in the event that adjacent rock is to be mined.

In this method, "run-of-the-mill" mine tailings are piped to an underground location close to the stopes to be filled. At this underground location, a centrifuge dewaters the 60% solids slurry to 75% solids by weight. A small amount of cement is then added to the thickened slurry, and transported by concrete pump to the stope for placement. An air-assisted nozzle is used to deposit the paste into the mined-out void. After the backfill is placed, there is very little drainage. The water separated during the thickening step is collected in the existing mine sumps and pumped directly back to the surface. This hydraulic technique can be used in mines where backfilling is required close to active mining areas.

The advantages to the paste backfill technique are (Golder Associates et al., 1992):

❶ the backfill is relatively impermeable, reducing potential oxidation;

❷ surficial tailings are moved to a less accessible area (up to 60% of the tailings);

❸ the backfill supplies good structural support for the underground;

❹ there is no slurry water, as with hydraulic backfilling, to pump back to the surface [authors' note: although this method apparently leads to some water production underground due to conversion of the slurry to a paste);

❺ dewatering problems are essentially eliminated;

❻ the bulkhead requirements to impound the paste are minimal; and

❼ environmental liabilities can be potentially reduced.

Although this method seems very promising, it may not be appropriate for mines which have already closed, such as the Denison Mine in Elliot Lake (Golder et al., 1992).

Barium/radium-sulfate treatment sludge, as an analog to fine tailings, can be backfilled as a paste in Rio Algom's Quirke mine in Elliot Lake (CANMET and Kilborn, 1981). Testwork showed that optimum procedure was to dredge the sludge (Section 3.2.1.2) and pump it to a processing facility. This facility removed coarser particles and dewatered the remaining fine particles with pressure filters to 25% solids.

4.3 Costs

Gaffney (1983) discussed a report prepared by the National Academy of Science in which the 1974 cost of pneumatic backfilling of underground workings with coal mine wastes was estimated to range from $1.18 to $1.86 per ton of clean coal, while hydraulic backfilling was estimated at $1.43 per ton of clean coal. As a comparison, Gaffney (1983) reviewed a U.S. Bureau of Mines study published in 1980 which reported 1978 cost estimates for coal mine waste surface disposal at $1.77 per ton of clean coal for mechanical disposal to $2.90 per ton of clean coal for pneumatic surface disposal, while hydraulic underground disposal was estimated at $1.84 per ton clean coal.

Based on demonstration projects funded by the U.S. Bureau of Mines and the Department of Energy in 1981, detailed cost estimates for underground disposal of coal-mine refuse were reported at $0.79 to $1.43 per ton of clean coal under optimal utilization conditions, whereas surface disposal costs were estimated at $0.70 to $1.50 per ton of clean coal (Gaffney, 1983). Similar costs were reported by Atwood (1974). Although all the costs reported by Gaffney (1983) were for specific site models, a tendency for the costs of underground disposal to be economically viable seemed evident. This may become more evident as the costs for surface reclamation increases sharply in cost (Sections 3.2.4 and 3.3).

The British Coal Corporation must dispose of up to 42,000,000 tonnes of waste annually from its underground mines (Astle and Knight, 1990). Surface placement of this waste costs between $0.80 and $3.3/tonne, which can increase to $11.50/tonne if the waste must be transported up to 20 km for disposal. Corresponding costs for underground slurry placement were $2.65/tonne of slurry or $3.50/tonne of solids, including capital depreciation, operating costs, and materials (Astle and Knight, 1990). Monetary benefits from reduced subsidence and roadway maintenance were not included. Similarly, Popovich et al. (1984) estimated costs at $1.60 per ton of backfill for slurry placement.

Grela and Kutyla (1978) reported that Polish mines at that time preferred wet-placement methods due to costs and lack of efficient pneumatic equipment. Costs for wet placement were reportedly 18.9 zlotys/ton whereas dry placement was 51.0 zlotys/ton.

At a base-metal mine in Germany, the placement of cemented tailings was converted from dry-pneumatic to dry-hauling with a "slinger belt" (Rohlfing, 1983). The conversion was due to the lower costs and higher performance of the belt method.

Culver et al. (1982), while discussing close-out options for the Elliot Lake uranium tailings area in Ontario, considered the removal of 93-95% of the pyrite and approximately 60% of the radium from tailings through underground disposal. Estimated costs were $1.00/tonne.

Senes Consultants (1991b) estimated that, to backfill 15 x 10^6 m^3 into Elliot Lake mines with engineered uranium tailings fill, would cost $350,000,000 CDN over a 19-56 year period. To backfill 20.6 x 10^6 m^3 (26% of available uranium tailings) underground with a slurry method would cost $66,500,000 CDN over a 3 year period. A breakdown of costs for the two disposal/placement methods is shown in Table 4-22 (Senes Consultants, 1991b).

Table 4-22
Cost Summary for Underground Disposal of Quirke and Panel Mines Uranium Tailings (from Senes Consultants, 1991b)

Engineered Backfill Method			
Dredging	30 x 10^6 m^3 * 1.3 tonne/m^3 @ $3/tonne	$117,000,000	
Mill Thickening/Cyclones/Backfilling	15 x 10^6 m^3 * 1.3 tonne/m^3 @ $10/tonne	$58,500,000	
Mine Rehabilitation/Operation	$5,000,000/yr for 19 years	$95,000,000	
Mine Water/Tailings Pond Treatment	$500,000/yr for 19 years	$9,500,000	
Hearings/Reports/Design and Contingency	(allow 25%)	$70,000,000	
Total Cost for Engineered Backfill			$350,000,000
Unit Cost for Engineered Backfill			$18/tonne
Slurry Placement Method			
Dredging Panel and Quirke	20.6 x 10^6 m^3 * 0.73 tonne/m^3 @ $3/tonne	$45,100,000	
Thickening, Pumping to Mine/ Reclamation	Capital for Pump Station/Pipelines	$6,000,000	
	Operations $0.14/tonne * 15 x 10^6 tonne	$2,100,000	
Hearings/Reports/Contingency	(allow 25%)	$13,300,000	
Total Cost for Slurry Placement			$66,500,000
Unit Cost for Slurry Placement			$4.4/tonne

Dreesen et al. (1982) discussed "thermal stabilization" of uranium tailings to inhibit release of radionuclides (Section 4.2.1.4). The cost of this type of remediation as estimated by Dreesen et al. (1982) was relatively high, ranging from $17.50 to 32.00/tonne operating either 450 or 900 tonne/day facilities, due to the energy requirements of the system.

Osborne (1982) reported on a model developed to determine the cost associated with dose limitations in managing uranium tailings. The model takes the costs for various management options versus the effectiveness of the options to control the dispersion of radionuclides in the biosphere calculated from the radiation dose. Four reference sites, one from Canada, two from the United States, and one from Australia were compared for incremental cost-effectiveness of their proposed management options. The option of placing uranium tailings underground was considered only for the reference sites of the U.S. Underground disposal of uranium tailings was found to be very effective in controlling the collective dose commitment in a situation where airborne contaminants are the greatest concern, such as in arid areas of the U.S. However, it was also determined through this cost-effect analysis that a small incremental decrease in collective dose commitment using this management method had a very large associated cost. This type of analysis may be very useful to mine and regulatory personnel when determining the most appropriate management option. Osborne (1982) points out in his paper that this type of analysis has to based on the regulatory specifications of the collective dose commitment level, so reasonable cost-effective data can be produced.

5. OTHER ISSUES

Many issues pertaining to tailings retrieval from surface impoundments and subsequent placement underground have been discussed in Sections 2, 3, and 4 of this report. There are a few topics, however, that could not be integrated into those sections, and are thus discussed here.

5.1 Biological Implications in Underground Disposal and Residual Surface Tailings

The biological implications in underground disposal include such diverse topics as human-worker issues and bacteriological control of contaminant release and migration. Such biological implications are not a major objective of this study, but this subsection summarizes some information and provides references for further information. Twenty percent of respondents to this study's questionnaire (Table 1-1 and Appendix B) mentioned one or more aspects of this topic.

To determine if the benefits to mankind and the environment are greater than the financial costs incurred in moving surface-impounded tailings underground, the risks to humans and the environment from these tailings must be determined (Atomic Energy Control Board, 1983a and 1983b; Hamel et al., 1982). Culver et al. (1982) cited research in the Elliot Lake area of Ontario which showed that the radiation exposure level to persons living adjacent to a surface tailings impoundment, prior to any reclamation, was approximately 10 mrem/a. They also note that emissions of radon and radioactive dust reached background levels within distances of less than 1 km from the tailings. In light of the general remoteness of most tailings (greater than 1 km from humans), the impact of radiation exposure to humans seemed low. However, radiation exposure to surrounding vegetation and wildlife must be evaluated separately.

Although milling of uranium ore provides a saleable product, Sethness and Holmes (1979) estimated that approximately 85-97% of total radioactivity contained in the original uranium ore was carried to tailings impoundments as milling wastes. The most potentially harmful radioactive parameter being the ^{226}Ra isotope which emits gamma radiation and alpha particles. The potential health problems associated with gamma radiation include:

❶ it is very penetrating; and

❷ it causes cell damage as it travels through the body (Sethness and Holmes, 1979).

The primary potential health problems associated with alpha particles is that, even though they have very little penetration capability, they can cause extensive cell damage (Sethness and Holmes, 1979).

Even though the radioactive content within the tailings is similar to the original ore after mining and milling, the radionuclide and heavy metals contaminants are more available for transport by air and water into the environment (Fry, 1982). However, Fry (1982) noted that direct exposure to uranium tailings over long periods of time would be required to produce significant radiation-caused health effects. He thus concluded that the most significant problem with uranium tailings is the sheer volume of tailings requiring management and long-term containment.

When an uranium tailings impoundment is disturbed, the risk of public exposure to radiological contaminants increases. For both disturbed and undisturbed tailings, Rogers (1978a) cites "the principal environmental radiological contaminants and associated health effects from uranium tailings are related to the radionuclides of the ^{238}U chain: primarily ^{230}Th, ^{226}Ra, ^{222}Rn, and ^{222}Rn daughters". The main exposure pathways to humans are (Rogers, 1978a):

❶ inhalation of ^{222}Rn daughters;

❷ external whole-body gamma exposure;

❸ inhalation of windblown tailings; and

❹ ingestion by man of ground or surface water contaminated with radioactivity, that is, "ingestion of ^{222}Rn ground-deposited daughter ^{210}Pb" (International Atomic Energy Agency, 1981).

Radon gas is an inert gas which can diffuse through pores in rock and into the atmosphere. The concern over exposure and inhalation of radon gas comes from the fact that decay of radon gas exposes the lungs to radiation and also deposits small amounts of solid lead within the lungs (Kauffman et al., 1987).

Lush et al., (1982) reported on an assessment of the aquatic pathways along which radionuclides can reach humans. The radionuclide exposure and dose calculations for uranium and thorium decay series were modelled for the Elliot Lake area which drains into the North Channel of Lake Huron. The potential for long-term radionuclide sediment sinks was also modelled. Lush et al. (1982) described suspended particulate concentrations in the receiving aquatic systems, settling velocities of the particulates, and the solid-aqueous phase distribution coefficient associated with each radionuclide as the key variables in determining population dose commitments. Secondary variables included the rates at which radionuclides leave the tailings, the rates at which radionuclides enter the receiving waters, water-to-food transfer coefficients, the rate of water and fish consumption, and dose conversion factors for ^{210}Pb and ^{210}Po.

Kilborn and Beak (1979) reported that "the estimated annual absorption of ^{226}Ra per capita living solely on vegetation grown near a tailings site would be 0.086 μCi/yr". This number was calculated under the conditions that the concentration in vegetation was 2,850 pCi/kg with a 150 kg annual per capita consumption, and a gastro-intestinal absorption of 20% the total amount of vegetation ingested.

Another potential exposure pathway for the public to radioactive constituents is through ingestion of animal products which have digested contaminated vegetation and tailings (Walsh, 1986). Kilborn and Beak (1979) cite the following example of potential exposure. If the concentration in vegetation through plant uptake near a tailings site was 2,850 pCi/kg for ^{226}Ra, and a cow consumed 50 kg/day of this vegetation, the cow would ingest 142,500 pCi/day or 50 μCi/yr. The beef and milk concentrations were calculated as: (1) beef concentration = 10^{-3} (^{226}Ra transfer coefficient) * 142,500 = 142.5 pCi/kg; and the (2) milk concentration = 2 x 10^{-4} (^{226}Ra transfer coefficient) * 142,500 = 28.5 pCi/L. If a person consumes 50 kg of this beef and 150 L of the milk, using a 20% gastro-intestinal absorption, he would absorb 0.114 μCi/yr of ^{226}Ra (Kilborn and Beak, 1979).

