ASSESSMENT OF THE RADIOLOGICAL IMPACT OF NON-URANIUM METAL MINING

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by

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Consultant Services, Inc.

for

Environment Canada
and
Atomic Energy Control Board

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ASSESSMENT OF THE RADIOLOGICAL IMPACT OF NON-URANIUM METAL MINING

PHASE 1 - SELECTION OF THE STUDY AREA AND PRELIMINARY CHARACTERIZATION OF SOURCE TERMS

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The successful completion of phase 1 of this study was made possible only through the cooperation and assistance of a number of participating mining companies, government agencies and individuals.

Acknowledgement must be accorded to the mining companies in the Timmins and Kirkland Lake regions who permitted the consultants to perform measurements on their property. The assistance of Ms. M. deLaplante of Kidd Creek mines and of Mr. Don Bruce of Wilroy mines in setting up detectors and taking samples is greatly appreciated.

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The cooperation of the Ontario Ministry of Environment, Ontario Ministry of Natural Resources and Ontario Ministry of Agriculture is acknowledged.

1.0 SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

1.1 Selection of the Potentially Significant Region

The overall objective of the study is to assess the radiological impact of technically enhanced natural radioactivity from non-uranium metal mining in a major mining camp in northern Ontario. The project was divided into three phases:

Phase I: Selection of the study region and the preliminary characterization of radiological source terms within this region.

Phase II: Detailed measurements of the source terms within the selected region.

Phase III: Assessment of the radiological impact due to the mining operations within the selected region.

This report presents the results of phase I. The primary purpose of this phase is to provide data on which to base the evaluation of the radiological significance of non-uranium metal mining in Canada.

The phase I plan was to review the relevant characteristics of the principal mining regions and select an appropriate region in which to measure the source terms. The magnitudes of the terms within this region were then evaluated. This region of interest called the Potentially Significant Region possesses attributes which tend to make the radiological impact in this region more significant than in the others. In order to select such a region, information on radiological conditions and radioactivity levels in effluents from non-uranium mining sites outside of Canada was reviewed to gain some insight into the nature and extent of the potential problem. A review was then made of the geology and mineralogy, radioactivity measurements, mining activities, demography, land uses and environmental aspects of six identified prominent mining areas in northern Ontario. These areas are:

- Timmins
- Kirkland Lake/Larder Lake
- Sudbury
- Manitouwadge
- Red Lake
- Cobalt

Using a systematic decision making process, the Kirkland Lake region was finally selected as the Potentially Significant Region. This choice was based on the following considerations:

1) Radioactivity measurements in the area, specifically radon daughter measurements in the mines, indicated the presence of radioactivity.
2) The geology and mineralogy of the area is consistent with the trace concentrations of uranium and thorium being in the range of 1-10 ppm.
3) The Kirkland Lake region has a relatively large population of more than 13 000 people. A potentially significant population exposure is possible within this region as, in many other locations within this region, tailings areas are situated very close to population centers.
4) The district in which Kirkland Lake camp is situated produces more grain and beef products than any other region in northern Ontario.

It was also decided to perform measurements at selected sites in the Timmins camp. Analysis of the mining activities and associated geology indicated that minimal concentrations of uranium and thorium would be expected in the gold mines while higher levels are probable at the Kidd Creek copper-zinc mine. As the Timmins area is geographically close to the Potentially Significant Region, Kirkland Lake, it was cost effective to also perform measurements in this region in order to test the selection analysis.

1.2 Mining, Milling and Waste Management Operations in Non-Uranium Mining

The sources of airborne emissions from gold and other non-uranium metal underground mines are very similar to those from underground uranium mines. Both non-uranium and uranium mills employ complex physical and chemical processes to separate the ore from the waste products, and many units are common to both types of concentrators.

However, with respect to the release of radionuclides to the environment, some differences can be identified between the non-uranium and uranium mills and mines:

(a) In non-uranium milling processes, uranium, not being the end product, is discharged along with the other radionuclides to the tailings areas, while in the uranium mill considerable attention is given to the minimization of uranium loss.
(b) In uranium milling, a major fraction of the airborne emissions of uranium is from the drying and packaging stage; this is not the case for non-uranium concentrator.

(c) As with uranium mines, mill wastes from non-uranium operations are discharged to the tailings areas. These areas are either natural small bodies of water or man-made tailings areas. At non-uranium mill sites, the tailings liquid effluent is not treated with barium chloride since, very often, the presence of radionuclides is neither recognized nor of concern in most base metal operations. However, some operators add lime to the mill circuit or to the tailings pond effluent to maintain pH in the neutral range to help counteract the well documented problem of acid mine drainage.

(d) In non-uranium mining, mine water is often released directly into the environment. The water is not treated with barium chloride but in some cases lime is added at the discharge point.

The factors influencing radionuclides kinetics in tailings areas, based primarily on experience with uranium milling, were reviewed and the behaviour of uranium, thorium and radium were described taking into account the site specific factors in the Kirkland Lake and Timmins mining areas.

1.3 Measurement of Source Terms

The sampling-measurement program was designed to determine the radioactivity concentration in major airborne and liquid effluent pathways and in selected waste products. Most samples were analyzed for radium-226, thorium-230, thorium-228, and thorium-232. This was done after it was found that the radiochemical analytical procedure for radium-228 did not have sufficient sensitivity. Gamma spectrometry identified the presence of radium-228 in two Kirkland Lake samples. A summary of the radionuclide concentrations in solid waste and water samples is presented in Table 1.1. Liquid effluent source terms were calculated based on the concentration of radionuclides and the effluent flow.

The radon emissions from the mine ventilation exhausts were measured using commercial track etch detectors. Radon fluxes from the tailings areas were estimated in two ways. Initially, radon emissions were calculated using the empirical Schaiger relationship, which was modified for site specific conditions. Subsequently, a more direct measurement technique using integrating radon detectors was employed to obtain an independent estimate for the radon flux. The radon emissions, as measured directly, were a factor of 5 less than predicted by the Schaiger relationship for the Kirkland Lake abandoned tailings areas and a factor of 50 less for the Kidd Creek site. Particulate

TABLE 1.1 RADIUM-226 AND THORIUM-230 CONCENTRATIONS MEASURED IN SOLID WASTE AND LIQUID WASTE SAMPLES FROM NON-URANIUM MINES

Sample	Radium-226			Thorium-230		
	Minimum	Maximum	*Average	Minimum	Maximum	*Average
Solid Tailings (Bq/kg)	15	1670	85 ± 56	250	9880	440 ± 15
Water (mB_q/L)	4	220	85 ± 48	260	440	340 ± 7

* The error shown is 1 standard deviation

radioactivity emissions from the tailings areas were calculated from fugitive dust emission estimates using the Cowherd and Travis models.

These models differ in their predictions by almost an order of magnitude and neither is very successful in handling the very fine tailings material found at operating sites. The particulate emissions from the mine ventilation exhausts were obtained from the ventilation exhaust flow and the radioactivity content of the host rock. The source terms are summarized and presented in Table 1.2.

1.4 Evaluation of Source Terms

The source terms can be evaluated by:

(1) comparison to radiological conditions undisturbed by a facility or to other sources of radioactivity;

(2) comparing the releases to accepted regulations, standards or guidelines; and

(3) the concept of ALARA (As low as reasonably achievable).

The annual loading of radioactivity to the environment and the concentration of radionuclides in site effluents appears to be greater than would be found if the non-uranium mining facilities had not been established and hence indicate that low levels of technologically enhanced natural radioactivity are present due to non-uranium metal mining in northern Ontario. However, confirmation of this finding would require base line data or an extensive environmental monitoring program.

Relevant Ontario and Canadian regulations and guidelines were compiled. It was found that the source terms are less than the limits specified for radioactivity levels in the effluents.

TABLE 1.2 SOURCE TERMS FOR AIRBORNE AND AQUATIC RELEASES FROM NON-URANIUM MINES

Source	Rn-222 (TBq/a)	Ra-226 (MBq/a)	Th-230 (MBq/a)
Airborne Releases			
Mine Ventilation Exhausts (*)	0.3	0.4	0.3
Tailings Areas: Kirkland Lake Abandoned (+)	11.8	25.9	55.5
Tailings areas Operational (*)	0.5	3.7	25.9
Aquatic Releases			
Mine Water (*)		100	590
Tailings Decant (*)		120	1260

(*) Average Releases from Wilroy, Kidd Creek, and Kerr Addison sites.

(+) Collective Emissions from Kirkland Lake Abandoned Tailings Areas.

Source terms for uranium mine/mill sites were summarized from publically available information. With the exception of the Beaverlodge site and the Elliot Lake area mines, the data are sparse and are only available for radium-226. As expected the airborne radiological emissions from non-uranium mines are considerably less than those from uranium mining. The releases to the aquatic environment from the Kirkland Lake area mines are generally a small fraction of those from uranium mining facilities. The loading of the aquatic environment from liquid effluents from the Kidd Creek site can be shown to be within a factor of 10 when compared to individual facilities in the Elliot Lake area. There are significant difficulties when comparing the radiological consequences of uranium and non-uranium facilities, as the release of radioactivity from a facility is dependent on many factors such as the size of a facility, the design of the tailings area, etc. In this study, it has been observed that the releases to the aquatic environment from well-managed uranium facilities are not grossly dissimilar from the non-uranium mine releases. A possible interpretation of this finding is that this demonstrates the effectiveness of the waste management treatments in the uranium mining industry.

1.5 Conclusions

(1) The concentrations of radioactivity in non-uranium mine effluents appear higher than those in bodies of water in areas having known elevated levels of uranium and thorium. This supports the inference that non-uranium mining produces low levels of technologically enhanced natural radioactivity.

(2) Radioactive releases from non-uranium mining are generally a small fraction of the limits specified in Ontario and Canadian regulations and standards.