Blanchard et al. (1981) determined that the increase in lifetime fatal cancer risk from airborne and waterborne contaminants at minesites is only 0.13% greater than the normal cancer risk to an individual from all causes. The calculations were done using models of underground and surface uranium-mining operations in the western U.S.

An example of the misuse of uranium tailings resulting in public exposure was reported by the Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects (1986). Uranium tailings from the Climax Uranium Mill, Grand Junction, USA, were used for the construction of schools, homes, and other buildings. A total of nearly 7,00 sites were contaminated, resulting in the passage of public laws for remedial action regarding uranium mill tailings. The report also listed 24 inactive uranium mill tailings sites with 25,000,000 tons of radioactive tailings, having uranium contents ranging from 0.005% to 0.040% U_2O_3. Of the 24 inactive uranium mill tailings sites, 14 are close to a river or stream, 12 have evidence of wind or water erosion, and 9 probably have contaminated groundwater. Licensed uranium mill tailings sites as of 1983 numbered 27 (14 active, 10 standby, and 3 closed), covering 4,344 acres, and holding 174,700,000 tons of uranium tailings.

Research into bacterial interactions within natural systems seems to imply that certain bacteria can help to retard or accelerate chemical reactions within non-radioactive, inorganic environments in and around tailings. If retardation of negative chemical reactions, such as the oxidation of pyrite resulting in acidic waters, is possible, then the addition of bacteria to uranium tailings prior to placement underground could assist in stabilizing potential contaminants within the placed material.

Dowse et al. (1975) felt that the migration and persistence of pollutants in a groundwater system were controlled in part by biological conditions including aerobic processes, anaerobic processes, bacterial food supply availability, and biodegradation. Thus biological processes could be important if groundwater quality is degraded upon retrieval and placement of tailings (Sections 3.1.4 and 4.1.2).

McCready (1986) reported on the initial stages of some experimental underground bioleaching of uranium ore. Mined out workings were flooded with water supplemented with nutrients and the bacteria *Thiobacillus ferrooxidans*. Experimentation took place at the Denison Mine, Elliot Lake, Ontario and started in 1984. McCready (1986) found that bioleaching of stopes containing an organic carbon source, such as in an underground maintenance area, resulted in the flourish of fungi which coated the uranium ore and prevented bioleaching by the added *T. ferrooxidans*. The fungi was also found to accumulate up to 12% of its body weight as uranium. This work introduced the possibility of using the fungi to concentrate uranium as a way to retard movement of uranium contaminants underground, or as a preparatory step for conditioning uranium tailings before placement by removing excess uranium. The fungi may also be useful in helping to reduce acid generation in backfilled tailings containing pyrite, by inhibiting bacterial activity. Such leaching presumably also took place at the Stanrock Mine in Elliot Lake, where MacGregor (1966) reported recovery of U_3O_8 from the underground workings.

Bench-scale leachability experiments were carried out by Constable and Snodgrass (1987) on 20-year-old tailings and fresh samples from the leaching pachucas, the partial neutralization tanks, and the final neutralization tanks located at a Rio Algom mill in Elliot Lake. Initial leaching resulted in the "dissolution of gypsum and minor amounts of carbonates" which in turn explained measured levels of conductivity, total dissolved solids and pH in the leachate. Exposure of pyrite to oxygenated water resulted in pyrite oxidation and "colonization of the tailings by pyrite oxidizers, dissolution of carbonate minerals, and rapid decrease in pH to 2-3". Higher pH values were observed but were determined to be caused by the flux of water diluting the mass of acidity produced by pyrite oxidation. They found that a continuous mode of water application appeared to slow the rate of colonization compared to a batch mode of application. Constable and Snodgrass (1987) also found that during pyrite oxidation, the rate of total dissolved solids produced is independent of flushing rate. This implies that, once the bacteria have colonized the tailings, the pyrite oxidation rate is independent of the hydraulic loading (flushing) rate, provided sufficient oxygen is available to allow pyrite oxidation to continue.

In determining the adsorption capacity of ^{137}Cs in the host rock, West et al. (1991) found that the contamination of a Fuller's Earth (calcium montmorillonite), which was considered a potential backfill material, by KSRB (Konrad mine [Germany] *Desulfovibrio desulfurican* bacteria) microbes caused erratic results under anaerobic conditions. Anaerobic conditions generally are favourable conditions for SRB's (sulfate reducing bacteria) and are generally used in batch experiments for determining the potential adsorption capacity of host rocks within radioactive waste repositories. But the results from West et al. (1991) suggested that irregularities may appear during these types of experiments. However, such notable irregularity was not observed in aerobic conditions (conditions unfavourable for KSRB growth). These results may indicate that the KSRB's may have adapted to their environment and that generalized statements regarding the behaviour of SRB's may not be appropriate under varying site conditions.

5.2 Monitoring and Corrections of Problems

This report has summarized a vast array of concerns and issues pertaining to surface retrieval and underground disposal of uranium tailings. Upon carrying out such disposal, another reasonable concern is assurance that the disposal has not resulted in additional problems, or at least that any additional problems can be corrected. This requires monitoring in and around the disposal site (Lakshmanan, 1985). Five percent of respondents to this study's questionnaire (Appendix B) raised this concern.

For low-level radioactive wastes, the U.S. Nuclear Regulatory Commission in its Code of Federal Regulations, Title 10, Part 61, 1982 stated that compliance with the regulation requires an applicant to "demonstrate that the groundwater monitoring system, as a component of the environmental-monitoring program, is capable of providing early warning of radionuclide releases before they reach the site boundary. Applicants should synthesize hydrogeologic information into conceptual and analytical models to develop sufficient understanding of the rates and directions of groundwater flow and contaminant transport to demonstrate that the groundwater monitoring system is capable of providing an early warning of radionuclide releases. Specific hydrogeologic information needs for such demonstrations may include the following types of information: hydraulic head, hydrogeologic unit geometry, hydraulic conductivity, effective porosity, storage characteristics, sorptive and attenuating characteristics, contaminant source terms, and supporting information" (Bloomfield, 1984).

For underground placement of tailings, Popovich and Adam (1985) discussed the need for sampling of water to delineate effects on groundwater chemistry. They suggested testing for pH, alkalinity, acidity, total and dissolved metals, total suspended solids, total dissolved solids.

For surface-impounded tailings, extensive monitoring of their effects on the receiving environment has been conducted for decades. This work is relevant for residual tailings that could not be placed underground. Markose et al. (1982) reported on an environmental surveillance program to monitor the effects of uranium tailings pond seepage and tailings spills on the local surface and ground water systems. The effects were not considered significant with respect to radioactive constituents.

Williams (1978) reviewed the literature and made recommendations on surface disposal of uranium mill tailings. In his paper, he noted that groundwater flow within fractured rock, such as that associated with mining activities, is an extremely difficult medium to thoroughly and reliably investigate and predict (Section 4.1.3). Based on this viewpoint, disposal of contaminated material underground may be very difficult to monitor, because groundwater flow and direction as well as any retardation of contaminants would be difficult or even impossible to determine.

Scott and Charlwood (1982) reported on Atomic Energy of Canada Ltd.'s work on granitic rock in the areas of geology, geomechanics, hydrogeology, geochemistry, and geophysics. The methods and equipment developed by AECL can be used to monitor the effects of placing uranium tailings underground. One possible application is the monitoring of radionuclide transport through fractures in underground rock.

Monitoring data when combined with predictive models can be valuable in forecasting when/if a problem may arise. For example, once uranium tailings have been placed underground, reliable monitoring data would indicate the rate of radionuclide transport from the underground. A model could then be used to estimate the rate at which the radionuclides might move to the biosphere. Sheppard and Mitchell (1984) reported on the Atomic Energy of Canada Ltd. computer model called SCEMR (Soil Chemical Exchange and Migration of Radionuclides). The computer model was used to determine the rate of migration of radionuclides upward in various soil types to the soil surface where they may be taken up by plant roots. The model also predicted the type of soil needed for a particular climatic area to inhibit upward migration of radionuclide contaminated groundwater. The model was originally developed for use with the underground disposal of nuclear wastes, but can have some relevant applications when determining the effects of placing uranium tailings underground. A user manual was developed by Sheppard (1981) and the program was updated to become SCEMR1 (Bera and Sheppard, 1984).

If a problem is detected through monitoring, the potential solutions are dependent on the specific nature of the problem itself. Obviously, a potential solution which requires re-entry of a filled and flooded underground working may not be viable. However, many problems which may arise from tailings placed underground could be alleviated through downhole techniques in drilled boreholes. Problems with any residual surface-impounded techniques would likely be easier to correct from the perspective of access.

Monetary bonding is becoming a common method to ensure future monitoring and corrections can be carried out. Bell (1989) described two of the factors which have enhanced the desire by regulatory agencies for bonds: (1) increased costs associated with long term maintenance of collect-and-treat facilities; and (2) social awareness, conscience and pressures. In British Columbia the government has required prospective, operating and closed mines establish a bond to the government to cover the costs for future reclamation at the site (Price and Errington, 1994). The amount of the bond is set on a site-specific basis using data collected on physical, chemical, and climatic conditions of the site and subsequent predictions of site conditions into the future.

5.3 Precedent

Bragg et al. (1982) wrote that in Canada the approach by regulatory agencies for reaching decisions regarding the supervision of the uranium milling industry was one of communication and cooperation. They felt that this approach was crucial in achieving solutions, both short- and long-term, which provided the greatest benefit to all people of Canada. They also reported that this interactive approach has benefits which far outweigh any possible liabilities at a specific site.

Similarly, Culver et al. (1982) emphasized that government regulations concerning short- and long-term handling of uranium tailings waste should be reviewed and altered on a site-specific basis. Close-out options for a particular mine may not be technically or economically feasible for other mines, and to enforce blanket close-out criteria may be inappropriate.

It is unlikely that Canada will enforce one specific set of closure criteria in light of changing knowledge and technology. For example, Bragg (1980) of the Atomic Energy Control Board presented a discussion paper on proposed interim close-out criteria for uranium tailings sites. In this paper, one of the long-term criteria was that "surface water recharge to the facility is to be limited to that from direct precipitation. Also, no permanent water pool will be allowed on the tailings area". This differs from current attitudes on the preference for water cover if it inhibits oxidation of pyritic materials or depresses radon emanation (Davé, 1992). This is an example of how ideas evolve with knowledge and, even if an idea or method is not technically or economical practical at the present time, it may be so in the future.

Through Canada's flexible approach to mine closure and reclamation, it is doubtful that underground disposal would be forced on a cooperative mining company. No case study of forced retrieval and disposal has been found during this review, although voluntary efforts on various aspects of retrieval or placement abound (Sections 3.2.1 and 4.2.2). Therefore, there is apparently no full-scale precedent for this retrieval-disposal option. However, there is one analog in Canada where fresh tailings are being directed into a mined-out pit at Rabbit Lake, Saskatchewan. This analog is described in the following paragraphs.

As a general analog to tailings placement in underground workings, uranium tailings have been placed into the depleted Rabbit Lake open-pit mine in Saskatchewan since 1984 (Cameco Corporation, 1992; PVS Technologies Ltd., 1994). By the end of 1993, the total weight of tailings placed in the pit was approximately 3,000,000 dry tonnes, filling the lower 60 meters of the pit.

The placement of tailings was conducted after careful design and modelling to create a "pervious surround". This pervious surround, or envelope, of crushed rock is draped upwards onto the pit walls as the pit is filled with tailings. A sand filter is then draped over the crushed rock to prevent tailings from migrating into the rock.

During placement of the tailings, the pervious surround allows the tailings to drain and consolidate. Drainage flows downward through the surround to the base of the pit, through a drift, to a raise where the water is then pumped to the surface. This water is then used in the mill, or treated and discharged.

After the Rabbit Lake pit is filled and flooded, the underlying concept of the pervious surround is to divert regional groundwater flow around the tailings so that no potential contaminants will be flushed out of the tailings into the surrounding environment. As a result, groundwater is expected to flow only in the crushed rock and sand filter, whereas porewater in the tailings will be stagnant. Molecular diffusion is expected to release a minor amount of contaminants into the pervious surround.

There has been detailed monitoring of the placed tailings shortly after the program was initiated. The results of the monitoring, success of the program, and the problems encountered during placement provide valuable lessons.

Tailings were initially transported to the base of the Rabbit Lake Pit by mechanical means, but problems arose over local plugging of the pervious surround and difficulty of moving vehicles over the placed tailings. After approval in 1990, tailings as approximately 30-40% solids, have been placed as a slurry through a moveable pipeline.

The tailings have been instrumented with settlement cells, pneumatic piezometers, and thermistors. Several monitoring instruments have failed, but sufficient information is available to conclude that placement, consolidation, and draining are consistent with expectations.

Rates of settlement in 1992 and 1993 were approximately 0.0009 to 0.0034 m/yr. These rates are in general agreement with modelling predictions.

Piezometric levels in deep piezometers currently lie tens of meters below the top of the tailings and showed no close relationship to the top elevation as tailings were placed over the years. Shallow piezometers show piezometric levels near or above the top of the tailings, and these levels increased as tailings were placed. Overall, the hydraulic gradient is downwards.

Partially frozen layers of tailings, formed during winter months and then buried by new tailings, were a concern from the perspective of consolidation. However, thermistors and drilling indicate these layers are slowly thawing.

Particle-size segregation has been noted in Rabbit Lake tailings during placement: grain size decreases with increasing distance from the discharge point. However, the segregation apparently does not affect rates of consolidation and draining. The percentage of silt and clay in the tailings (less than 0.044 mm) increases from approximately 30% at the discharge to 100% at a distance of 177 m. The percentage of coarse sand and gravel (greater than 0.55 mm) decreases from approximately 8% at the discharge to 0% at 140 m.

Approximately 9,000,000 tonnes of tailings can be placed in the Rabbit Lake Pit. After all tailings have been placed, up to 20 years will be required for dissipation of pore pressures and consolidation based on modelling.

Despite the lack of a major precedent for surface retrieval and underground disposal, except for the Rabbit Lake analog, this literature review documents the efforts of governments and mining companies to understand the concerns and issues that would be involved in underground disposal as a full-scale option. For example, Haw (1982) reported on the Canadian National Uranium Tailings Program (NUTP) to stimulate research into understanding mechanisms affecting uranium tailings, so long term solutions to control and reclamation of Canadian uranium tailings sites can be made. That program has been followed, with emphasis only on non-radioactive constituents, by the Reactive Acid Tailings Stabilization (RATS) Program and the current Mine Environment Neutral Drainage (MEND) Program (Bell, 1989; MEND, 1994) administered by CANMET (Energy, Mines, and Resources Canada).

Perhaps Blight (1979) summarized the overall issue best:

> "It is well to concede at this point that any mining or industrial activity will inevitably cause some environmental damage. The overall benefit to the country must be offset against this damage. It must also be recognized that whatever control measures are instituted, due regard must be paid to local conditions and current circumstances. The costs of the waste disposal operation in relation to the revenue-producing operation that must pay for it, the practicability of the environmental protection measures proposed, and the short and long-term consequences of these measures, both for the safety of the public and for their quality of life, must all receive careful and due consideration".

5.4 Permafrost

In northern areas of Canada, continuous and discontinuous permafrost conditions exist (Geocon, 1993). Some of the Canadian uranium producers, past and present, are located in these regions. Permafrost conditions can offer significant benefits for tailings such as immobilization of porewater (EBA Engineering, 1992). However, disadvantages also exist. For example, if tailings freeze prior to consolidation and then are allowed to thaw, there could be a significant release of contaminated porewater that might seep from the tailings into the environment. Also, if weak frozen soils were present beneath a surface impoundment, thawing could cause instability in the dam structure (EBA Engineering, 1992).

In an attempt to understand and predict permafrost conditions and the thawing effect on frozen tailings deposits, EBA Engineering (1992) reviewed permafrost mechanisms. Along with their report they developed a computer model to simulate the effects of freezing and thawing on impoundment structures. The report was prepared for the Atomic Energy Control Board, which also contracted EBA Engineering to conduct a review of existing geothermal models and to complete a geothermal analysis of the uranium tailings impoundment at Key Lake, Saskatchewan.

In their report, EBA (1992) stated that "in Canada, the boundary between continuous and discontinuous permafrost normally lies between the -7.5 °C and the -10 °C mean annual air temperature isotherms, but extends almost to the -5 °C isotherm to the west of James Bay, the boundary shifts northward to its typical position around the -8 °C isotherm to the east of Hudson Bay". The formation of permafrost basically requires that the annual amount of heat being absorbed at the ground surface be less than that leaving the ground surface for several years. The development or deterioration of permafrost conditions are dependent on the extent of (EBA, 1992):

❶ surface heat fluxes;

❷ initial ground temperatures;

❸ geothermal flux;

❹ soil conductive heat transfer (heat flow); and,

❺ the release of latent heat due to moisture phase change in the soil (heat flow).

Ancillary considerations include (EBA, 1992):

❶ frost heave due to the migration of moisture to the freezing surface;

❷ porewater expulsion during freezing;

❸ moisture flow through the tailings media; and,

❹ solute redistribution causing thermal property changes during freezing and thawing.

To complete a geothermal analysis at a specific tailings site, EBA (1992) listed the natural conditions that must be simulated:

❶ variation in surface heat flux/temperature;

❷ initial conditions dependent on depth;

❸ soil property variations dependent on depth and lateral variations; and,

❹ phase conditions of water in the soil and concurring physical and thermal soil property variations dependent on depth and temperature.

EBA (1992) reported on frost features found at three Canadian uranium tailings sites, Gunnar, Rabbit Lake, and Key Lake:

❶ elevated mounds, such as those at Gunnar which have been found at times to be 1 m high and several metres across in low lying areas with free standing water;

❷ segregated ice lenses such as those found in boreholes drilled at Rabbit Lake up to 17 m in depth (Section 5.3);

❸ "hummocky" surfaces due to porewater expulsion during freezing in an area of water storage on the tailings at Key Lake, reported to be 1-1.2 m high with a seasonal frost depth of approximately 600 mm; and,

❹ permafrost, which at Key Lake was found to be fully frozen up to 5.5 m thick at the east side, fully and partially frozen up to 9 m thick in the centre, and sporadic frozen zones at the west side of the tailings impoundment.

Each of these conditions can significantly affect the success with which tailings can be retrieved and placed underground (Section 3.2 and 4.2) or stabilized in surface impoundments.

Permafrost conditions can also reduce radon emanation on the surface while the freezing conditions exist. For example, Kilborn and Beak (1979) reported that 15 cm of frozen ground can reduce radon flux rate by 40%.

Udd et al. (1983) reported on laboratory studies on the physical and mechanical properties of "representative mill tailings" under frozen conditions. This work was the first step in assessing the use of frozen tailings for underground support.

At Cominco's Polaris Mine on Little Cornwallis Island in the Canadian High Arctic, underground mining takes place within permafrost (Keen, 1992). Backfilling of stopes is accomplished by two methods depending on the season, summer or winter. In the winter, backfill material, silty limestone and shale mixed with mine waste, is dumped from the surface down a raised borehole into the stope. A D-8 dozer mixes the backfill material with water to make a tight-filling backfill. In the summer, the backfill material has sufficient free water to be dumped directly into the stope. In 1991, 315,000 m^3 of quarried material and 5,000 m^3 of mine waste was used (Keen, 1992).

6. CONCLUSION

This report has been structured like a catalogue of facts and information, with each paragraph presenting some concept, concern, theory, or case study involving the retrieval of tailings from surface impoundments (Chapter 3), placement of tailings underground (Chapter 4), and related issues (Chapter 5). The structure of this catalogue is displayed in the Table of Contents (p. i). In fact, the Table of Contents can be used as a flowchart or checklist for submissions by mining companies pertaining to the retrieval and/or underground placement of tailings.

The International Atomic Energy Agency (1981) developed a more general checklist for evaluating the feasibility of any uranium-tailings related project. Several of the topics extend to underground disposal of tailings; however, issues such as land use of a filled underground mine are not particularly relevant. The IAEA checklist is:

(1) Toxic metal and chemical impacts
- Hazardous material identification
- Concentrations and inventories
- Mobilities
- Reagent attack on liners (geological and synthetic)
- Pathway analysis
- Operational considerations
- Long-term considerations

(2) Land use
- Future resource availability
- Demography and projected growth patterns
- Socioeconomics of the area
- Impoundment site proliferation
- Positive or adverse impacts from proposed use
- Limited future use
- Buffer zone provision
- Historical and archaeological areas

(3) Long-term considerations
- Stabilization and rehabilitation versus maintenance and monitoring
- Maintenance of cover

Radon exhalation
Gamma shielding
Vegetative protection and control
Burrowing animal control
Financial guarantees
Stabilization and rehabilitation
Monitoring, surveillance, and maintenance
Governmental and implementing organizations
Viability
Agency transfers
Record availability and use
File review and update
Guardianship of property
(4) Legal requirements and limitations
Areas covered by law and regulations
Areas not covered by law and regulations
(5) Aesthetics
Harmony with environment

Based on information in this report, the underground disposal of uranium tailings is not usually a reasonable alternative for many uranium mines. This stems primarily from one fact: for many mines, all tailings generated by the operation cannot be returned underground. Therefore, some tailings would remain on the surface at many minesites. Chapter 3 explains in detail the implications of disturbing surface-impounded tailings for the purpose of placing only some of the volume underground. The potential effects on the environment caused by partial retrieval should be risked only if one or more key "intangible costs" are lessened significantly. These "costs" are basically long-term concerns, such as over environmental protection and impoundment stability.

7. REFERENCES

Ackman, T.E. 1982. Sludge Disposal from Acid Mine Drainage Treatment. U.S. Bureau of Mines, Report of Investigations 8672, 25 p.

Acres International Ltd. 1992. Liquefaction of Uranium Tailings. AECB Project No. 5.130.1, Document INFO-0409, Atomic Energy Control Board, Ottawa, February 1992, 61 p.

Airey P.L. 1986. Radionuclide migration around uranium ore bodies in the Alligator Rivers Region of the Northern Territory of Australia - Analogue of radioactive waste repositories. Chemical Geology, 55, pp. 255-268.

Aljoe, W.W., and J.W. Hawkins. 1991. Hydrologic characterization and in-situ neutralization of acidic mine pools in abandoned coal mines. IN: Proceedings of the Second International Conference on the Abatement of Acidic Drainage, September 16-18, Montreal, Quebec, Volume 1, p. 69-89.

Allison, J.D., D.S. Brown, and K.J. Novo-Gradac. 1990. MINTEQA2/PRODEFA2, A Geochemical Model for Environmental Systems: Version 3.0 User's Manual. U.S. Environmental Protection Agency, Environmental Research Laboratory, Athens, Georgia, USA.

Alpers. C.N., and D.K. Nordstrom. 1991. Geochemical evolution of extremely acid mine waters at Iron Mountain, California: Are there any lower limits to pH? IN: Proceedings of the Second International Conference on the Abatement of Acidic Drainage, September 16-18, Montreal, Quebec, Volume 2, p. 321-342.

Anderson M.T. 1992. Aqueous Uranium Concentrations in the Natural Environment. AECB Project No. 5.138.1, Document INFO-0411, Atomic Energy Control Board, Ottawa, Ontario.

Anonymous. 1983. Backfilling with a Schwing. South African Mining Engineering Journal, 92, No. 8, p. 33-34.

Anonymous. 1978. Anglo American's Research Lab Reports Progress in Mining Techniques. Engineering Mining Journal, 179(9), 1978, pp. 299.

Anonymous. 1977. Electrokinetic Densification. Engineering and Mining Journal, Feb 1977, pp. 122.

Archibald, J.F., and J.H. Nantel. 1984. Prediction of Radiation Levels and Ventilation Requirements in Underground Uranium Mines. CIM Bulletin, Vol. 77, No. 861, January 1984, pp. 60-64.

Aref, K., F.P. Hassani, and D.Churcher. 1989. A study on liquefaction potential of paste backfill. IN: F.P. Hassani, ed., Proceedings of 4th International Symposium on Innovations in Mining Backfill Technology, Montreal, Quebec, October 2-5, p.405-414. Canadian Institute of Mining and Metallurgy.

Asami, T. 1988. Soil Pollution by Metals from Mining and Smelting Activities. IN: Chemistry and Biology of Solid Waste, Dredged Material and Mine Tailings, Springer-Verlag, New York, pp. 143-169.

Astle, R., and G.C. Knight. 1990. Underground Disposal of Mine Waste. IN: Proceedings of the Western Regional Symposium on Mining and Mineral Processing Wastes, Berkeley, California, May 30- June 1.

Atkins, A.S., R.N. Singh, A. Barkhordarian, and A.H. Zadeh. 1987. Pumpability of Coal Mine Tailings for Underground Disposal and for Regional Support. IN: Reclamation, Treatment and Utilization of Coal Mining Wastes, Elsevier Science Publishers, Amsterdam, pp 401-414.

Atomic Energy Control Board. 1983a. An Approach to the Calculation of Dose Commitment Arising from Different Methods for the Long-Term Management of Uranium Mill Tailings: Summary Report. Atomic Energy Control Board Report INFO-0097.

Atomic Energy Control Board. 1983b. Probabilistic Calculation of Dose Commitment from Uranium Mill Tailings. Atomic Energy Control Board Report INFO-0139.