(3) Radionuclide releases from the non-uranium mining facilities in Kirkland Lake to the aquatic environment are only a small percentage of the Elliot Lake uranium facilities total loading. However, the releases from individual uranium mining facilities in Elliot Lake and uranium facilities elsewhere in Ontario and Canada are typically within an order of magnitude of those from the Kidd Creek sites.

(4) In view of the low levels of radioactivity released from non-uranium mining, there does not appear to be a justification to alter current waste management practices in order to satisfy the requirement of ALARA.

(5) The following are findings which are considered important as they can indicate improvements of methodologies which can be used in other studies:

   (a) Although there are recognized limitations, the analysis of the rock types and mineralogy of a mining area can be interpreted to predict elevated levels of radioactivity.

   (b) Direct measurement of radon emissions from tailings areas in the region of the study demonstrated that the Schaiger relationship, even when modified for site specific conditions, is very conservative. Measurement of radon emissions from other tailings areas would be required before any general statement could be made.

   (c) The gamma spectroscopy method identified the presence of radium-228 in samples from the Wilroy mine at Kirkland Lake where conventional radiochemical techniques were unsuccessful. The presence of this nuclide in samples from other areas such as Elliot Lake might be identified by this technique.

1.6 Recommendations

(1) It is recommended that phases 2 and 3 of the study, as originally formulated, need not be carried out as many of the objectives of the study have already been achieved.

However, it is recommended that screening surveys be made of other areas in Canada where potential problems of high levels of radioactivity can be identified either because of high radiological measurements or from an analysis of the geology of the area.

(2) Agencies working on the concept of de minimis levels should be aware of and take into account the concentrations of radionuclides in the wastes from non-uranium mining.

(3) The use of the geochemical modelling as developed by Langmuir should be pursued and site specific analyses should be undertaken to interpret the behaviour of uranium, thorium and radium in the tailings ponds and in the environment.

(4) Further development of the gamma spectrometry method to quantitatively measure the concentration of radium-228 in solid and liquid samples should be performed. By combining this technique with a simple radiochemical procedure it appears most probable that the technique could be made both sensitive and reliable.

(5) The use of passive integrating radon detectors for measuring radon emissions from tailings areas is very promising and the optimal procedure should be developed. This could result in more realistic estimations of the radiological impact from uranium tailings areas.

(6) It is recommended that an empirical study be performed to measure the fugitive dust emissions from different types of tailings areas. Based on these results the current models would be optimized to enable realistic predictions or new models developed.

2.0 INTRODUCTION

2.1 Project Background and Objectives

Although the extent of technologically enhanced natural radioactivity from non-uranium mining has not been extensively researched, there are indications that, given the appropriate conditions, a potential problem could exist. For example, the exposure to radon daughters of workers in non-uranium underground mines has been identified as a potential hazard over the last two decades. Significant concentrations of radon daughters have been measured in mines in Canada, the United States and elsewhere. In addition, data are available from two studies in the United States. These investigations were commissioned by the U.S. Environmental Protection Agency to evaluate the radon emissions to air (Silhanek, 1981) and the concentrations of particulate radioactivity in liquid effluents (PEDCo, 1981). These studies demonstrate that elevated levels of radioactivity in non-uranium mining effluents have been observed. Further details regarding these and other studies are provided in Appendix A.

It was not possible to identify reports on the measurement and assessment of radioactivity in the effluents from non-uranium mine/mill sites in Canada. The primary objective of this study is to evaluate the radiological impact of non-uranium metal mining, thereby filling the information gap. The study would also serve to indicate whether waste management and effluent control practices are adequate, particularly for the releases to the aquatic environment, where no specific procedures are in place to minimize radioactivity content. The study would also provide a basis of comparison to evaluate releases from uranium mining facilities.

As uranium and thorium occur at elevated levels in numerous occurrences throughout the Canadian Shield, there is the potential for technologically enhanced natural radioactivity from the exploitation of almost any type of metaliferous deposit in the geological region. It was decided to perform the study in northern Ontario as:

(1) The Ontario metallic mineral production accounts for over 40% of the Canadian total.

(2) There are large identified mineral tailings masses in Ontario covering 10 000 ha. Approximately 1/3 of the tailings by area may be considered as non-operating.

(3) In some Ontario mining regions there are relatively large centres of population located near the sites of potential releases of radioactivity.

Given these conditions, a study of the radiological impact from non-uranium mining in Ontario should be generally applicable to non-uranium metal mining activities in the Canadian Shield. However, it is recognized that in some mining regions, other than those investigated, site specific conditions may result in larger radiological impacts than those reported in this study.

2.2 Project Strategy

The project has been divided into three phases:

Phase I: Selection of the Potentially Significant Region and the preliminary characterization of source terms for the area. (The Potentially Significant Region is a mining area having radioactivity, demographic and environmental characteristics that tend to maximize the radiological impact.)

Phase II: Detailed measurements in the Potentially Significant Region.

Phase III: Assessment of the regional radiological impact.

This report covers only phase I, the strategy of which is outlined below:

Review:
(1) Mining activities in principal mining camps in northern Ontario.
(2) Geology, mineralogy and environmental conditions in northern Ontario.
(3) Radiological data for the camps.
(4) Sources and behaviour of radioactivity at the mine/mill sites.
(5) Environmental regulations and standards.
(6) Source terms for uranium mine/mill sites.

Select: Potentially Significant Region

Sample and Analyze:
(1) Radioactivity levels in airborne and liquid effluents.
(2) Radioactivity levels in selected solid waste products.

Results:
(1) Preliminary characterization of source terms.
(2) Evaluation of the significance of the source terms by comparison to standards and to uranium mine source terms.

Recommend:
(1) Further studies in the mining region selected in the study.
(2) Developments; where gaps of knowledge or where requirements for improvements of techniques are identified in this study.

A major element of this study is the selection of the Potentially Significant Region. The analyses of the geology, mineralogy, mining and milling processes and

environmental conditions in the northern Ontario mining camps were performed to the extent necessary to rank the potential importance of the regions. Having selected the region to investigate, emphasis is placed on the sampling of effluents and waste products and the determination of source terms within the region.

2.3 Project Scope

The scope of phase I was limited to the characterization of source terms from the mining, concentrating, and waste management operations principally in one mining region. Both operating and non-operating sites were assessed. Emissions from refineries and smelters were considered outside the scope of the study. The measurement of radioactivity in the environment outside of the property line of the mine/mill facility is also not part of this phase of the project nor is the assessment of radiological conditions in housing in the vicinity of the mining sites.

3.0 SELECTING THE POTENTIALLY SIGNIFICANT REGION

3.1 Introduction

Non-uranium metal mining is a major industrial activity in northern Ontario. Although metal mining is carried out at various locations throughout the northern Ontario Precambrian Shield, six major mining regions can be identified (Figure 3.1). These regions produce a variety of metals having a revenue greater than $2 billion annually (Ont. MNR, 1977). They are located in the Superior and Southern geological provinces of the Precambrian Shield. The types of ores in the operating mines as well as the number of mines and the regional throughput are summarized in Table 3.1.

A major aspect of this study is the selection of one area to be examined in detail, the Potentially Significant Region, herein defined as a region having characteristics such that the radiological impact from technologically enhanced natural radioactivity due to the mining industries in the region is greater than that in the other regions. The choice is based on a number of selection criteria which are discussed in section 3.3 of this chapter. The general conditions in northern Ontario and the site specific characteristics of each of the mining regions were analyzed with respect to the criteria. The Potentially Significant Region was identified as the region having the characteristics which best satisfied the selection criteria.

3.2 Conditions in Northern Ontario

3.2.1 General Geological Setting. With the exception of a relatively small area around Hudson Bay, northern Ontario is underlain by rocks of the Precambrian Canadian Shield. The three major structural provinces of this area are Southern, Superior, and Grenville; the latter two extending into, and forming, the greater part of Quebec. Only the Superior and Southern provinces are germane to this study as none of the mining regions considered are within the Grenville province.

The Superior province is characterized by east-west greenstone belts of weakly metamorphosed Archean volcanic and sedimentary rocks separated by much larger areas of granitic rocks. With the greenstone belts are stratiform base metal sulphide deposits (Cu-Pb-Zn-Ag-Au) as, for example, at Timmins, Temagami, and Manitouwadge and banded iron formations, examples of which occur at Kirkland Lake and Temagami.

The Southern province is one of the smallest structural provinces of the Canadian Shield but is of great mineral importance as it contains the famous Sudbury nickel deposits. In addition there are large quantities of copper and lesser amounts of

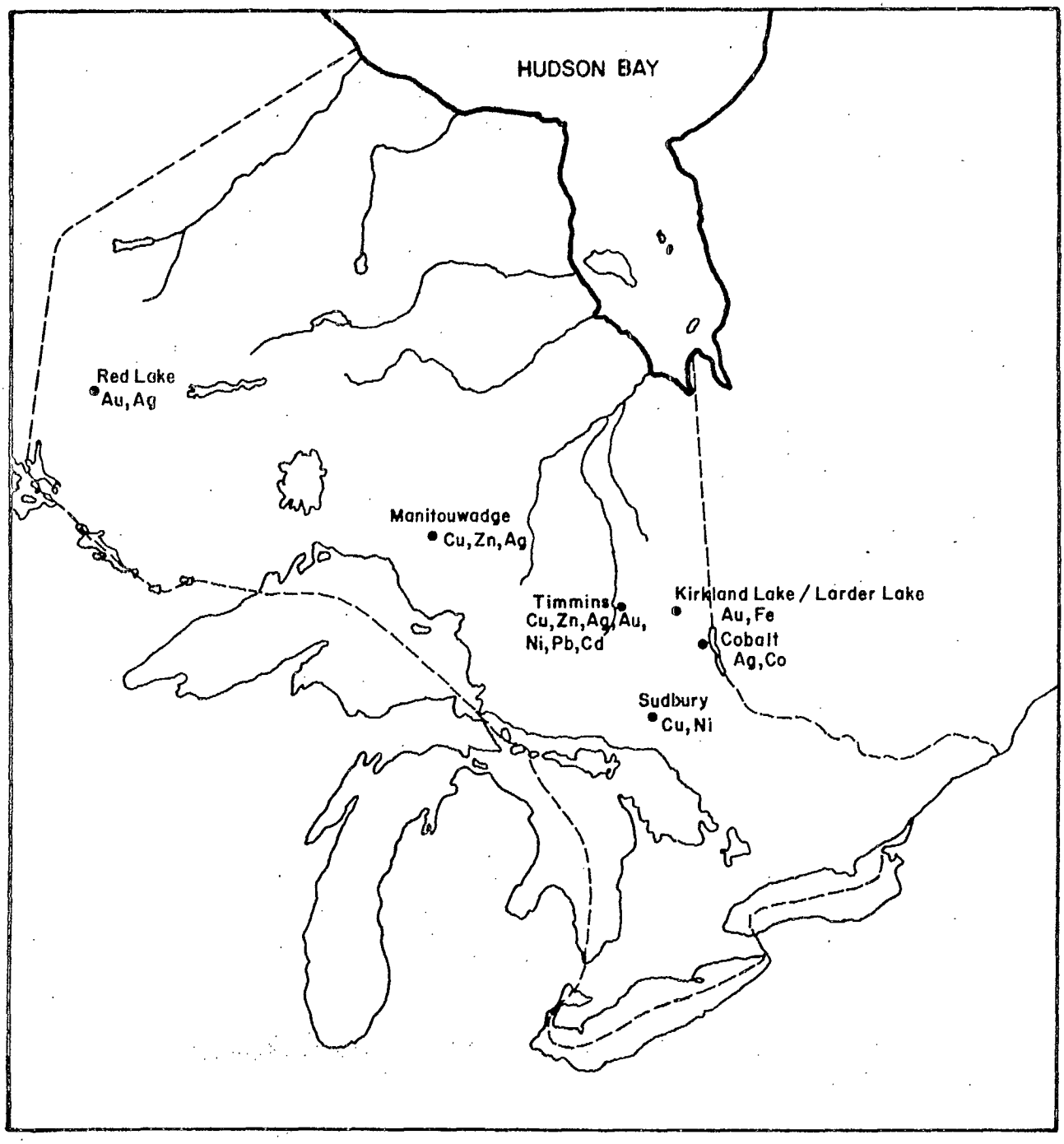


FIGURE 3.1 NORTHERN ONTARIO MAJOR NON-URANIUM MINING REGIONS

TABLE 3.1 MAJOR NORTHERN ONTARIO MINING REGIONS

Region	Number of Operating Mines	Principal Metals	Throughput Tonnes/a
Kirkland Lake Larder Lake	4	Au, Fe	14 000
Timmins	5	Au, Cu, Zn, Ag	19 000
Sudbury	20	Ni, Cu	69 000
Red Lake	3	Au	2 300
Manitouwadge	2	Ag, Cu, Zn, Pb	1 300
Cobalt	2	Ag	5 000

gold, silver and platinum group metals. Large tonnages of uranium ore occur in the quartz pebble conglomerates in the Blind River area. Lithologically, the Southern Province comprise a variety of sedimentary and volcanic rocks intruded by several generations of basic rocks and small granitic batholiths.

3.2.2 Uranium and Thorium Occurrences. There are more than 100 documented occurrences of uranium mineralization in northern Ontario (Robertson, 1968), almost 50% of which are associated with quartz pebble conglomerates and arkoses lying at or near the base of the Huronian (Table 3.2). The conglomerates are commonly pyritic and locally show gold values, though platinum group metals, silver and cobalt are rare. About 30% of uranium-thorium minerals occur in diabase/granite associations as, for example, the pitchblende in veins associated with the Keewanawan dykes of the Theano Pt.-Montreal River area. Twenty percent occur in pegmatites along with rare-earth minerals. The remainder occur in carbonatite, syenite, granophyre, and argillite.

3.2.3 Meteorology in Northern Ontario. The climate of northern Ontario (Chapman, 1968) is classified as modified continental, the modification mainly due to the Great Lakes in the south and Hudson Bay in the north. The mean daily temperature is about 3 degrees centigrade. The average frost-free season lasts for approximately 80 days per year. The annual mean snowfall is about 240 cm. During four months per year the ground is covered with snow which accumulates to 100 cm in some areas. The yearly average precipitation in the Timmins area is about 81 cm. The potential evapo-transpiration, or water need, amounts to 50 cm in the Sudbury region.

TABLE 3.2 URANIUM AND THORIUM MINERALIZATION IN NORTHERN ONTARIO

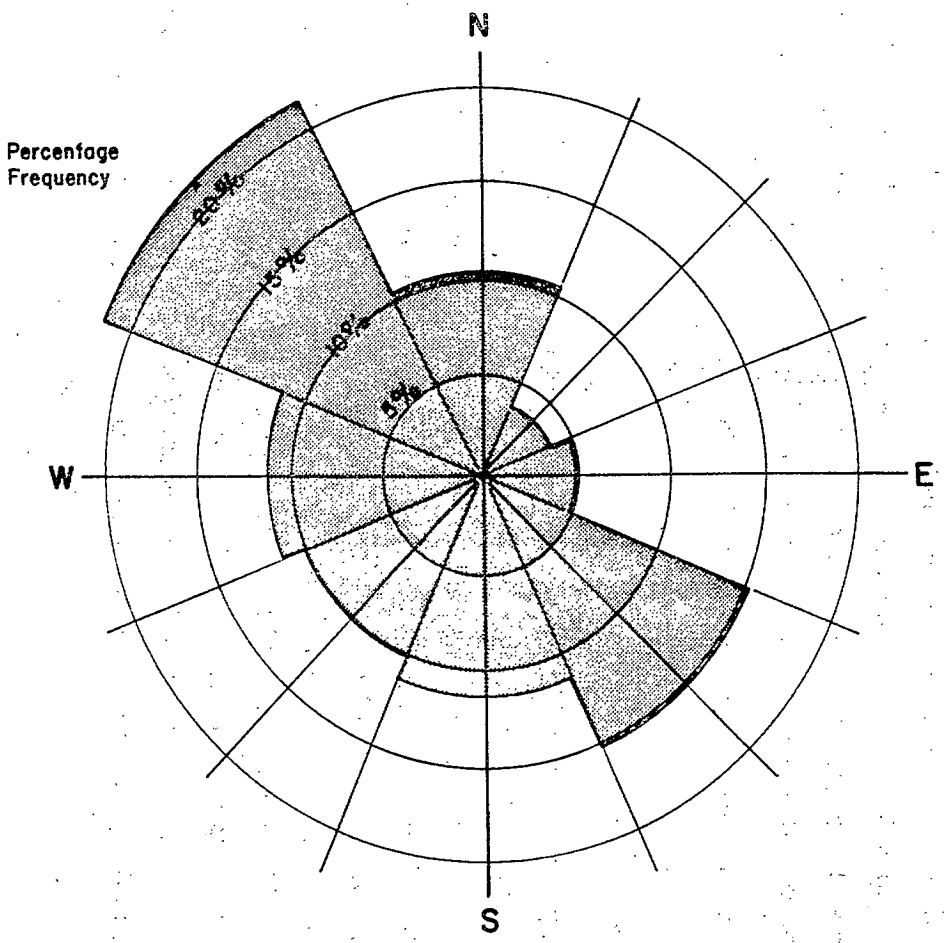
Rock Type	Number of U & Th Occurrences
Conglomerates and Arkose	44
Diabase (granite association)	32
Pegmatite	22
Carbonatite	2
Syenite	2
Granophyre	1
Argillite	1
Others, unspecified	80

Note: With the exception of the occurrences in carbonatite and pegmatite, the majority are of importance for uranium mineralization rather than thorium.

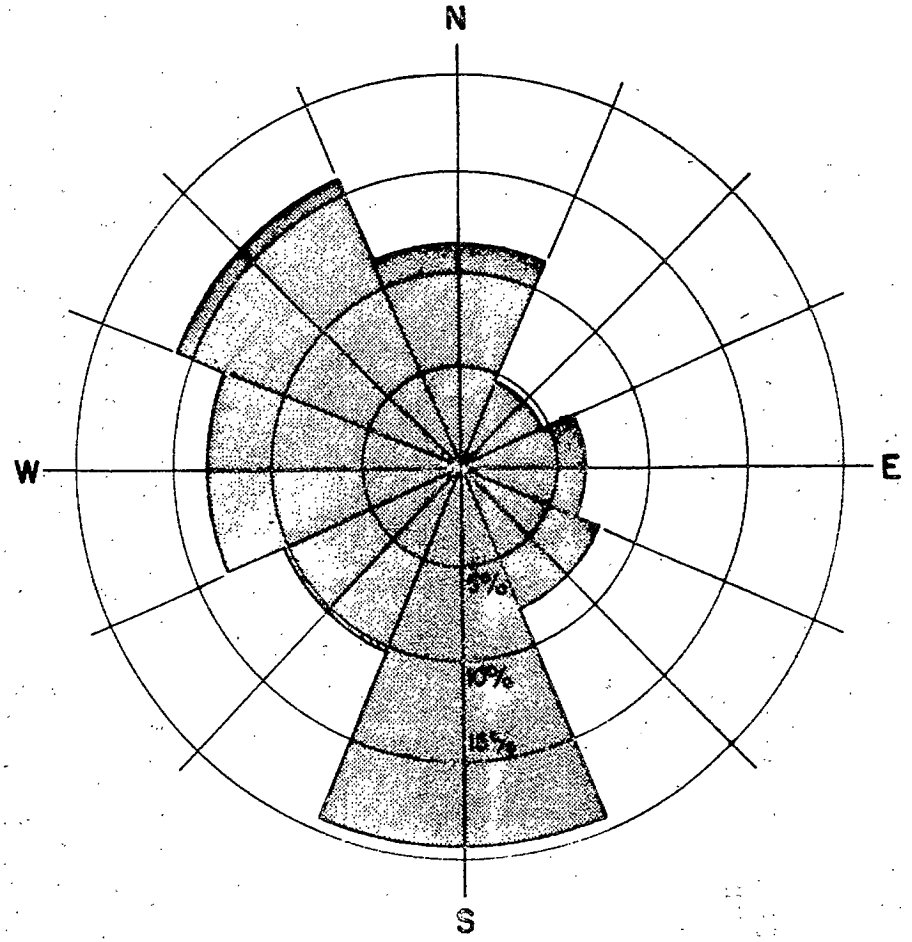
Source: Robertson, 1968.