Atwood, G. 1974. Technical and Economic Feasibility of Underground Disposal Systems. IN: Proceedings of the Symposium on Mine and Preparation Plant Refuse Disposal, Coal and the Environment Technical Conference, Louisville, Kentucky, October 22-24, p.147-160. National Coal Association.

Aydan, Ö, and T. Kawamoto. 1990. Discontinuities and Their Effect on Rock Mass. IN: Proceedings of the International Symposium on Rock Joints, Leon, Norway, June 4-6 1990, p. 149-156. A.A. Balkema, Rotterdam, 1990, ISBN 90 6191 1095.

Barbreau, A. 1989. Session II(b) - Chairman's Summary. IN: Proceedings of an NEA Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, April 26-28, p.345-348. Atomic Energy of Canada Limited.

Batsche, H., W. Rauert, and K. Klarr. 1979. Hydrogeological Research at the Site of the Asse Salt Mine. IN: Proceedings of a Symposium on the Underground Disposal of Radioactive Wastes, Vol. II, Otaniemi, Finland, July 2-6 1979, pp.345-370. International Atomic Energy Agency, Vienna, 1980.

Bean, J. J. 1972. Tailings as an Ore Body. IN: Tailings Disposal Today, Proceedings of the First International Symposium, Tucson, Arizona, October 31, November 1, 2 and 3, 1972, pp 606-614.

Bell, A.V.. 1989. The Implications of Reclamation Requirements on the Design and Operation of Tailings Impoundments. IN: Proceedings of the International Symposium on Tailings and Effluent Management, Halifax, Nova Scotia, August 20-24 1989, pp. 73-81. Pergamon Press, Toronto.

Belyaev, E.V., and N.A. Posahennikova. 1983. Stowing Mineral Wastes in a Mine Within a Framework of Low Waste Technologies in the Coal Industry [in Russian]. IN: Obogashchenie I Briketirovanie Uglya, p. 2-25. Ehkspress-Informatsiya, Vypusk.

Bearman, P.J. 1979. A review of the environmental problems associated with the disposal of uranium tailings. Minerals and the Environment, July, p. 64-79.

Bera, K.E., and M.I. Sheppard. 1984. SCEMR1 - Simulation of Soil Chemical Exchange and Migration of Radionuclides on a Geologic Timescale - A User's Manual. Atomic Energy of Canada Ltd., Technical Record TR-249, Pinawa, Manitoba, 30 p.

Bird, G.W., and P.C. Fung. 1979. Surface Chemistry of Feldspars. Atomic Energy of Canada Ltd., Technical Record TR-76, Pinawa, Manitoba, October 1979, 22 p.

Birgersson, L., and I. Neretnieks. 1990. Diffusion in the matrix of granitic rock: Field test in the Stripa Mine. Water Resources Research, Vol. 26, No. 11, November 1990, pp. 2833-2842.

Blanchard, R.L., T.W. Fowler, T.R. Horton, and J.M. Smith. 1981. Potential Health and Environmental Hazards of Wastes at Active Surface and Underground Uranium Mines. IN: International Conference Radiation Hazards in Mining: Control, Measurements, and Medical Aspects, October 4-9, 1981, Colorado School of Mines, Golden, Colorado, p. 687-696. Society of Mining Engineers of American Institute of Mining, Metallurgical, and Petroleum Engineers, Inc., New York.

Blaylock, B.G., and C.S. Fore. 1979. Radioactive Wastes. Journal of the Water Pollution Control Federation, Vol. 51, No. 6, pp. 1410-1417.

Blight, G.E. 1979. The Disposal of Mining and Industrial Waste - Viewpoint. Die Sivielle Ingenieur in Suid-Afrika, June, 1979, p. 133.

Bloomfield, R.A. 1984. Current Research into the Potential for Utilization of Mine Waste. Journal of Testing and Evaluation, JTEVA, Vol. 12, No.2, pp. 119-122.

Bower, D.R., C.C. Davison, and E.T. Kozak. 1986. Tidal Observations in Piezometers at the Site of Canada's Underground Research Laboratory. Atomic Energy of Canada Ltd., Technical Record TR-393, Pinawa, Manitoba, November 1986, 34 p.

Boyd, J.M., T.G. Carter, R.A. Knapp, and K.B. Culver. 1982. Hydrogeological Investigations and Evaluation of the Stanleigh Mine Tailings Impoundment Site. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 141-156. International Atomic Energy Agency, Vienna, 1982.

Bragg, K. 1980. Long Term Aspects of Uranium Tailings Management (An AECB Discussion Paper on Proposed Interim Close-out Criteria. IN: First International Conference on Uranium Mine Waste Disposal, May 19-21 1980, Vancouver, BC, pp.599-605. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers, Inc., New York, 1980.

Bragg, K., C. Potter, A. James. 1982. Recent Developments in the Regulation and Management of Canadian Uranium Tailings. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 85-101. International Atomic Energy Agency, Vienna, 1982.

Branch, S. 1983. PCA Solves Tailings Problem with Underground Disposal. Canadian Mining Journal, October, 1983, pp. 7-9.

Brewitz, W. 1986. Feasibility of Disused Underground Mines for Low Level and Intermediate Level Radioactive Waste Disposal. IN: Conference on the Treatment and Containment of Radioactive Waste and Its Disposal in Arid Environments, Cape Town, South Africa, September 7-12, 1986, pp. 163-181.

Brewitz, W., and U. Löschhorn. 1980. Geoscientific Investigations in the Abandoned Iron Ore Mine Konrad for Safe Disposal of Certain Radioactive Waste Categories. IN: Proceedings Series Underground Disposal of Radioactive Wastes, Proceedings of a Symposium on the Underground Disposal of Radioactive Wastes, Otaniemi, Finland, 2-6 July 1979, Vol. II, pp. 89-102. International Atomic Energy Agency, Vienna, 1980.

British Columbia AMD Task Force / Steffen Robertson Kirsten (B.C.) Inc. / Norecol Environmental Consultants / Gormely Process Engineering. 1989. Draft Acid Rock Drainage Technical Guide, Volume 1. B.C. AMD Task Force Publication.

Broman, P.G. 1989. Reclamation Plans for Two Tailings Ponds in Sweden - Examples of Site Specific Solutions. IN: Proceedings of the International Symposium on Tailings and Effluent Management, Halifax, August 20-24, 1989, 28th Annual Conference of Metallurgists of CIM, Toronto, 1989, pp. 103-118.

Brookins, D.G. 1983. Geochemical Aspects of Radioactive Waste Disposal. Springer-Verlag, New York.

Burgess, A.S. 1979. Pluton Hydrogeology. Atomic Energy of Canada Ltd., Technical Record TR-73, Pinawa, Manitoba, November 1979, 72 p.

Busse, E. 1974. Acid Drainage Control and Water Treatment at Heath Steele. Environmental Control, CIM Bulletin, November, 1974, pp. 54-61.

Cameco Corporation. 1992. Environmental Impact Statement: Collins Bay A-Zone D-Zone and Eagle Point Development (Revised - 1992). Prepared for the Federal Environmental Assessment Review Panel.

CANMET and Kilborn. 1981. Removal, dewatering and disposal of $Ba/RaSO_4$ precipitates from uranium tailings water treatment ponds. CANMET, Department of Energy, Mines and Resources Canada.

Carlsson H. 1981. The Stripa Project: An International Project in Radioactive Waste Management. Underground Space, Vol. 6, July/August 1981, pp. 43-45. Pergamon Press.

Chakravatti, J.L., E. LaRocque, D.W. Reades, and E.I. Robinsky. 1982. Thickened Tailings Experiment for Close-Out of Uranium Mill Tailings at Denison Mines Limited. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 417-428. International Atomic Energy Agency, Vienna, 1982.

Chapman, N., and I. McKinley. 1990. Radioactive waste: Back to the future?. New Scientist, May 5, 1990, pp. 54-58.

Chilton, B.D., and H.A. Pfuderer. 1989. Overview of cleanup and treatment of radioactively contaminated sites. Nuclear Safety, Volume 30, No. 4, October-December 1989, pp. 519-533.

Clausen, G., and J.F. Archibald. 1983. Uranium mill tailings as backfill in underground mines. Mining Science and Technology, 1, p. 69-75.

Constable, T.W., and W.J. Snodgrass. 1987. Leachability of Radioactive Constituents From Uranium Mine Tailings - Final Report. Environment Canada, Environmental Protection, Report EPS 3/MM/2.

Cramer, J.J.. 1986. Sandstone-hosted uranium deposits in northern Saskatchewan as natural analogs to nuclear fuel waste disposal vaults. Chemical Geology, 55, pp. 269-279.

Culver, K.B., J.L. Chakravatti, D.M. Gorber, R.A. Knapp, and J.B. Davis. 1982. Close-Out Concepts for the Elliot Lake Uranium Mining Operations. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 157-168. International Atomic Energy Agency, Vienna, 1982.

Davé, N.K. 1992. Water Cover on Acid Generating Mine/Mill Wastes; A Technical Review. CANMET Report for Project 30.14.99.

DLugosz, J.J., and M.S. Bedinger. 1989. The Rationale for Geologic Disposal of High-Level Radioactive Wastes in the United States. IN: Proceedings of the Thirteenth Annual British Columbia Mine Reclamation Symposium, Water Management at Minesites, Vernon BC, June 7-9 1989, pp. 205-216. Technical and Research Committee on Reclamation.

Down, C.G., and J. Stocks. 1976. The Environmental Problem of Tailings Disposal at Metal Mines. Department of the Environment, United Kingdom, 1976, 33 p.

Down, C.G., and J. Stocks. 1977. Environmental Problems of Tailings Disposal. Mining Magazine, July 1977, pp 25-33.

Dowse, L.H., and K.H. Selby. 1975. Groundwater Pollution Control in an Industrialized Part of the Trent Basin. Water Pollution Control, Vol. 74, pp. 526-543.

Dreesen, D.R., E.J. Cokal, L.E. Wangen, J.M. Williams, P.D. O'Brien, and E.F. Thode. 1982. Uranium Mill Tailings Conditioning Technology. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 311-324. International Atomic Energy Agency, Vienna, 1982.

Drew, D.J., D.M. Grondin, and T.T. Vandergraaf. 1990. The Large Block Radionuclide Migration Facility. Atomic Energy of Canada Ltd., Technical Record TR-519, Pinawa, Manitoba, November 1990, 27 p.

Dubrovsky, N.M., K.A. Morin, J.A. Cherry, and D.J.A. Smyth. 1984. Uranium tailings acidification and subsurface contaminant migration in a sand aquifer. Water Pollution Research Journal of Canada, 19, p.55-89.

EBA Engineering Consultants Ltd. 1992. Frost Evolution in Tailings Final Report. AECB Project No. 5.135.1, INFO-0383, Atomic Energy Control Board, Ottawa, Ontario.

Evans, D.M. 1970. Pollution Goes Underground. IN: Air and Water Pollution, Proceedings of the Summer Workshop, August 3-15 1970, University of Colorado, Colorado, pp. 89-111. Associated University Press, Boulder, Colorado, 1970.

Everitt, R.A., D. Woodcock, P. Gann, and D. Robinson D. 1990. Geology of the 420 Level of the Underground Research Laboratory. Atomic Energy of Canada Ltd., Technical Record TR-529, Pinawa, Manitoba, December 1990, 13 p.

Everitt, R.A., P. Chernis, D. Good, and A. Grogan. 1989. Mapping of the excavation damage zone around the circular access shaft at Atomic Energy of Canada Limited's Underground Research Laboratory. IN: Proceedings of an NEA Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, April 26-28, p.271-282. Atomic Energy of Canada Limited.

Falkie, T.V., J.E. Gilley, and A.S. Allen. 1974. Overview of Underground Refuse Disposal. IN: First Symposium on Mine Preparation Plant Refuse Disposal, Coal Environmental Technology Conference, Louisville, Kentucky, October, p. 128-144.

Fleer, V.N., and R.M. Johnston. 1986. A Compilation of Solubility and Dissolution Kinetics Data on Minerals in Granitic and Gabbroic Systems. Atomic Energy of Canada Ltd., Technical Record TR-328-2, Pinawa, Manitoba, February 1986, 170 p.

Fleer, V.N., and R.M. Johnston. 1985. The Dissolution Behaviour of Minerals in Granitic and Gabbroic Systems. Atomic Energy of Canada Ltd., Technical Record TR-328-1, Pinawa, Manitoba, November 1985, 45 p.