In northern Ontario, data based on ten year measurements show that in general, northeast winds are most frequent in winter, while west and north winds are quite common. Southwest winds prevail in summer, however, west and northwest winds are also frequent in some areas. Wind speeds average between 3.7 and 5.5 m/s in winter and between 2.8 and 4.6 m/s in summer. Figure 3.2 shows the annual windrose in the Timmins and Kirkland Lake regions. In Kirkland Lake the predominant wind is from the northwest while in Timmins the southerly direction is slightly more prevalent than the northwest.

3.2.4 **Demography, Land and Water Use in Northern Ontario.** The population within a 50 km radius of each of the mining camps is given in Table 3.3 (Statistics Canada, 1979). The Sudbury area has the largest population with over 154 000 people within the 50 km zone while the Red Lake area has only approximately 2000. Investigations of aerial photographs prepared by the Ontario Ministry of Natural Resources of the Timmins, Sudbury, and Kirkland Lake regions indicate that the population centres are very close to the potential release points. Figure 3.3 illustrates the proximity of residential dwellings to non-operating tailings areas as well as to operating sites in the Kirkland Lake area. As may be observed from Table 3.3, only the Sudbury mining area has a recognized Indian reserve which in fact has a relatively small population.



Annual Windrose (8 Compass Points)
for Earlton, Ontario
(taken to be applicable to Kirkland Lake)



Annual Windrose (8 Compass Points)
for Timmins, Ontario

FIGURE 3.2 ANNUAL WINDROSES FOR KIRKLAND LAKE AND TIMMINS

TABLE 3.3 DEMOGRAPHY OF PRINCIPAL NORTHERN ONTARIO MINING REGIONS

Region	Population Within 50 km	Native Population Within 50 km
Kirkland Lake/ Larder Lake	20 000	0
Timmins	60 000	0
Sudbury	154 000	265
Red Lake	2 000	0
Manitouwadge	4 000	0
Cobalt	13 000	0

Because of the limitations of climate and soil, minimal agricultural activity is performed in northern Ontario. More grains are harvested and cattle raised in the Timiskiming region than near any other mining camp (Table 3.4). Hunting and sport fishing is carried out throughout northern Ontario. A study of the sport fishing in Round Lake in the Kirkland Lake area determined that the average harvest in the lake was 2 kg/ha/a, the composition being primarily pickerel and pike (Charles, 1977). Whitefish and perch are the main harvests of the commercial fishing industry of the Sudbury district; the annual catch being approximately 200 000 kg (Hughson, 1971).

3.2.5 Detailed Description of the Geology and Radioactivity in the Major Mining Regions.

1. Timmins (Pyke, 1970; Pyke, 1975)

The central portion of the Timmins ore zone is rich in gold minerals while the non-gold mining areas occur to the south and to the northwest. Gold mineralization was first discovered in the Timmins area in 1909 and more than 50 million ounces of gold have been extracted since that time. The ore minerals occur in the ultramafic rock mainly in the form of oxides. The radioactivity of this area is expected to be low.

Nickel deposits are located south of Timmins and occur within ultramafic intrusions. The geological characteristics of the area suggest that the radioactivity levels associated with the ore are expected to be low as in the gold producing area. The famous Kidd Creek Mine north of Timmins is located on a volcanogenic massive sulphide deposit considered to be unique both for its grade and size and for the fact that the deposit is still

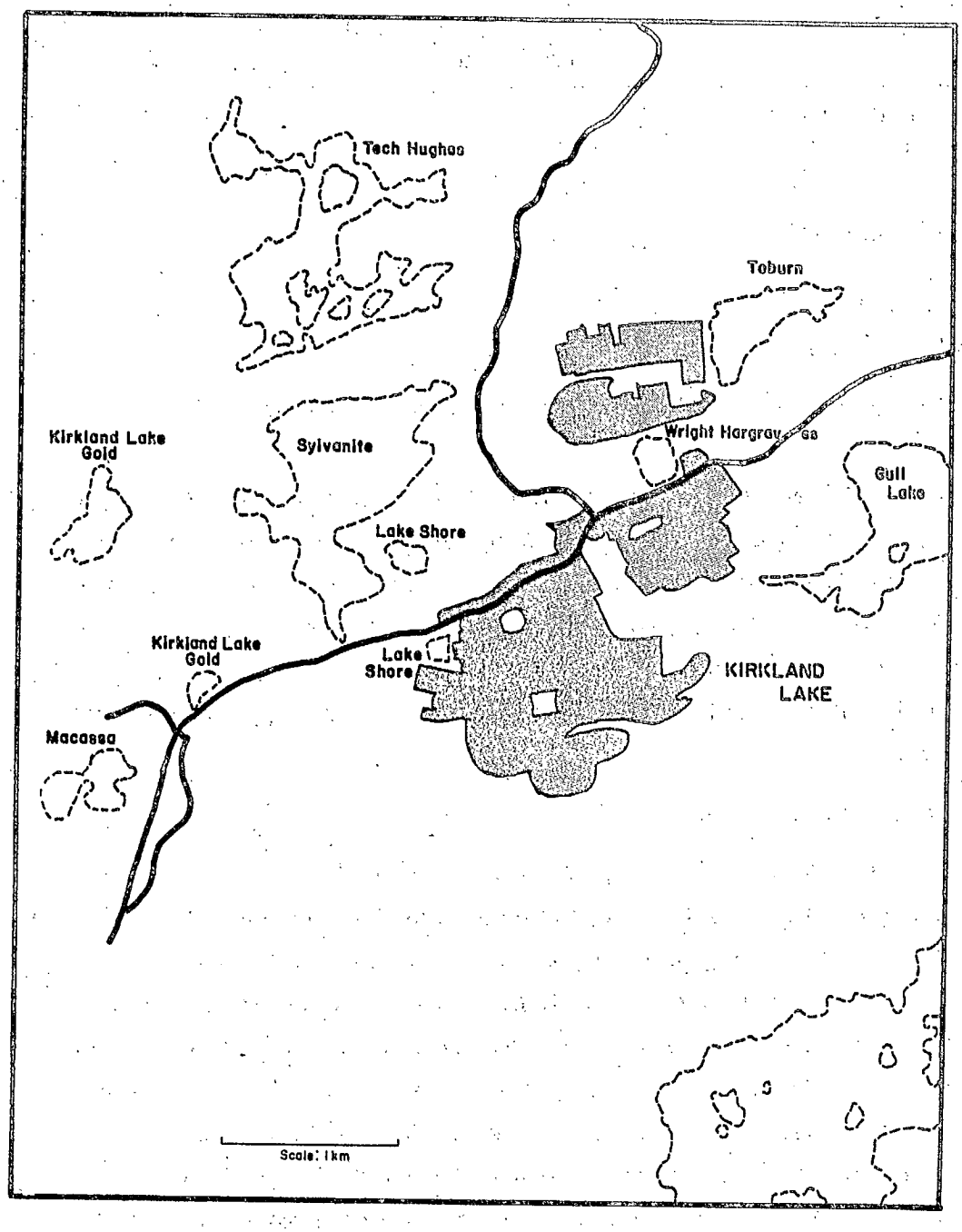


FIGURE 3.3 KIRKLAND LAKE AND SURROUNDINGS FEATURING METAL MINE TAILINGS AREAS

TABLE 3.4 AGRICULTURAL PRODUCTION IN NORTHERN ONTARIO

District	Grains (Oats, Barley, Mixed Grain) x 1000 tonnes	Cattle (Dairy, Beef)
Timiskaming (Kirkland Lake, Cobalt)	41.4	27 200
Sudbury	6.3	8 300
Cochrane (Timmins)	5.2	12 450
Thunder Bay (Manitouwadge)	5.0	11 400
Patricia (Red Lake)	-	-

Note: Data extracted from "Agricultural Statistics for Ontario" (OMAF, 1980)

open to depth. Reserves totalling 88 million tons are composed of copper, zinc, lead and silver with average grades of 2.86%, 5.06%, 0.18% and 65.5 oz/tonne respectively. The Kidd Creek massive sulphides lie at the top of steeply dipping felsic volcanics. At the top of the sulphides, a black graphite-carbonaceous horizon is situated which contains sphalerite, pyrite, chert, rhyolite, argillite, dacite and quartz porphyry fragments. These characteristics of the mining site are favourable for finding elevated levels of uranium and thorium.

2. Kirkland Lake (Charlewood, 1964)

The Kirkland Lake camp was discovered in 1910 and by 1955 it became, with Larder Lake, the largest gold mining region in Canada. The mining activities have since decreased markedly leaving only two operating gold mines and one iron mine along with large operating and non-operating tailings areas. The mines of the Kirkland Lake - Larder Lake camp lie in an east-west trending trough of Archean sediments, tuffs and trachytes set in older basic volcanics. The rocks dip steeply and face south towards a complex and persistant zone of schist (Douglas, 1976). The Larder Lake break, on which the Kerr-Addison mine is located, forms the south boundary of the trough. At Kirkland Lake seven mines (six of which are now abandoned) were developed on veins in a composite fault comprising the Kirkland Lake fault. The zone cuts sediments and a complex of syenitic intrusions elongated parallel into the sediments and pitching west (Douglas, 1976). Ore is found in all rocks traversed by the fault system and consists of quartz-filled fissure veins, breccias and gash veins tributary to the fault fissures. Quartz,

with some calcite, is the principal gangue although it does not form a major part of the ore.

Gold is native in gangue and in fractures in the wall rock. The ore bodies are generally complex silicified zones with tourmaline, calcite and quartz of several ages. Gold and tellurides occur in the quartz and with pyrite in the wall base. Fine pyrite constitutes approximately 2% of the ore. At Kerr Addison the ore bodies lie in and just south of a zone of carbonatised volcanics. At Kirkland Lake, as the main ore body consists principally of composite plutonic syenite rocks which has alkalic feldspar as one of its major consitituent minerals, we may expect to have elevated concentrations of uranium and thorium associated with the mining activities. This is consistent with Robertson (1981) who reported uranium occurrences in the Kirkland Lake area, the uranium being associated with trachyte and trachytic tuffaceous greywacke. In addition, as potassium is one of the elements in alkalic feldspar, high levels of potassium-40 may add to the potential radiological hazard.

3. Sudbury (Thomson, 1960; Thomson, 1969)

Sudbury is one of the world's most important mining camps producing more than 10 billion dollars of mineral wealth from over 60 sulphide occurrences and from more than 20 producing nickel-copper mines. In the Sudbury region there has been extensive exploration for uranium. Robertson (1968) estimated the reserve of uranium in the area to be 8300 tonnes. The exploration for uranium has been performed at the base of the Huronian sedimentary group which is at the same stratigraphic horizon as the uranium deposits in the Elliot Lake area. Radioactive quartz-pebble conglomerates similar to Elliot Lake were discovered but these were much smaller in extent and less concentrated. Nineteen non-commercially exploitable occurrences of radioactive conglomerates have been found. Uranium of grades between 0.05% to 0.1% is confined to widths of 2 metres or less while the lower grade uranium, 0.01%-0.03% occurs over widths up to 20 metres. The uranium to thorium ratio is variable but less than that of Elliot Lake.

There is good evidence of deformation between the Bruce and Cobalt sedimentary groups of the Huronian sequence. In places, the Bruce group lies under the Cobalt group, in others the reverse is true, whereas locally one of the groups is missing. As quartz pebble conglomerates are found at the base of the Huronian sequence, it may be postulated that there is more chance of finding uranium where the Cobalt group is missing or where the Cobalt group underlies the Bruce as this would tend to make the uranium more accessible. It has been determined that the Bruce group becomes thicker towards

the west of the Sudbury region. Also the operating mine sites are generally to the north and east of the main uranium occurrences. Therefore, because of the local geological characteristics of the operating mining sites and absence of information supporting the presence of radioactive material near the operating mines, it is judged that the Sudbury region is somewhat less favourable than other regions for finding elevated radioactivity associated with the operating mines; however, it is recognized that this conclusion is tenuous.

4. Manitouwadge (Pye, 1955)

Mineral deposits in the region contain over 15 million tonnes of ore with exploitable copper, zinc and silver. The ore is in the form of sulphides principally pyrrhotite followed in abundance by pyrite. In a few places chalcopyrite, sphalerite and galena may be associated with the iron sulphides. Information on uranium content in the area has not been reported in the literature and there is no indication of exploitable uranium deposits. However, it is possible to infer that elevated trace levels of radioactivity may be present as the most abundant rock in the area is biotite granite, which usually contains zircon and monazite and other minor minerals carrying radioactive constituents.

5. Red Lake (Ferguson, 1966)

Red Lake is the third largest gold producing area in Ontario. The bedrock in the vicinity of the gold and silver mines consists of Precambrian volcanic and sedimentary rocks. These supracrustal rocks were invaded by later acid intrusives with which the gold minerals are associated.

Studies of the distribution and interrelationship of radioactive elements with zirconium, silica and heavy metals in the intrusive rocks demonstrate the following (Gross, 1952):

- The levels of radioactivity in the gold producing locations are 3 to 4 times higher than in the surrounding area.
- The rocks with the highest concentrations of radioactive material also have the highest concentrations of silica, zirconium and heavy metals which are located in the eastern section of the Dome stock.
- No identified uranium deposits have been uncovered near Red Lake. The closest deposits are 200 km south in Atikoken.

6. Cobalt

Silver is the most important metal mined in the camp with lesser amounts of cobalt and gold. Most of the ore minerals are found in the mafic intrusive rocks which belong to the middle Precambrian group. There have been no reports of elevated radioactivity in the Cobalt area and in fact the levels appear to be lower than expected as the arkosic sedimentary rocks, a major rock type in the area, typically contain approximately 5 ppm thorium. A uranium deposit has been reported in the Cobalt embayment (Meyn, 1979) 100 km from the mining area, however, there is no indication that the deposit extends beyond the Cobalt embayment to the Cobalt mining area.

3.3 Selection Process

3.3.1 Selection Criteria. The radiological impact is assessed by considering the following:

- dose to individuals in a critical group
- collective dose to the exposed population
- changes in the local ecology due to releases of radioactivity to the environment

The selection of the Potentially Significant Region was based on criteria which influence the magnitude of the above. These criteria are summarized in Table 3.5 and the rationale for selecting these factors is explained in the following sub-sections. However, as no rare or endangered species could be identified in any of the mining regions, the criterion based on the presence of such species was eliminated from further consideration in the selection of the Potentially Significant Region.

3.3.1.1 Factors influencing the magnitude of the source terms.

(a) Geological Occurrences of Uranium and Thorium

There is a wide range of associations of radioactivity through a variety of rock types. The general distribution of uranium and thorium in certain igneous and sedimentary rocks has been documented (Figure 3.4). Specific relationships derived from the literature (Table 3.6) provide some guidance as to which areas may contain elevated levels of radioactivity, but the prediction of levels of radioactivity in the absence of direct information is uncertain. What is needed are the site analyses and measurements; however, in many cases these are not available. In spite of the limitations, the information in Figure 3.4 and Table 3.6 can be used with a knowledge of the rock types

TABLE 3.5 CRITERIA FOR SELECTION OF THE POTENTIALLY SIGNIFICANT REGION

(1)	Elevated Levels of Radioactivity	Indicators of potentially high levels are: (a) radiological measurements (b) rock types mineralogical characteristics of area.
(2)	Mining Developments	Size and type of mining developments can affect radiological releases.
(3)	Population Distributions	Dose to individuals may be increased if they live in dwellings near the locations where radioactivity is released. Collective dose is to the first order dependent on the population of the area. Native population may receive higher doses than the general population due to their life style.
(4)	Land and Water Use	Individual and collective doses can be increased by local agricultural activities. The use of local waters as a drinking water supply and for fishing can increase dose.
(5)	Local Ecology	Ecosensitive areas can be identified by presence of rare or endangered species.

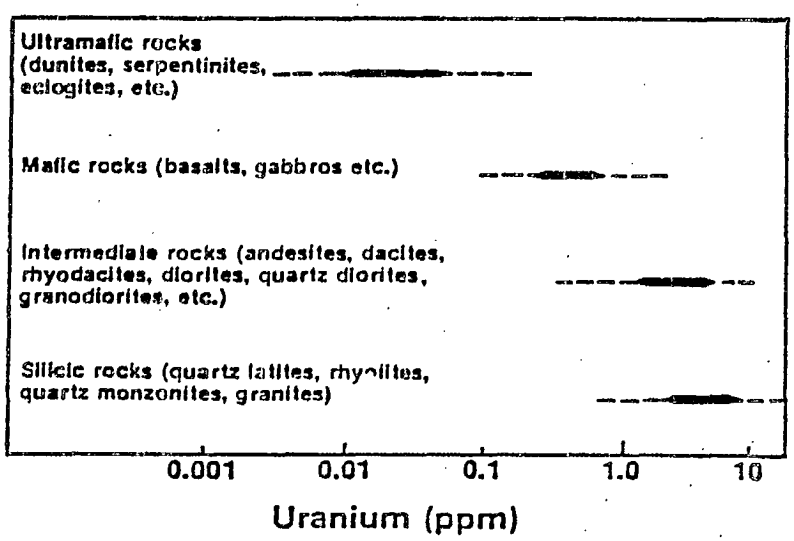


FIGURE 3.4 GENERALIZED DISTRIBUTIONS OF URANIUM IN VARIOUS GROUPS OF IGNEOUS ROCKS

- (From Bliss, 1978)

TABLE 3.6 EMPIRICAL OBSERVATIONS ON URANIUM AND THORIUM OCCURRENCES

1.	No persistent pattern of radioactivity is associated with most base metal ore bodies. Copper, lead and zinc deposits are typically minimally radioactive.
2.	Radioactivity generally increases from basic to the acidic rock type, i.e. uranium and thorium contents generally increase with the silica content, usually in pegmatites and associated with zirconium and rare earth elements.
3.	Carbonatites can be enriched in uranium and thorium particularly if the rare earth content is high.
4.	Although the list of metallic minerals found with uranium veins is extensive, iron minerals constitute the most universal association. Magnetite ore bodies are generally not abnormally radioactive.
5.	Silver minerals and uraniferous veins tend to exist together within the same general region but not usually within the same local environment.

and mineralology in the mining camps to provide indicators as to the probable levels of radioactivity.

(b) Mining Operations

It has been observed in the studies in the United States that the larger the mining development the greater the releases of radioactivity (Appendix A). This relationship is valid since, for example, radon and particulate emissions to the first order are proportional to the surface areas of the tailings areas. Similarly higher annual throughput can be usually related to larger wall areas in underground mines. This increases the probability for radon emanation into the mine atmosphere with eventual emission to the environment. Many other factors must be applied when calculating radiological emissions; however, when initially determining if a region should be screened more closely, the types and magnitudes of the mining activities in a camp are valid considerations.