Ford, Bacon and Davis Utah Inc. 1977. [Several reports including:]
- Engineering Assessment of Inactive Uranium Mill Tailings, Maybell Site, Maybell, Colorado.
- Engineering Assessment of Inactive Uranium Mill Tailings, Rifle Site, Rifle, Colorado.
- Engineering Assessment of Inactive Uranium Mill Tailings, Tuba City Site, Tuba City, Arizona.
- Engineering Assessment of Inactive Uranium Mill Tailings, Lakeview Site, Lakeview, Oregon.
- Summary of the Phase II, Title I Engineering Assessment of Inactive Uranium Mill

Tailings, Spook Site, Converse County, Wyoming.
- *Summary of the Phase II, Title I Engineering Assessment of Inactive Uranium Mill* Tailings, Lowman Site, Lowman, Idaho.
- Phase II, Title I Engineering Assessment of Inactive Uranium Mill Tailings, Phillips/United Nuclear Site, Ambrosia Lake, New Mexico.
- Engineering Assessment of Inactive Uranium Mill Tailings, Shiprock Site, Shiprock, New Mexico.

Franklin, J.C., K.D. Weverstad, J.W. Black, and J.E. Cleveland. 1982a. Radiation Hazards in Backfilling with Classified Uranium Mill Tailings. IN: Proceedings of the Fifth Annual Uranium Seminar, Albuquerque, New Mexico, September 20-23, p. 171-173. Society of Mining Engineers, New York.

Franklin, J.C., R.A. Washington, J.C. Kerkering, H. Montone, and R. Regan. 1982b. Radon Emanation from Stopes Backfilled with Cemented Uranium Mill Tailings. U.S. Bureau of Mines Report of Investigations RI 8664, 8 p.

Freeze, R.A., and J.A. Cherry. 1979. Groundwater. Prentice-Hall of Canada, Ltd., Toronto.

Fry, R.M. 1982. Criteria for the Long-Term Management of Uranium Mill Tailings. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 71-83. International Atomic Energy Agency, Vienna, 1982.

Fuhrmann, M., and P. Colombo. 1990. Leachate Geochemistry at Low-Level Radioactive Repository Sites - Effects of Site Geohydrology on Nature and Production of Leachates. Proceedings of the Safe Disposal of Radionuclides in Low-Level Radioactive-Waste Repository Sites: Low-Level Radioactive-Waste Disposal Workshop, U.S. Geological Survey, July 11-16, 1987, Big Bear Lake, California, pp 44-48. U.S. Geological Survey Circular 1036.

Gaffney, D.V. 1983. Underground Coal Waste Disposal: Can It Be Cost Effective? Coal Mining & Processing, January 1983, pp. 56-59.

Garisto F., and N.C. Garisto. 1985. A UO_2 Solubility Function for the Assessment of Used Nuclear Fuel Disposal. Nuclear Science and Engineering, 90, pp. 103-110.

Geocon (SNC-Lavalin Environment). 1993. Permafrost in Tailings. Report for the Canadian Mine Environment Neutral Drainage (MEND) Program.

Geologic Testing Consultants. 1986. Compilation of Subsurface Hydrogeologic Data. Atomic Energy Control Board, Report INFO-0203 (Volumes 1-3).

Goedde, E. 1980. Hydraulic Handling in Mining - Fully Automatic Large-Scale Test Unit on a Practical Scale. IN: New Zealand Conference of the Australasian Institute of Mining and Metallurgy 1980, pp. 105-110. The Australasian Institute of Mining and Metallurgy, Victoria, Australia.

Golder Associates Ltd. 1991a. Review of Long Term Stability of Flooded Workings and Evaluation of Areas of Potential Surface Subsidence at Quirke I, Quirke II and Panel Mines, Elliot Lake, Ontario - Report to Rio Algom Ltd. Contract No. 543-A-16, Mississauga, Ontario.

Golder Associates Ltd. 1991b. Hydrogeological Assessment of the Effects of Quirke/Panel Mine Flooding on Regional and Local Groundwater Flow Systems, Elliot Lake, Ontario. Contract No. 543-A-16, Mississauga, Ontario.

Golder Associates Ltd. with Senes Consultants Limited and Cumming Cockburn Limited. 1992. Evaluation of Closure Options Decommissioning Study Denison Mine Tailings Management Area TMA-1 and TMA-2 [sic]. Mississauga, Ontario.

Goode, J.R. 1993. The relocation of tailings. IN: CIM Metsoc Meeting Symposium, Advances in Environmental Protection for Metallurgical Industries, Quebec City, August 30 - September 1.

Grela, C., and J. Kutyla. 1978. Management of Non-Coal Mineral Waste Materials Produced in Coal Mines: Methods of Utilization and Costs. Wiad. Gorn., 29, No. 1, p. 11-16.

Grondin, D.M., and D.J. Drew. 1988. Radionuclide Migration Along Machined and Natural Fractures in Granite; A Laboratory Study. Atomic Energy of Canada Ltd., Technical Record TR-449, Pinawa, Manitoba, 1988, 22 p.

Guvanasen, V. 1984. Hydrological Modelling at the Underground Research Laboratory. IN: Proceedings of the 16th Information Meeting of the Nuclear Fuel Waste Management Program, Atomic Energy of Canada Ltd., Technical Record TR-218, Pinawa, Manitoba, pp. 98-114.

Guvanasen, V., J.A.K. Reid, and B.W. Nakka. 1985. Predictions of Hydrogeological Perturbations Due to the Construction of the Underground Research Laboratory. Atomic Energy of Canada Ltd., Technical Record TR-344, Pinawa, Manitoba, June 1985, 175 p.

Hajas, W., and W.F. Heinrich. 1987. INTRACOIN Level 1 Benchmark Calculations Using SYVAC2 Geosphere Response Functions. Atomic Energy of Canada Ltd., Technical Record TR-435, Pinawa, Manitoba, 1987, 15 p.

Halbert, B.E., J.M. Scharer, J.L. Chakravatti, and E. Barnes. 1982. Modelling of the Underwater Disposal of Uranium Mine Tailings in Elliot Lake. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 127-140. International Atomic Energy Agency, Vienna.

Hamel, P., and J. Howieson. 1982. A Summary of the Canadian Uranium Mill Tailings Situation. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 21-39. International Atomic Energy Agency, Vienna.

Hamel, P.E., W.D. Smythe, R.M. Duncan, and J.R. Coady. 1982. Overview of the AECB's Strategy for Regulating Radioactive Waste Management Activities. Atomic Energy Control Board, Report INFO-0078.

Hans, J.M. Jr., G.E. Eadie, and M.F. O'Connell. 1981. Environmental Condition and Impact of Inactive Uranium Mines. IN: International Conference on Radiation Hazards in Mining: Control, Measurement, and Medical Aspects, October 4-9, 1981, Golden, Colorado, pp. 163-175. Society of Mining Engineers of American Institute of Mining, Metallurgical, and Petroleum Engineers Inc. New York.

Hart, K.P., D.M. Levins, and A.G. Fane. 1986. Steady-state Rn^* diffusion through tailings and multiple layers of covering materials. Health Physics, Vol. 50, No. 3, pp. 369-379.

Hartley, J.N., G.W. Gee, H.D. Freeman, J.F. Cline, P.A. Beedlow, J.L. Buelt, J.R. Relyea, and T. Tamura. 1982. Uranium Mill Tailings Remedial Action Project (UMTRAP) - Cover and Liner Technology Development Project. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 429-448. International Atomic Energy Agency, Vienna.

Hassani, F.P., editor. 1989. Proceedings of 4th International Symposium on Innovations in Mining Backfill Technology, Montreal, Quebec, October 2-5. Canadian Institute of Mining and Metallurgy.

Haw, V.A. 1982. The Canadian Research Programme into the Long-Term Management of Uranium Mine Tailings. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 663-677. International Atomic Energy Agency, Vienna.

Heinrich, W.F. 1984. Geosphere Submodel for the Second Interim Assessment of the Canadian Concept for Nuclear Fuel Waste Disposal - Post-Closure Phase. Atomic Energy of Canada Ltd., Technical Record TR-286, Pinawa, Manitoba, November 1984, 61 p.

Heinrich, W.F., and J.R. Walker. 1987. MCROC (Microcracking in Rock) Computer Code: Description and User Guide. Atomic Energy of Canada Ltd., Technical Record TR-420, Pinawa, Manitoba, 1987, 16 p.

Heremans, R.H. 1989. Session IV - Chairman's Summary. IN: Proceedings of an NEA Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, April 26-28, p.531-532. Atomic Energy of Canada Limited.

Herrmann, A.G., H.J. Brumsack, and H. Heinrichs. 1985. Necessity, possibilities and limiting factors for underground disposal of toxic substances of anthropogenic origin. Naturwissenschaften, 72, No. 8, p. 408-418.

Hladka, E., I. Burian, and S. Svoboda. 1991. Impact of uranium tailings on the environment and population [in Czech]. Ceskoslovenka Hygiena, 36, No. 5-5, p. 226-232.

Intera Technologies Ltd. 1992. An Evaluation of Contaminant Retardation Mechanisms. AECB Project No.5.124.1 Atomic Energy Control Board. INFO-0406. Ottawa, Canada. 96 p.

Intera Technologies Ltd. 1985. Hydrogeological Modelling of the Underground Research Laboratory Site: Characterization and Prediction. Atomic Energy of Canada Ltd., Technical Record TR-345, Pinawa, Manitoba, September 1985, 80 p.

International Atomic Energy Agency. 1981. Current Practices and Options for Confinement of Uranium Mill Tailings. International Atomic Energy Agency, Technical Reports Series No. 209, Vienna, 1981.

Jakubick, A.T., R. Klein, M.N. Gray, L.D. Keil. 1989. Characteristics of the excavation response zone as applied to shaft sealing. IN: Proceedings of an NEA Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, April 26-28, p.157-174. Atomic Energy of Canada Limited.

Jessop, A.M., P.B. Robertson, and T.J. Lewis. 1979. A Brief Summary of the Thermal Conductivity of Crystalline Rocks. Atomic Energy of Canada Ltd., Technical Report TR-12, Pinawa, Manitoba, January 1979, 19 p.

Johnson, J.R. 1979. Reclamation of Abandoned Mine Site. Journal of the Environmental Engineering Division, pp. 597-603.

Johnston, L.M. 1982. A Review of the Geochemistry of Groundwater in Crystalline Rocks. Atomic Energy of Canada Ltd., Technical Record TR-154, Pinawa, Manitoba, September

1982, 99 p.

Kamineni, D.C., P.A. Brown, D. Stone. 1980. Fracture-Filling in the Atikokan Area, Northwestern Ontario. Atomic Energy of Canada Ltd., Technical Record TR-109, Pinawa, Manitoba, May 1980, 16 p.

Katsube, T.J., T.W. Melnyk, and J.P. Hume. 1986. Pore Structure from Diffusion in Granitic Rocks. Atomic Energy of Canada Ltd., Technical Record TR-381, Pinawa, Manitoba, April 1986, 28 p.

Kaufmann, D., J.C. Emery, and J.P.H. Lin. 1987. Radon emissions during mill tailings backfill operations in a uranium mine. Environmental Geology and Water Science, 10, No. 3, p. 129-133.

Keen, A.J.. 1992. Polaris Update. CIM Bulletin, Underground Mining, Volume 85, No. 961, June, 1992, pp. 51-57.

Kegel, W.G. 1975. Problems with Underground Refuse Disposal. Transactions, Vol. 258, Society of Mining Engineers, December 1975, pp. 250-283.

Kennedy, R.H., L.J. Deal, F.F. Haywood, and W.A. Goldsmith. 1977. Management and Control of Radioactive Waste from Uranium Milling Operations in the United States of America. IN: Nuclear Power and Its Fuel Cycle, Proceedings of an International Conference on Nuclear Power and Its Fuel Cycle, International Atomic Energy Agency in Salzburg, May 2-13 , 1977, Vol. 4, pp. 545-560. International Atomic Energy Agency, Vienna.

Kepford, C.R. 1977. Radioactive Waste: The Gift That Never Stops Giving. IN: Workshop on Policy and Technical Issues Pertinent to the Development of Environmental Protection Criteria for Radioactive Wastes, Albuquerque, New Mexico, April 12-14, p. 1.27-1.45. U.S. Environmental Protection Agency.

Ketron Inc. 1982. Returning Coal Waste Underground; Technical Progress Report, October 1, 1981 - December 31, 1981. Department of Energy, Report DOE/FE/14185-T3.

Khan, A.H., and M. Raghavayya. 1989. Radon problem in uranium industry. IN: Proceedings of the International Symposium on Uranium Technology, Bombay, India, December 13-15, Volume 2, p. 833-847.

Kilborn Ltd. and Beak Consultants Ltd. 1979. An Assessment of the Long Term Suitability of Present and Proposed Methods for the Management of Uranium Tailings. Atomic Energy Control Board, INFO-0024, Ottawa, Ontario.

King, R.M., and H.L. MacPhie. 1991. Hydrology of Quirke and Panel Uranium Tailings Basins. AECB Project No. 5.136.1, INFO-0392, Atomic Energy Control Board of Canada.