3.3.1.2 Factors influencing the radiological exposure of the population.

(a) Demography

Analysis of the demographic and environmental characteristics of a region was limited only to providing input into the ranking. Two of the major factors by which to measure the radiological environmental impact of a facility is the radiation dose to the

individual within the critical group and the collective dose of the regional population. People residing close to the points of release will normally receive larger exposures from direct inhalation or ingestion than will someone living further away. Also the larger the population in the region normally the higher the collective dose. It is possible that because of their living and dietary habits the doses to members of a so-called critical group are higher than those of the general population. Accordingly, centers of native people close to sources of radioactivity should be given special consideration as a significant fraction of the food supply for native people originates in the local area from hunting and fishing.

(b) Land Use

In a recent study to assess the radiological impact of uranium mining in northern Saskatchewan (Beak, 1980) it was determined that the inhalation dose from particulates and the ingestion dose from drinking water were the two major components in the dose to individuals. The dose due to the uptake of lead-210 from the ingestion of grains was the largest component of the collective dose. Accordingly, the effect of agricultural activities of a region on the potential population dose must be considered when selecting the Potentially Significant Region.

3.3.2 Analysis and Selection of the Potentially Significant Region. The process of selecting the Potentially Significant Region was based on the Kepner-Tregoe (1965) weighted score decision analysis technique. The criteria discussed in Section 3.3.1 were weighted and the extent to which each mining camp satisfied the criteria was scored. The decision process has some judgemental aspects which are explained below.

It was judged that the most important factors in selecting the Potentially Significant Region relate to different indications of elevated radioactivity associated with the mining developments. Thus higher weightings are generally given to these indicators.

The decision Table is presented in Table 3.7. Radon daughter levels, which are direct indicators or the presence of radioactivity in the micro-environment associated with the mining developments, particularly in the absence of data on the radioactive content of the ore or host rock, were arbitrarily given a rating of 10. The score of a region was obtained by setting the region with the highest average values equal to 10 and the other regions were scored proportionally. The values of the average and maximum radon daughter levels are presented in Table 3.8.

Rock types and mineralogy are indirect indicators of the presence of radioactivity and hence a weighting factor of 7 was assigned. The score is based on an

TABLE 3.7 DECISION MATRIX FOR THE SELECTION OF THE POTENTIALLY SIGNIFICANT REGION

	CRITERIA						
	Radon Daughter Levels	Geology/ Mineralogy	Regional Popul- ation	Nature/Size Mine Sites	Land/ Water Use	Native Popul- ation	Score
Weighting Factor	10	7	6	4	4	2	
Kirkland Lake	7	10	3	4	10	0	214
Timmins	3	5	5	6	3	0	131
Sudbury	3	5	10	7	3	10	185
Cobalt	0	3	1	2	10	0	75
Manitouwadge	10	8	1	6	0	0	186
Red Lake	1	7	1	1	0	0	69

TABLE 3.8 RADON DAUGHTER LEVELS IN ONTARIO MINING CAMPS

Region	Number of Observations	Radon Daughter Concentration (Jm^{-3})		
		Average	Maximum	Minimum
Kirkland Lake	11	1.7E-6	3.7E-6	nil
Timmins	9	6.2E-7	1.7E-6	nil
Sudbury	23	6.2E-7	8.3E-7	nil
Red Lake	7	2.1E-7	4.2E-7	2.1E-7
Manitouwadge	4	2.4E-6	5.8E-6	nil
Cobalt	4	<2.1E-7	2.1E-7	nil

Note: Measurements tabulated were taken in active mine areas or from ventilation exhausts. Data provided by the Mines Accident Prevention Association of Ontario and by Wilroy Mines.

evaluation of the typical radioactivity content of the ore and host rock. A region's score was increased depending on the number of reported uranium occurrences.

The presence of radioactivity does not appear to follow a pattern based solely on the type of mine. However, the larger the mine and the more diversified the types of metal extracted the greater the probability of an increased radiological loading of the environment. Accordingly a weighting factor of 4 was assigned and the score was divided equally based on the throughput and diversity of ores.

The collective dose, to a first approximation, is dependent on the population of a district. Therefore, population of a region was given a relatively high weighting factor of 6. The scores are generally proportional to the population. Some modification of the score was made to account for the potential increase of the dose to the critical group due to the presence of large abandoned tailings areas being very near residential dwellings. An examination of aerial photographs of the three most important areas, Timmins, Sudbury, and Kirkland Lake reveal that they all have tailings areas near residential locations.

Land and water usage can indirectly affect the radiological impact particularly the collective dose. Accordingly a weighting factor of 4 was assigned. The scores were equally calculated based proportionally on the amount of grain harvested and the cattle raised.

The presence of native groups in a region was given a low rating of 2 as it was judged that this would not be a major influence in the selection of the potentially significant region. The Sudbury area was given a score of 10 as it has the only identified native population.

As shown in Table 3.7, the mining camps in order of assessed priority as the Potentially Significant Region are:

1 Kirkland Lake
2 Manitouwadge
3 Sudbury
4 Timmins
5 Cobalt
6 Red Lake

The radon daughter measurements, geology and mineralology of the Manitouwadge camp indicate that the radioactivity associated with the mining developments in this camp is potentially higher than that of the others. The demography of the Sudbury region is consistent with a large radiological impact given that there are significant quantities of radioactive materials released. The Kirkland Lake region possesses both favourable conditions for elevated radioactivity and a relatively large population and was accordingly chosen as the Potentially Significant Region. It was also decided to take selected measurements in the Timmins region as the region has identifiable mining areas where minimal radioactivity is expected on the basis of geology and mineralogy and whereas in others higher levels of radioactivity may be expected. The measurement program undertaken to validate the selection analysis for the Potentially Significant Region was cost-effective as the Timmins and Kirkand Lake areas are geographically close.

4.0 CHARACTERIZATION OF SOURCE TERMS

4.1 Gamma Exposure Rate Surveys

Gamma exposure rate surveys were performed at the major mine sites in the Kirkland Lake and Timmins areas using a low range gamma survey meter. The measurements were performed generally during the initial visit to a site; the main purpose of which was to acquaint mine management about the study and to gain approval for future entries to the site in order to perform detailed sampling. The measurements were intended as a low cost, albeit insensitive, method of potentially discriminating the levels of radioactivity at the different sites. Not unexpectedly, normal background levels (5-10 uR/h) were measured at most sites. At the Wilroy mine site, however, elevated gamma fields up to 15 times normal background were measured. Subsequent gamma spectroscopy analysis indicated that the major contribution to the gamma field was from potassium-40 and not from radionuclides of the uranium or thorium series.

4.2 Radioactivity in Solid and Liquid Wastes and in Liquid Effluents

The concentrations of radionuclides of tailings materials, waste rock, liquid wastes and liquid effluents are summarized in Table 4.1. The radiochemical analyses were performed by Beak Analytical Services and by the Radiochemical Laboratory of Carleton University. Where errors are shown in Table 4.1, they represent the differences between two independent analyses of the sample. From the data, it is seen that the average ratio of thorium-230 to radium-226 levels is approximately 5 for solid waste samples and 4 for water samples. This is consistent with the data reported for acid leach uranium mills. One set of results in the table warrants scrutiny. The very high concentrations of radium-226 and thorium-230 in the Lakeshore tailings sample was verified by repeating the analyses three times. The significance of this sample is discussed in the context of the frequency distribution of the sample radionuclide concentration near the end of this section.

The concentration of radium-228 in the samples was determined by measuring the beta from the daughter Actinium-228 (Smithson, 1979). As explained in the footnote to Table 4.1, this procedure does not have sufficient sensitivity, and it was not possible to determine by this technique, if radium-228 was indeed present in the samples. However, an airborne gamma-ray spectrometer survey demonstrated the presence of both thorium and uranium in the Kirkland Lake and Timmins regions (Ont. MNR, 1979), and accordingly it would be expected that samples from these areas should contain both radium-226 and

TABLE 4.1 RADIOACTIVITY IN SOLID WASTE AND WATER SAMPLES

1. Kirkland Lake/Larder Lake Area

(i) Solid Waste Samples*

Sample Location	Date of Sampling	U	Ra-226	Th-230	Th-232	Th-228	Gross Ra-228	Alpha
Wilroy Tailings A	Nov/81		19					
Wilroy Tailings B	Nov/81		56					
	Mar/82		37 ± 7	380 ± 50	n.d.	15		110 ± 30
Wilroy Waste Rock	Nov/81	0.4	19	410 ± 50	150 ± 20	230 ± 40	<540	130
	Nov/81		85					
Wilroy Mill Discharge	Nov/81		37					
Kerr Addison Tailings 1	Nov/81		19					
Kerr Addison Tailings 2	Nov/81		<15					
Adams Tailings	Nov/81		<15					
Abandoned Areas								
Lakeshore	Nov/81	0.4	1670 ± 50	9860 ± 240	420 ± 50	41 ± 7	<680	1110
	Nov/81		110					
Sylvanite	Nov/81	<0.1	15 ± 7	250 ± 50	≤4	≤15	<480	63
	Nov/81		110					
Wright Hargrave	Nov/81		110					
Teck Hughes	Nov/81		56					
Toburn	Nov/81		19					

* Concentration is expressed in mBq/g (dry weight) except in the case of uranium, which is expressed in ppm.

TABLE 4.1 RADIOACTIVITY IN SOLID WASTE AND WATER SAMPLES (CONT'D)

(ii) Water Samples*

Sample Location	Date of Sampling	U	Ra-226	Th-230	Th-232	Th-228	Ra-228	Gross Alpha
Wilroy Pond A	Nov/81		< 4					
Wilroy Pond B	Nov/81		70					
	Nov/81	2.9	7	370	48	37		≤56
Wilroy after Decant	Mar/82		26	240	19	<22		70
Wilroy Mine Water	Nov/81		22					
Kerr Addison Pond 1	Nov/81		4					
Kerr Addison Mine Water	Nov/81		44					
	Nov/81	0.99	63	380	<33	41	650	≤11
Kerr Addison Decant	Mar/82		33	270	< 7	22		7
Kirkland Lake Drinking Water	Nov/81		< 4					
Adams Mine Water	Nov/81		22					
Adams Tailings Decant	Nov/81		< 4					

* Concentration is expressed in mBq/L (dissolved) except in the case of uranium, which is expressed in ppb.

TABLE 4.1 RADIOACTIVITY LEVELS IN SOLID WASTE AND WATER SAMPLES (CONT'D)

2. Timmins Area

(i) Solid Waste Samples*

Sample Location	Date of Sampling	U	Ra-226	Th-230	Th-232	Th-228	Ra-228	Gross Alpha
Dome Tailings Slime	Nov/81		< 15					
Kidd Creek								
Tailings Slime	Nov/81		37					
Mill Discharge	Nov/81		200					
	Mar/82		37 ± 19	320 ± 30	30	7		7 ± 7

(ii) Water Samples+

Sample Location	Date of Sampling	U	Ra-226	Th-230	Th-232	Th-228	Ra-228	Gross Alpha
Dome Mine Water	Nov/81		< 4					
Dome Tailings Discharge	Nov/81		11					
Kidd Creek Mine Water Before Treatment	Nov/81		150				170	
	Nov/81	2.7	220 ± 30	370 ± 41	7 ± 3.7	59 ± 59		44
After Treatment	Nov/81		150				280	
	Nov/81		93 ± 19	440 ± 41	26	41 ± 26		≤100
	Mar/82		11 ± 7	260 ± 37	11	<74		22 ± 7
Kidd Creek Tailing Decant	Mar/82		26 ± 15	280 ± 19	7	<37		22 ± 7
Timmins Drinking Water	Nov/81		11					

(*) Concentration is expressed in Bq/kg (dry weight) except in the case of uranium, which is expressed in ppm.
(+) Concentration is expressed in mBq/L (dissolved) except in the case of uranium, which is expressed in ppb.

FOOTNOTES FOR TABLE 4.1

(a) Determination of Ra-228 was done by isolating and counting the daughter Ac-228, which has a 6.13 hour half life. In three separate determinations, using two quite different procedures, no statistically significant decay was seen over two half lives. The reported concentration represents the maximum activity which could be concealed in the background and escape detection, accordingly the concentrations of Ra-228 are given in terms of "less than" certain values.

(b) Thorium-228 concentrations were calculated from the alpha spectrum. This may include Po-210 and is therefore an upper limit.

radium-228. A similar problem with analytical procedure to measure radium-228 may have occurred in the Elliot Lake study (MacLaren, 1978) in which all the radium-228 results for water samples were presented as 0 ± 74 mBq/L. This result appears to be low, given the geology of the Elliot Lake area.

After it was determined that the initial radiochemical analyses were inconclusive with respect to radium-228, two samples from Wilroy mines were analyzed by gamma spectroscopy using a high resolution Ge(Li) detector to determine if radium-228 could be detected. The results of the analyses of the Wilroy waste rock and tailings decant presented in Table 4.2 confirm that waste rock and tailings effluent samples contain both radium-226 and radium-228. The data indicate that the relative activity of radium-226 and radium-228 are approximately equal. However, the most prominent nuclide in the samples was potassium-40, having a relative activity many times that of radium-226. The relative activity of each nuclide was derived from data presented in Table 4.3 as follows. The radium-226, and radium-228 daughters are assumed to be in equilibrium with their parents. The counting rate attributed to each identified gamma ray line was corrected for detector relative efficiency. This value, the relative abundance of each gamma ray line was then compensated by the gamma abundance in order to calculate the relative activity of each daughter. Using the assumption that the daughters are in equilibrium with the parents, and compensating for the branching ratios, an average relative abundance of radium-226 and radium-228 was derived. The technique of performing gamma spectroscopy with Ge(Li) detectors to measure the activities of uranium and thorium series nuclides has recently been reported (Whittaker, 1982) at the 1982 Health Physics conference.

TABLE 4.2 RELATIVE ABUNDANCE OF RADIUM-228 AND POTASSIUM-40 WITH RESPECT TO RADIUM-226 IN WILROY SAMPLES

	Ra-228 / Ra-226	K-40 / Ra-226
Sample		
Waste rock	1.75	57
Tailings decant	0.73	5.6

For this study, all aquatic samples were filtered before analysis to determine the dissolved radionuclide concentration. This procedure is consistent with the

TABLE 4.3 GAMMA ANALYSIS OF WILROY WASTE ROCK

Energy (keV)	Activity (counts x 10^3)	Radionuclide Identified
74.4	7.5	
76.4	7.7	
86.8	2.5	
92.4	4.8	several
185.6	3.2	
209.2	1.8	
238.4	26.5	^{212}Pb
241.6	4.1	^{224}Ra
270.4	2.2	^{228}Ac
295.2	6.9	^{214}Pb
338.4	5.4	^{228}Ac
352.0	11.9	^{214}Pb
511.2	4.8	^{208}Tl
583.6	8.3	^{208}Tl
610.0	8.9	^{214}Bi
912.0	6.4	^{228}Ac
969.6	3.2	^{228}Ac
1121.2	1.8	^{214}Bi
1461.6	49.2	^{40}K

requirements in the Canadian Metal Mining Liquid Effluent Regulations and Guidelines (Fisheries and Environment, Canada, 1977). Measurement of total activity was not attempted as experience has shown that the analysis is highly non-reproducible and the data are generally very difficult to interpret due primarily to the difficulties of obtaining consistent samples (Tai-Pow, 1982).

Radioactivity measurements in the environment have been found to be lognormally distributed (Waite, 1977). The concentrations of radium-226 and thorium-230 in solid wastes and in aquatic samples follow a lognormal distribution (Figures 4.1 and 4.2); the distribution parameters, median, mean and 90 percentile values (Table 4.4) were calculated by methods reported by Aitchison (1957). These results show that the average concentrations of radium-226 in the solid wastes and in the various bodies of