Kühn, K. 1983. Pilot Research Projects for Underground Disposal of Radioactive Wastes in the Federal Republic of Germany. International Atomic Energy Agency, IAEA-CN-43/169, pp.351-369.

Kurzydlo H. 1991. Mining the Legnica Brown Coal Deposits. Mining Magazine. pp. 215-221.

Lakshmanan, V.I. 1985. Long Term Monitoring of High Grade Uranium Mill Tailings: Final Report. CANMET Contract Report.

Lang, P.A. 1989. Room 209 excavation response test in the underground research laboratory. IN: Proceedings of an NEA Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, April 26-28, p.295-329. Atomic Energy of Canada Limited.

Lang, P.A. 1986. Overview of the Fracture Testing Program at the Underground Research Laboratory. Atomic Energy of Canada Ltd., Technical Record TR-408, Pinawa, Manitoba, December 1986, 17 p.

Langer, M., A. Pahl, and I.M. Wallner. 1988. Excavation Responses in Underground Repositories in Salt Formations. IN: Proceedings of an NIE Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, Manitoba, April 26-28, 1988. OECD Nuclear Energy Agency & Atomic Energy of Canada Ltd.

Larson, W.C. 1980. Uranium In Situ Leach Mining - Environmental Controls. IN: First International Conference on Uranium Mine Waste Disposal, May 19-21, 1980, Vancouver BC, Canada, pp. 175-192. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers Inc., New York.

Laul, J.C., C.W. Thomas, M.R. Peterson, and R.W. Perkins. 1988. Radionuclide Disequilibria Studies for Investigating the Integrity of Potential Nuclear Waste Disposal Sites: Subseabed Studies. IN: Radioactive Waste Management and the Nuclear Fuel Cycle, Vol. 11(1), pp. 89-108. Harwood Academic Publishers GmbH, United States.

Lee, Y.H., and R.B. Stuebing. 1990. Heavy metal contamination in the river toad, *Bufo juxtasper* (Inger), near a copper mine in east Malaysia. Bulletin of Environmental Contamination and Toxicology, 45, pp. 272-279.

Litjer, Z. 1985. Application of pneumatic conveying systems in the mining industry. Chamber of Mines Journal, 27, No. 6, p.39.

Lloyd, P.J.D. 1980. Ninety Years Experience in the Preservation of Uranium Ore Dumps. IN: First International Conference on Uranium Mine Waste Disposal, May 19-21 1980, Vancouver, British Columbia, pp. 33-42. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers Inc., New York.

Lush, D.L., W.J. Snodgrass, P. McKee. 1982. Aquatic Pathway Variables Affecting the Estimation of Dose Commitment from Uranium Tailings. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14 1982, pp. 483-503. International Atomic Energy Agency, Vienna.

Lyon, R.B., and E.L.J. Rosinger. 1980. Safety Assessment for Deep Underground Disposal Vault - Pathways Analysis. IN: Proceedings of a Symposium on the Underground Disposal of Radioactive Wastes, Vol. II, Otaniemi, Finland, July 2-6, 1979, pp. 453-463. International Atomic Energy Agency, Vienna.

MacDonald, R.J.C., P.D. Kondos, S. Crevier, P. Rubinsky, M. Wasselauf. 1989. Generation of, and Disposal Options for, Canadian Mineral Industry Effluent Treatment Sludges. IN: Proceedings of the International Symposium on Tailings and Effluent Management, Halifax, August 20-24, 1989, 28th Annual Conference of Metallurgists of CIM, Toronto, 1989, pp. 139-157.

MacGregor, R.A. 1966. Recovery of U_3O_8 by Underground Leaching. The Canadian Mining and Metallurgical Journal, Bulletin for May 1966, Montreal, pp.583-587.

MacLaren Plansearch Inc. 1987. Optimization in the Decommissioning of Uranium Tailings. Atomic Energy Control Board Research Report INFO-0321.

Marcus, D., and D.A. Sangrey. 1982. Uranium Mill Tailings Stabilization with Additives. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 449-467. International Atomic Energy Agency, Vienna.

Marine, I.W. 1980. Studies of the Subsurface Effects of Earthquakes. IN: Workshop on Subsurface Seismic Studies, Seattle, Washington, February 20. Department of Energy Report CONF-800245-1.

Markose, P.M., S. Venkataraman, K.P. Eapen, G.K. Srivastava, and M. Raghavayya. Environmental Surveillance Around the Uranium Complex at Jaduguda. 1982. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 541-551. International Atomic Energy Agency, Vienna.

Matalucci, R.V. 1989. Session III - Chairman's Summary. IN: Proceedings of an NEA Workshop on Excavation Response in Geological Repositories for Radioactive Waste, Winnipeg, April 26-28, p.457-462. Atomic Energy of Canada Limited.

Mathers, W.G., and R.L. Hembroff. 1986. Analysis of Flow Through Fractured Rock by the Hele-Shaw Method. Atomic Energy of Canada Ltd., Technical Record TR-392, Pinawa, Manitoba, October 1986, 14 p.

McCready, R.G.L. 1986. Underground Bioleaching: Extracting from Low-Grade Ore. GEOS, 3, 1986, pp. 9-10.

McFarlin, R.F., G.M. Lloyd, H. El-Shall. 1989. Tailings Management in the Florida Phosphate Industry. IN: Tailings and Effluent Management, Proceedings of the International Symposium on Tailings and Effluent Management, Halifax, August 20-24, 1989, 28th Annual Conference of Metallurgists of CIM, pp. 33-41.

McInnis, P. 1985. Literature Review for the National Uranium Tailings Program (NUTP) - Long-term Effects of Natural Events on Uranium Tailings Impoundments. The National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, Energy, Mines and Resources Canada, Ottawa, Ontario, September 1985.

McKeon, T.J., and R.W. Nelson. 1984. Evaluation Methods for the Consequences of Below Water Table Mine Disposal of Uranium Mill Tailings. Batelle Pacific Northwest Labs Report PNL-4904. Nuclear Regulatory Commission, Washington, DC.

McPherson, M.J. 1984. Role of Mine Ventilation in Site Selection for a Nuclear Waste Repository. IN: Third International Mine Ventilation Congress, Harrogate, pp. 309-314. The Institution of Mining and Metallurgy, London, England.

MEND (Mine Environment Neutral Drainage Program). (1994) 1993 Annual Report. CANMET, Government of Canada, Ottawa.

Mohiuddin, G. 1985. Radioactive Waste Containment - A Literature Study [in German]. Bundesministerium des Innern (Germany), Report BMI-1985-088; GRS-A-1096.

Morin, K.A. 1994. Prediction of water chemistry in open pits during operation and after closure. IN: Proceedings of the 18th Annual British Columbia Mine Reclamation Symposium, April 11-14, Vernon, British Columbia, p.72-86.

Morin, K.A., and N.M. Hutt. 1995. MINEWALL Version 2.0: Literature Review, Field Studies, and User's Manual. MEND Project.

Morin, K.A., J.A. Cherry, N.K. Dave, T.P. Lim, and A.J. Vivyurka. 1988a. Migration of acidic groundwater seepage from uranium-tailings impoundments, 1. Case study and conceptual model. Journal of Contaminant Hydrology, 2.

Morin, K.A., J.A. Cherry, N.K. Dave, T.P. Lim, and A.J. Vivyurka. 1988b. Migration of acidic groundwater seepage from uranium-tailings impoundments, 2 Geochemical behavior of radionuclides in groundwater. Journal of Contaminant Hydrology, 2, pp.304-321.

Nantel, J., and N. Lecuyer. 1983. Assessment of slag backfill properties for the Noranda Chadbourne Project. CIM Bulletin, 76, No. 849, p. 57-60.

National Academy of Science. 1975. Underground Disposal of Coal Mine Wastes. National Academy of Science Report, 172 p.

National Research Council. 1975. Returning mine wastes underground. Summary article of "Underground Disposal of Coal Mine Wastes", Ground Water, 13(3), 1975, pp. 313.

Neller, R.R., J.D. Sandy, and V.H.R Oliver. 1973. How Mufulira Has Been Rehabilitated. World Mining, United States Edition, September, 1973, pp. 30-37.

Nelson, J.D., S.R. Abt, R.L Volpe, D. van Zyl, N.E. Hinkle, and W.P. Staub. 1986. Methodologies for Evaluating Long-Term Stabilization Designs of Uranium Tailings Impoundments. U.S. Nuclear Regulatory Commission, NUREG/CR-4620, ORNL/TM-10067.

Nelson, J.D., and D.B. McWhorter. 1980. Influence of Impoundment and Substratum Configuration on Seepage from Impoundments. IN: First Annual Conference on Uranium Mine Waste Disposal, May 19-21 1980, Vancouver British Columbia, pp. 193-203. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers Inc, New York.

Nordstrom, D.K., T. Olsson, L. Carlsson, and P. Fritz. 1989. Introduction to the hydrogeochemical investigations within the international Stripa Project. Geochimica et Cosmochimica Acta, 53, pp.1717-1726.

Northwest Geochem. 1992. Physical and Chemical Hydrogeology of the Lynx Area of the Westmin Myra Falls Operations [Westmin Resources Limited]. Mine Report submitted to the British Columbia Ministry of Energy, Mines, and Petroleum Resources.

Osborne R.V. 1982. Optimizing Radiation Protection in the Management of Uranium Tailings. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 471-481. International Atomic Energy Agency, Vienna.

Osiensky, J.L., G.V. Winter, and R.E. Williams. 1984. Monitoring and mathematical modeling of contaminated ground-water plumes in fluvial environments. Ground water, 22, No. 3, May-June, 1984, pp. 298-306.

Patton, F.E. 1952. Backfilling at Noranda. Canadian Mining and Metallurgical Bulletin, April, p. 191-197.

Pearson, F.J. Jr. 1987. Geochemical and Isotopic Evidence on the Origin of Brines in the Canadian Shield. Atomic Energy of Canada Ltd., Technical Record TR-429, Pinawa, Manitoba, 1987, 94 p.

Peterson, S.R., R.L. Erikson, G.W. Gee G.W. 1982. The Long Term Stability of Earthen Materials in Contact With Acidic Tailings Solutions. U.S. Nuclear Regulatory Commission, Washington, November, 1982.

Petterson, G.W.E. 1975. Computer control applications at the Strathcona Concentrator. ISA Transactions, 14, No. 1, pp. 78-88.

Pickens, J.F., and G.E. Grisak. 1979. Finite-Element Analysis of Liquid Flow, Heat Transport and Solute Transport in a Groundwater Flow System: Governing Equations and Model Formulation. Atomic Energy of Canada Ltd., Technical Record TR-81, Pinawa, Manitoba, September 1979, 30 p.

Pidgeon, R.T. 1982. Review of the Non-Radiological Contaminants in the Long-Term Management of Uranium Mine and Mill Wastes. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 263-284. International Atomic Energy Agency, Vienna.

Popovich, J.M., and R.F.J. Adam. 1985. Hydraulic Transportation of Coal Mine Refuse. IN: Proceedings from the Tenth International Conference on Slurry Technology, March 26-28, Lake Tahoe, Nevada, pp.327-332. Slurry Technology Association, Washington, DC.

Popovich, J.M., R.F.J. Adam, A.J. Miscoe, and G.R. Desko. 1984. Returning Coal Waste Underground. IN: Proceedings of the International Congress on Technology and Technology Exchange: Technology and the World Around Us, October 8-10, p. 140-141.

Pratt, H.R., W.A. Hustrulid, and D.E. Stephenson. 1978. Earthquake Damage to Underground Facilities. Department of Energy Contract EY-76-C-09-0001.

Price, W.A., and J.C. Errington. 1994. ARD Policy for mine sites in British Columbia. IN: Proceedings of the Third International Conference on the Abatement of Acidic Drainage, April 24-29, Pittsburgh, USA, Volume 4, p.285-293.

PVS Technologies Ltd. 1994. Tailings Disposal in Rabbit Lake Pit: 1993 Annual Report. Prepared for Cameco Corporation. Report No. 94-01, dated 15 February 1994.

Raicevic D. 1980. Removal of Radionuclides from Uranium Ores and Tailings to Yield Environmentally Acceptable Waste. IN: First International Conference on Uranium Mine Waste Disposal, May 19-21, 1980, Vancouver, British Columbia, pps. 351-360. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers Inc., New York.

Ramsey, R.W. Jr. 1982. Tailings Technology. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 367-371. International Atomic Energy Agency, Vienna.

Raven, K.G. 1986. Hydraulic Characterization of a Small Ground-water Flow System in Fractured Monzonitic Gneiss. Canadian National Hydrology Research Institute Paper No. 30. 133 p.

Raven, K.G., and J.E. Gale. 1986. A Study of the Surface and Subsurface Structural and Groundwater Conditions at Selected Underground Mines and Excavations. Atomic Energy of Canada Ltd., Technical Record TR-177, Pinawa, Manitoba, 81 p.

Rio Algom Limited. 1993. Closure Plan: East Kemptville Mine.

Rohlfing, I. 1983. Slinger Belt Stowing Technique for Cemented Backfill at the Meggen Mine. IN: Proceedings of the International Symposium on Mining with Backfill, Lulea, June 7-9, p. 189-198.

Roberts, A. 1981. Applied Geotechnology, A Text for Students and Engineers on Rock Excavation and Related Topics. Pergamon Press, New York, 1981.

Roberts, J.M., and A.L. Masullo Jr. 1985. Pneumatics for Mining Applications. IN: Proceedings of the 1985 Symposium on Surface Mining, Hydrology, Sedimentology, and Reclamation, Lexington, Kentucky, December 9-13, p. 83-91.

Rogers, V.C. 1978a. The Environmental Impact of Uranium Mill Tailings Generated Between 1975 and 2030. IN: Transactions of the American Nuclear Society, 1978 Winter Meeting, November 12-16, 1978, Washington, DC, pp 91-92. American Nuclear Society, Inc. Illinois.

Rogers, V.C. 1978b. Radiation Pathways and Potential Health Impacts from Inactive Uranium Mill Tailings. IN: Transactions of the American Nuclear Society, 1978 Winter Meeting, November 12-16, 1978, Washington, DC, pp 92. American Nuclear Society, Inc. Illinois.

Rogers, V.C., K.K. Nielson, D.C. Rich, M.W. Grant, M.L. Mauch, G.M. Sandquist, and G.B. Merrell. 1982. Uranium Mill Tailings Containment Systems Performance and Cost. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 197-212. International Atomic Energy Agency, Vienna.

Rubin, L.S., M. Burnett, A. Amundson, G.J. Colaizzi, and R.H. Whaite. 1981. Disposal of Coal Mine Waste in Active Underground Coal Mines. IN: U.S. Bureau of Mines Technology Transfer Conference, Denver, Colorado, July 16, p.8-13.

Sassos, M.P. 1986. Backfilling stopes at BM&S, new semi-automatic system combines in-pit crushing with underground conveying to supply cemented rockfill to mined stopes. Engineering and Mining Journal, 187, No. 6, pp 58-61.

Scales, M. 1982. High Arctic Wizardry: Polaris Mine On Stream. Canadian Mining Journal, July, p. 24-41.

Scarano, R.A. 1980. Review of Uranium Mill Tailings Management Programs Involving Below Grade Disposal. IN: First International Conference on Uranium Mine Waste Disposal, May 19-21, 1980, Vancouver, BC, pp. 121-143. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers Inc., New York.

Schneider, H.J. 1988. Geotechnical requirements of subsurface repositories in mines, rock and salt caverns for interim or final disposal of hazardous wastes. Bulletin of the International Association of Engineering Geology, 37, pp. 71-76.

Scott, J.S., and R.G. Charlwood. 1982. Canadian Geoscience Research and Design Concepts for Disposal of High-Level Waste in Igneous Rocks. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp.413-441. International Atomic Energy Agency, Vienna.

Scott, R.B., R.C. Wilmoth, and R.D. Hill. 1972. Cost of reclamation and mine drainage abatement, Elkins Demonstration Project. Society of Mining Engineers AIME, Transactions, Volume 252, June 1972, pp. 187-193.

SECOR Inc. 1981. Cost of Implementing AECB Interim Criteria for the Closeout of Uranium Tailings Sites. Atomic Energy Control Board, INFO-0023-3, Ottawa, Ontario, April 30 1981, 27 p.

Senes Consultants Ltd. 1991a. Environmental Screening Panel Mine Underground Workings Decommissioning Proposal. Richmond Hill, Ontario, January, 1991.

Senes Consultants Ltd. 1991b. Review of Mine Backfilling and Quirke Lake Disposal as Tailings Reclamation Alternatives for the Quirke and Panel Waste Management Areas. Richmond Hill, Ontario, April, 1991.

Senes Consultants Ltd. 1986. Approaches to Risk Assessment for Canadian Uranium Mill Tailings. National Uranium Tailings Program. DSS File No. 14SQ.23241-5-1717, Contract No. OSQ85-00225.

Sethness, E.D. Jr., and C.W. Holmes. 1979. Low-Level Radioactive Waste Disposal Techniques. IN: Corpus Christi Surface Leach Mine In-situ Pollution Seminar, South Texas Uranium Seminar, Corpus Christi, TX, Sept 10-13, 1978, pp 69-73. Society of Mining Engineers of American Institute of Mining, Metallurgical, and Petroleum Engineers, Inc., New York.

Sheppard, M.I. 1981. SCEMR: A Model for Soil Chemical Exchange and Migration of Radionuclides in Unsaturated Soil. Atomic Energy of Canada Ltd., Technical Record TR-175, Pinawa, Manitoba, November 1981, 51 p.

Sheppard, M.I., and J.H. Mitchell. 1984. SCEMR: Generic Application Using Precambrian Shield Parameter Values and a Proposed SCEMR/SYVAC Interface. Atomic Energy of Canada Ltd., Technical Record TR-240, Pinawa, Manitoba, January 1984, 22 p.

Sims, W.N. 1972. Remining of Tailings by Hydraulicking and Other Methods. IN: Tailings Disposal Today, Proceedings of the First International Tailing Symposium, Tuscon, Arizona, October 31 - November 3, 1972, pp. 615-633.

SNC-Shawinigan Inc., Geocon, and Vinod Garga. 1995. Workshop on Current Developments for Assessment of Long-Term Stability of Uranium Mill Tailings Retention Structures: Generic Elliot Lake Conditions. Atomic Energy Control Board publication.

Snodgrass, W.J., and D.R. Hart. 1990. Remeasurement of Th-230 in the Porewater of Lacnor Tailings. Atomic Energy Control Board, Government of Canada, Ottawa, Ontario, Project No. 4066.2, February 1990.

Snodgrass, W.J., D.L. Lush, and J. Capobianco. 1982. Implications of Alternative Geochemical Controls on the Temporal Behaviour of Elliot Lake Tailings. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 285-308. International Atomic Energy Agency, Vienna.

Snodgrass, W.J., R.V. Nicholson, and N. Garisto. 1985. Descriptive Algorithm Development for Physical, Chemical and Biological Processes Operational in Uranium Mill Tailings. The National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, September, 1985.

Sprute, R.H., and D.J. Kelsh. 1976. Using slimes for backfill in deep mines. Mining Congress Journal, April, p.22-26.

St-Arnaud, L., E.K. Yanful, R. Prairie, and N.K. Dave. 1989. Evolution of Acidic Pore Water at the Waite Amulet Tailings Site, Noranda, Quebec. IN: Proceedings of the International Symposium on Tailings and Effluent Management, Halifax, August 20-24, 1989, 28th Annual Conference of Metallurgists of CIM, pp. 93-102.

Steffen, Robertson and Kristen Inc. (SRK). 1986a. Characteristics Relevant to Releases of Three Generic Canadian Uranium Tailings Sites - Technical Memorandum II. The National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, September, 1985.

Steffen, Robertson and Kristen Inc. 1986b. The Stability of Control Structures for Uranium Mill Tailings. National Uranium Tailings Program Report 58902, Energy, Mines and Resources, Ottawa, March 1986.

Steffen, Robertson, and Kirsten (SRK). 1987. Canadian Uranium Mill Waste Disposal Technology. The National Uranium Tailings Program, Canada Centre for Mineral and Energy Technology, 1987.

Stephens, M.E., B.W. Goddwin, and T.H. Andres. 1989. Guidelines for Defining Probability Density Functions for SYVAC3-CC3 Parameters. Atomic Energy of Canada Ltd., Technical Record TR-479, Pinawa, Manitoba, February 1989, 45 p.

Striegl, R.G. 1990. Radioactive Gases at Low-Level Radioactive-Waste Repository Sites. IN: Proceedings from the Safe Disposal of Radionuclides in Low-Level Radioactive-Waste Repository Sites: Low-Level Radioactive-Waste Disposal Workshop, U.S. Geological Survey, July 11-16, 1987, Big Bear Lake, California, pp.40-44. U.S. Geological Survey Circular 1036.

Tabori, J.E., and S.R. Wilkinson. 1984. WASTE: A Computer Program for Three-Dimensional Groundwater Flow Systems. Atomic Energy of Canada Ltd., Technical Record TR-255, Chalk River, Ontario, January 1984, 33 p.

Tallin, J.E., D.E. Pufahl, and S.L. Barbour. 1990. Waste management schemes of potash mines in Saskatchewan. Canadian Journal of Civil Engineering, 17, pp 528-542.

Tang, D.H., E.O. Frind, and E.A. Sudicky. 1980. Contaminant Transport in Fractured Porous Media: An Analytical Solution for a Single Fracture. Atomic Energy of Canada Ltd., November 1980, 37 p.

Task Committee on Low-Level Radioactive Waste Management of the Technical Committee on Nuclear Effects. 1986. Management of inactive uranium mill tailings. Journal of Environmental Engineering, 112, No. 3, Paper No. 20665, pp.490-537.

Taylor, M.J. 1980. Radionuclide Movement in Seepage and Its Control. IN: First International Conference on Uranium Mine Waste Disposal, May 19-21, 1980, Vancouver, BC, pp. 205-244. Society of Mining Engineers of the American Institute of Mining, Metallurgical, and Petroleum Engineers Inc., New York.

Thomson, B.M. and R.J. Heggen. 1982. Water Quality and Hydrologic Impacts of Disposal of Uranium Mill Tailings By Backfilling. IN: Proceedings Series Management of Wastes from Uranium Mining and Milling, Albuquerque, New Mexico, May 10-14, 1982, pp. 373-384. International Atomic Energy Agency, Vienna.

Thomson, B.M., P.A. Longmire, and D.G. Brookins. 1986. Geochemical constraints on underground disposal of uranium mill tailings. Applied Geochemistry, 1, pp. 335-343.

Thomson, M.J. 1989. Slurry disposal from Kellingley colliery. Colliery Guardian, February, p.34-36.

Ticknor, K.V., T.T Vandergraaf, and D.C. Kamineni. 1985. Radionuclide Sorption on Mineral and Rock Thin Sections Part 1: Sorption on Selected Minerals. Atomic Energy of Canada Ltd., Technical Record TR-365, Pinawa, Manitoba, November 1985, 107 p.

Toro T. 1991. How to close a uranium mine. New Scientist, 22 June, 1991, pp. 42-45.

Udd, J.E., J.F. Archibald, and J.H. Nantel. 1983. Concept for Using Frozen Fill for Underground Support. IN: J.E. Udd, ed., Proceedings of the Underground Support Systems Symposium, September 19, Sudbury, Ontario.

Vilks, P., J.J. Cramer, D.B. Bachinski, D.C. Doern, and H.G. Miller. 1993. Studies of colloids and suspended particles, Cigar Lake uranium deposit, Saskatchewan, Canada. Applied Geochemistry, 8, p. 605-616.

Voisey, R., and R.J. Spencer. 1990. From nickel to gold. CIM Bulletin, Mining Processing, 83, No. 943, 1990, pp. 90-96.

Walsh, M.L. 1986. Doses Resulting From Intrusion into Uranium Tailings. Atomic Energy Control Board, Government of Canada, Ottawa, Ontario, Report No. INFO-0182.

Walton, F.B. 1982. Design of Laboratory-Scale Experiments to Measure Radionuclide-Geologic Material Interactions: Part 1. Hydraulic Considerations. Atomic Energy of Canada Ltd., Technical Record TR-194-1, Pinawa, Manitoba, May 1982, 40 p.

Weber M.F. 1990. Hydrogeologic Information Needs for Low-Level Radioactive-Waste Disposal Sites. IN: Safe Disposal of Radionuclides in Low-Level Radioactive-Waste Repository Sites: Low-Level Radioactive-Waste Disposal Workshop, U.S. Geological Survey, July 11-16, 1987, Big Bear Lake, California, pp.80-84. U.S. Geological Survey Circular 1036.

Wei, L., and J.A. Hudson. 1990. Permeability Variation Around Underground Openings in Jointed Rock Masses: A Numerical Study. IN: Proceedings of the International Symposium on Rock Joints, Leon, Norway, June 4-6, 1990, pp.565-569. A.A. Balkema, Rotterdam, ISBN 90 6191 1095.

Weimer, W.C., R.R. Kinnison, and J.H. Reeves. 1981. Survey of Radionuclide Distributions Resulting from the Church Rock, New Mexico, Uranium Mill Tailings Pond Dam Failure. Pacific Northwest Laboratory for the U.S. Nuclear Regulatory Commission, December 1981, NUREG/CR-2449 PNL-4122, Washington, D.C.

Weitzman, M. 1985. Modelling of the Groundwater Flow System Around the URL Excavation Before and After Its Construction. Atomic Energy of Canada Ltd., Technical Record TR-233, Pinawa, Manitoba, June 1985, 87 p.

Werniuk, J. 1990. Canadian Gold Mines - Mainstay of the Placer Dome Empire. Canadian Mining Journal, June, 1990, pp. 17-20.

West, J.M., D.G. Haigh, P.J. Hooker, and E.J. Rowe. 1991. Microbial Influence on the Sorption of ^{137}Cs onto Materials Relevant to the Geological Disposal of Radioactive Waste. Reviews, Experientia 47 (1991), Birkhäuser Verlag. CH-4010 Basel/Switzerland, 1991, pp. 549-552.

Wheeland, K., and S. Payant. 1991. Criteria for Underground Disposal of AMD-Generating Materials. IN: Proceedings of the Second International Conference on the Abatement of Acidic Drainage, September 16-18, Montreal, Quebec, Volume 1, p. 577-579.

Whiteway, P. 1991. Full Circle. Canadian Mining Journal, November-December 1991, pp. 25-30.

Whiteway, P. 1994. Getting back on track. Canadian Mining Journal, 115, no. 5 (October), p.31-32.

Wilkins, B.J.S., A.R. Reich, and W.R. Wallace. 1984. Slow Microcracking in Plutonic Rocks. Atomic Energy of Canada Ltd., Technical Record TR-264, Pinawa, Manitoba, April 1984, 29 p.

Wilkins, B.J.S., and G.L. Rigby. 1991. MCDIRC Update 1. Atomic Energy of Canada Ltd., Technical Record TR-546, Pinawa, Manitoba, June 1991, 5 p.

Williams, R.E. 1978. Control and Prevention of Seepage from Uranium Mill Waste Disposal Facilities. IN: Management Stabilisation and Environmental Impact of Uranium Mill Tailings Proceedings of NEA Conference, Albuquerque, United States, July 1978, pp.289-297. Nuclear Energy Agency, Environmental Protection Agency.

Williams, R.E. 1972. Modernization of Coeur d'Alene Tailings Disposal Practices. IN: Tailings Disposal Today, Proceedings from the First International Symposium, October 31 - November 3, 1972, Tucson, Arizona, pp. 634-671. Miller Freeman Publications Inc., San Francisco, California.

Witherspoon, P.A., N.G. Cook, and J.E. Gale. 1981. Geologic storage of radioactive waste: Field studies in Sweden. Science, 211, February 27 1981, pp. 894-900.

Wollenberg, H.A., and S. Flexser. 1985. The distribution of uranium and thorium in the Stripa Quartz Monzonite, Sweden. Uranium, 2, pp. 155-167.

Wood, P.A. 1983. Underground Stowing of Mine Waste. IEA Coal Research, Report No. ICTIS/TR23, London, England, April 1983, 67 p.

Wuschke, D.M., K.K. Mehta, K.W. Dormuth, T. Andres, G.R. Sherman, E.L.G. Rosinger, B.W. Goodwin, J.A.K. Reid, and R.B. Lyon. 1981. Environmental and Safety Assessment Studies for Nuclear Fuel Waste Management, Volume 3: Post-Closure Assessment. Atomic Energy of Canada Ltd., Technical Report TR-127-3, Pinawa, Manitoba, September 1981, 109 p.

Yamaguchi, U., and J. Yamatoni. 1983. A Consideration on the Effect of Backfill for Ground Stability. IN: Proceedings of the International Symposium on Mining with Backfill, Luleå, 7-9 June 1983, pp. 443-451. A.A. Balkema, Rotterdam, 1983.

Zimmerman, R.M., R.A. Bellman, K.L. Mann, D.P. Zerga, and M. Fowler. 1988. G-Tunnel Welded Tuff Mining Experiments Evaluations. Sandia National Laboratories (USA) Report SNL-87-1433. U.S. Department of Energy.

APPENDIX A

Results of Computer-Based Literature Searches

KEYWORDS
(with wildcards for approximate matching)

TAILINGS or URANIUM TAILINGS or MINE WASTE

and

UNDERGROUND or REMOVAL or TRANSFER or BACKFILL or IMPACTS

SEARCHED DATABASES
(N = number of matches)

DIALOG

- ASFA (N=0)
- Biosis (N=7)
- Chemical Abstracts (N=39)
- Chemical Engineering (N=2)
- El Compendex Plus (N=152)
- Engineered Materials Abstracts (N=0)
- Enviroline (N=61)
- Environmental Bibliography (N=0)
- Geoarchive (N=33)
- GPO Monthly Catalog (N=33)
- Life Sciences Collection (N=2)
- Metadex (N=11)
- NTIS (N=481)
- Oceanic Abstracts (N=0)
- Pascal (N=13)
- Pollution Abstracts (N=0)
- Scisearch (N=10)
- Water Resources Abstracts (N=67)
- Waternet (N=1)
- Aluminum Industry Abstracts (N=0)
- World Translation Index (N=0)
- Zoological Record (N=0)

CAN/OLE

- COAL (N=47)
- MINTEC (N=384)
- MINPROC

CD-ROM

- POL/TOX (N=31)
- Aqualine (N=59)
- Waterlit (N=41)
- Selected Water Resources Abstracts (N=41)
- INIS (at AECB; N=784)
- AECB Library and Publications (N=~100)

Other

- University of British Columbia Library
- BVAEP (Environment Canada Library in Vancouver)
- Atomic Energy of Canada Limited Publications

APPENDIX B

Results of Questionnaire Survey

QUESTIONS INCLUDED IN THE QUESTIONNAIRE

#1: Do you know of any persons/organizations/references which deal with the implications of disturbing or retrieving surface-impounded tailings? If so, please list them in detail below.

#2: What do you believe are the major issues in disturbing or retrieving tailings from surface impoundments?

#3: Do you know of any persons/organizations/references which deal with the implications of placing tailings into underground workings? If so, please list them in detail below.

#4: What do you believe are the major issues in placing tailings into underground workings?

RESPONDENTS

More than 100 hundred questionnaires were mailed internationally. Forty-five people responded. We thank the following people who took the time to respond.

QUESTIONNAIRE RESPONDENTS

Rao Avadhanula
Atomic Energy Control Board
P.O. Box 1046, Station B
270 Albert Street
Ottawa, Ontario
Canada K1P 5S9

Louis Bienvenue
Quebec Ministry of Energy & Resources
Mineral Research Centre
2700 Einstein St.
St. Foy, Quebec
Canada G1P 3W8

Marcia Blanchette
Coordinator
MEND - Prediction & Monitoring
Canada Centre for Mineral and Energy Technology
Mineral Sciences Laboratories
555 Booth Street
Ottawa, Ontario
Canada K1A 0G1

Dave Blowes
Waterloo Centre for Ground water Research
Waterloo, Ontario
Canada N2L 3G1

Dave Corkill
Atomic Energy Control Board
P.O. Box 1046, Station B
270 Albert Street
Ottawa, Ontario
Canada K1P 5S9

Vic Enns
Environment Canada
Environmental Protection
Box 6010
100 Hamilton Blvd.
Whitehorse, Yukon
Canada Y1A 5L7

Grant Feasby
Manager MEND Secretariat
Canada Centre for Mineral and Energy Technology
Mineral Sciences Laboratories
555 Booth Street
Ottawa, Ontario
Canada K1A 0G1

Keith Ferguson
Senior Environmental Engineer
Placer Dome Inc.
P.O. Box 49330
Bentall Postal Station
1600 - 1055 Dunsmuir St.
Vancouver, BC
Canada V7X 1P1

Mike Filion
Teck Corporation
Suite 600
200 Burrard St
Vancouver, BC
Canada V6E 3L9

Peter Flavelle
Atomic Energy Control Board
P.O. Box 1046, Station B
270 Albert Street
Ottawa, Ontario
Canada K1P 5S9

Wayne Fraser
Rattan Mine
Hudson Bay Mining & Smelting
P.O. Box 1500
Flin Flon, Manitoba
Canada R8A 1N9

Ross Gallinger
Environmental and Quality Standards Engineer
Northwood Timber
P.O. Box 9000
Prince George, BC
Canada V2L 4W2

Jeff Harris
Harris Exploration Services
534 Ellis St.
North Vancouver, BC
Canada V7H 2G6

Igor Holubec
Geocon/SNC-Lavalin Environmental Inc.
3210 American Drive
Mississauga, Ontario
Canada L4V 1B3

George Hope
Manager, Mineral Processing
LAC Minerals Ltd.
200 Bay St
Suite 2100, North Tower
Royal Bank Plaza
P.O. Box 156
Toronto, Ontario
Canada M5J 2J4

Dr. John Errington
Resource Management Branch
Ministry of Energy, Mines and Petroleum Resources
Mineral Division
Province of British Columbia
Room 105
525 Superior Street
Victoria, BC
Canada V8V 1X4

Margaret Kalin
Boojum Research Ltd.
468 Queen St. East
Suite 400
Box 19
Toronto, Ontario
Canada M5A 1T7

Randy Knapp
SENES Consultants
52 West Beaver Creek Rd.
Unit #4
Richmond Hill, Ontario
Canada L4B 1L9

Garth Lee
Boss Mining Division
Maclaren Forest Products
Noranda Minerals
Box 1134
100 Mile House, BC
Canada V0K 2E0

Dave Lee
AECL Research
Chalk River Laboratories
Chalk River, Ontario
Canada K0J 1J0

Hugh Liebscher
Regional Hydrogeologist
Environmental Canada
224 West Esplanade
North Vancouver, BC
Canada V7M 3H7

Tjoe Lim
CANMET - Elliot Lake Laboratories
P.O. Box 100
Elliot Lake, Ontario
Canada P5A 2J6

Les MacPhie
Vice President
Geocon/SNC-Lavalin Environmental Inc.
2 Place Felix-Martin
Montreal, Quebec
Canada H2Z 1Z3

Ross McArthur
Noranda Minerals Inc.
Bell Mine
P.O. Box 2000
Granisle, BC
Canada V0J 1W0

Ken Munro
Superintendent, Environmental Protection
Highland Vally Copper
P.O. Box 1500
Logan Lake, BC
Canada V0K 1W0

Dr. Nand Dave
CANMET - Elliot Lake Laboratories
P.O. Box 100
Elliot Lake, Ontario
Canada P5A 2J6

Ron Nicholson
Waterloo Centre for Ground water Research
Waterloo, Ontario
Canada N2L 3G1

Bob Patterson
Gibraltar Mines Ltd.
Box 130
McLeese Lake, BC
Canada V0L 1P0

Angela Putz
Canada Centre for Mineral and Energy Technology
552 Booth Street
Ottawa, Ontario
Canada K1A 0G1

Kevin Scissons
Project Officer
Decommissioning and Mine Impacts Section
Wastes and Impacts Division
Atomic Energy Control Board
P.O. Box 1046, Station B
270 Albert Street
Ottawa, Ontario
Canada K1P 5S9

Len Sinclair
Saskatchewan Environment
12th Flr.
McIntosh Mall
800 Central Avenue
Box 3003
Prince Albert, Saskatchewan
Canada S6V 6G1

Adrian Smith
2555 Edgemont Dr.
North Vancouver, BC
Canada V7R 2M9

Luc St-Arnaud
Nornada Technology Centre
Environmental Technology Department
240 Hymus Blvd
Pointe Claire, Quebec
Canada H9R 1G5

Bill Stewart
Spokane Research Center
US Bureau of Mines
E. 315 Montgomery
Spokane, Washington
USA 99207

Glen Strachan
Consulting Geologist
Engineering Geosciences Inc.
13419 NE 137th Place
Kirkland, Washington
USA 98034

Peter Townsend
Dennison Mines Limited
P.O. Box 2600
Elliot Lake, Ontario
Canada P5A 2K2

Gilles Tremblay
Coordinator - Prevention & Control
MEND
Canada Centre for Mineral and Energy Technology
Mineral Sciences Laboratories
555 Booth Street
Ottawa, Ontario
Canada K1A 0G1

Rudy van Dyk
Westmin Resources Ltd.
Myra Falls Operation
P.O. Box 8000
Campbell River, BC
Canada V9W 5E2

Don Welsh
Golder Associates
2180 Meadowvalle
Mississauga, Ontario
Canada L5N 5S3

Ken Whelland
Noranda Technology Centre
Environmental Technology Department
240 Hymus Blvd
Pointe Claire, Quebec
Canada H9R 1G5

Bernie Zgola
Atomic Energy Control Board
P.O. Box 1046, Station B
270 Albert Street
Ottawa, Ontario
Canada K1P 5S9