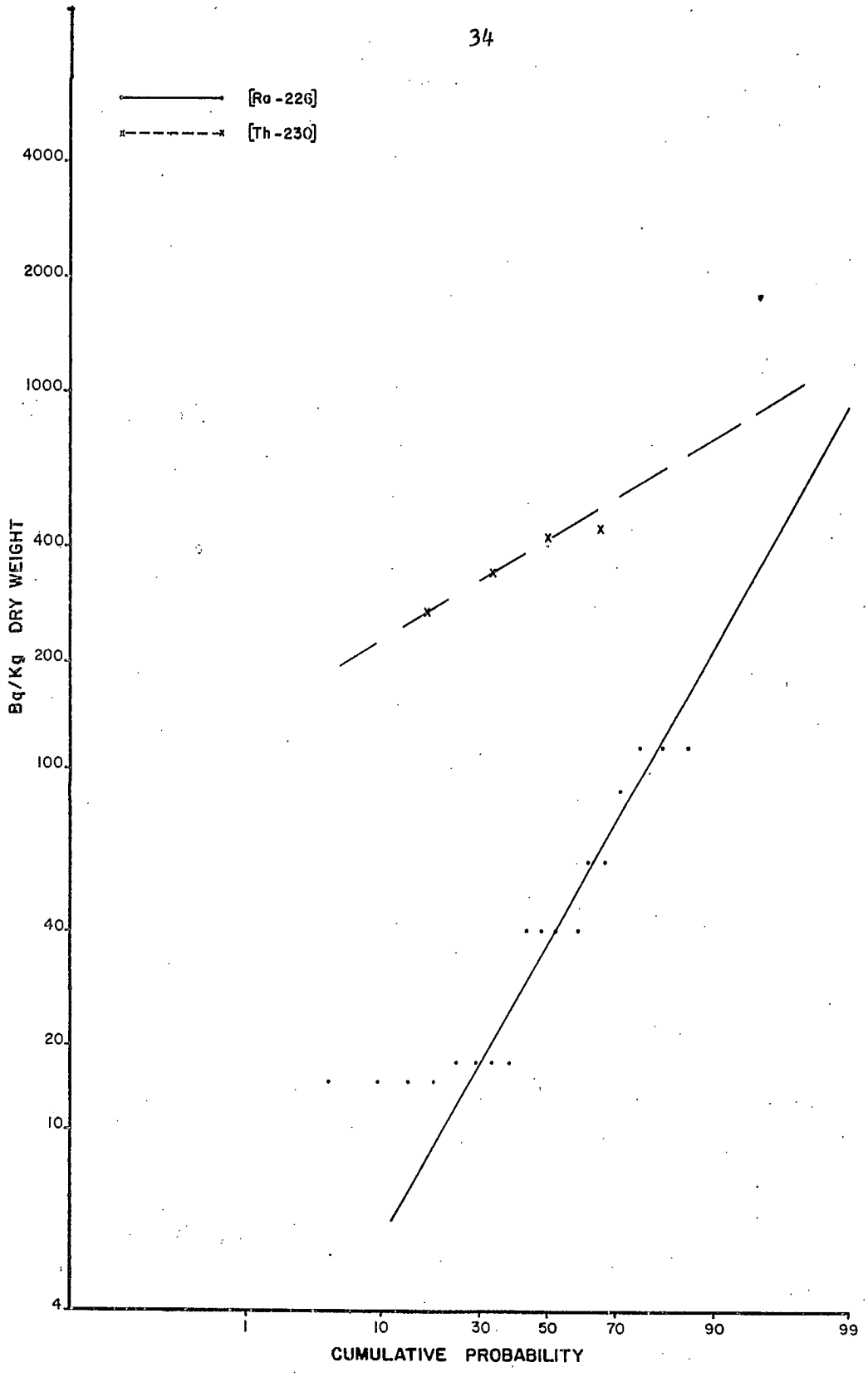


FIGURE 4.1 CONCENTRATIONS OF RADIUM-226 AND THORIUM-230 IN KIRKLAND LAKE AND TIMMINS MINE/MILL SOLID WASTES

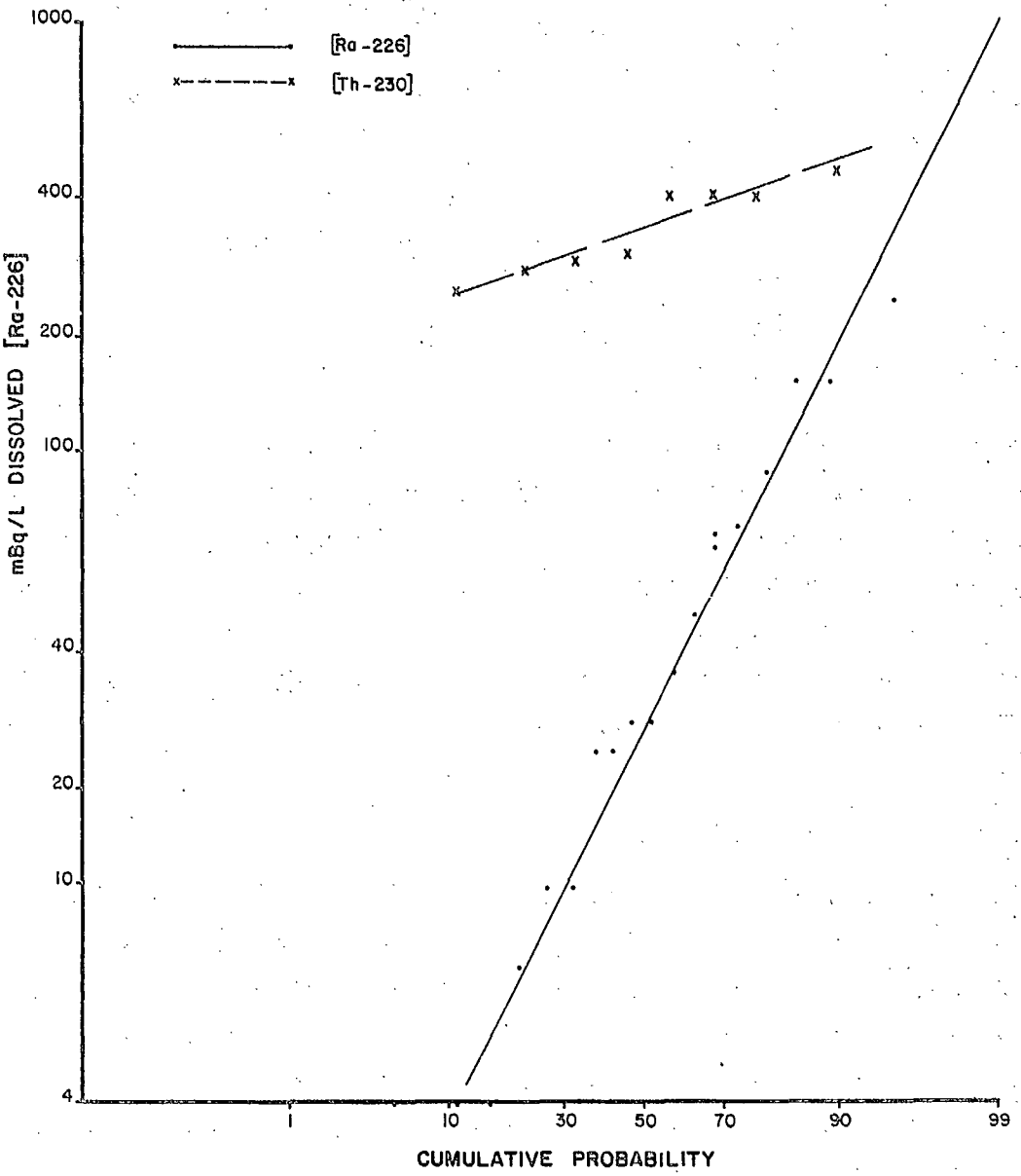


FIGURE 4.2 CONCENTRATIONS OF RADIUM-226 AND THORIUM-230 IN KIRKLAND LAKE AND TIMMINS MINE/MILL LIQUID EFFLUENTS

TABLE 4.4 SOLID WASTE AND WATER SAMPLE RESULTS FOR Ra-226 AND Th-230 FITTED TO LOGNORMAL DISTRIBUTION

	Median	Mean	Bq/kg (Dissolved) Std. Deviation	90 percentile
Solid Waste - Ra-226	35	85	52	200
Th-230	370	440	17	670
			mBq/L (Dissolved)	
Water - Ra-226	25	85	56	185
Th-230	290	340	10	440

water are less than 110 Bq/kg and 110 mBq/L respectively with the majority of samples having activities less than 37 Bq/kg or 37 mBq/L respectively. A sample of Lakeshore tailings, an abandoned area in Kirkland Lake, contained very high levels of radioactivity, 1.7 kBq/kg of radium-226 and 9.9 kBq/kg of thorium-230. However, as shown in Figure 4.1 these values are outliers from the main distribution of the concentrations, that is they do not belong to the same population of radionuclide concentrations as the rest of the samples. The significance of this sample can only be determined by further measurements.

4.3 Liquid Effluent Source Terms

The liquid effluent source terms are summarized in Table 4.5. The source terms are calculated from the values of the concentrations of radionuclides in effluent samples as taken at the final control point or equivalent. These data are presented in Table 4.1. The loading from a site of the aquatic environment is taken to be the sum of the loading of the mine water and the tailings effluents. The loading of radium-226 ranges from less than 22 MBq/a from the Dome site, where minimal activity was expected, up to 550 MBq/a at the Kidd Creek site. The major factor for the high loading from the Kidd Creek site was not only due to the high concentrations of radioactivity in the effluent streams but also the large volumes of waste waters released to the environment. The loading of thorium-230 is higher than that of radium-226 by a factor of 2.9 to 11, the average ratio being 7.8. The activities of thorium-232 and thorium-228 discharged to the environment are significantly less than that of thorium-230. It was not possible to report radium-228 loading from the results of radiochemical analyses;

TABLE 4.5 SUMMARY OF AQUATIC SOURCE TERMS

Site	Flow L/s	Ra-226		Th-230		Th-232		Th-228	
		mBq/L	MBq/a	mBq/L	MBq/a	mBq/L	MBq/a	mBq/L	MBq/a
Kidd Creek									
Mine Water	57	130	220	360	630	15	26	48	85
Tailings Decant	400+	25	330	280	3520	7	93	<37	<440
Dome									
Mine Water	19	< 4	< 2						
Tailings Decant	53	11	19						
Wilroy									
Mine Water	28	22	20						
Tailings Decant	27	26	22	240	210	19	16		
Kerr Addison									
Mine Water	44	56	78	380	520	<33	<48	41	56
Tailings Decant	1.1	33	1	270	11	< 7	< 0.3	22	0.7

+ This value is the mean flow rates based on summer and winter conditions as estimated by the operator.

however, as discussed above, the gamma spectroscopy analyses suggest that the radium-228 loading may be of the same order as that of radium-226.

4.4 Airborne Emissions

4.4.1 Radon Emissions.

4.4.1.1 Radon emissions from tailings areas. The radon flux from tailings areas can be estimated either indirectly from the concentration of radium-226 in the tailings or measured more directly using various active or passive radon detectors systems. Radon emissions from tailings areas can be estimated by the empirical expression (Schaiger, 1974):

$$Rn = 1.6\ CRa$$

where: Rn is the radon flux (mBq/m^2-s)

CRa is the concentration of radium-226 in the tailings (Bq/kg)

This expression must be modified to account for the following factors.

(1) In northern Ontario the ground is snow covered and frozen approximately 35% of the year. The effect of freezing is to reduce the radon emanation by a factor of 3 to 400 depending on the moisture content of the tailings (Zettwoog, 1981). The snow cover will further reduce the radon emanation. Accordingly, the contribution to the annual radon loading during the winter is negligible compared to that during summer and can be neglected.

(2) Laboratory analyses of surface (0.5 m depth) tailings materials from the abandoned areas have shown the water content to be of the order of 18%. At depths greater than 0.5 m the tailings are saturated with water and are referred to colloquially as "soup" by the Kirkland Lake residents. Wet tailings have traditionally been considered to emit less radon than dry tailings; for example, in the assessment of the radiological impact of uranium mining in northern Saskatchewan (Beak, 1980) the radon emissions were estimated to be 45% lower than predicted by the Schaiger expression due to the water content of the tailings materials. Recent work (Strong, 1982), however, has indicated that in fact wet tailings emit more radon than do dry tailings by a factor of approximately 3.5 at the 18% moisture content. However, for this study the Schaiger relationship was used with the same proportionally constant (0.65) as in the Beak (1980) study in order to facilitate comparison between source terms of uranium and non-uranium mines.

(3) Operating tailings areas in the regions of interest are mainly water covered. The percentage of tailings area that becomes "dry beach" in the summer months can only be crudely estimated by the operators to be approximately 10%. In calculating the radon emissions it is assumed that negligible radon is released from the water covered portion of the tailings area and that the Schaiger relationship as modified in item (2) is appropriate.

The emissions calculated from the Schaiger relationship are summarized in Table 4.6. The radium concentration in the tailings are given in Table 4.1. There are two significant sources of radon, the Kidd Creek site with an annual release of approximately 1.5 TBq and the collective release from the Kirkland Lake abandoned tailings areas with an annual release of over 11 TBq. Given the magnitude of these releases and the difficulty of using the Schaiger expression it was decided to attempt to measure the radon emissions at selected tailings locations directly using independent methodology.

TABLE 4.6 RADON EMISSIONS FROM TAILINGS AREAS (FROM MODIFIED SCHAIGER EXPRESSION)

Site	Area (Ha)	Radium Level (mBq/g)	Radon Emissions (GBq/a)
Wilroy	29	37	14
Kerr Addison	36	19	8
Kidd Creek	1200	93	1 480
Kirkland Lake Abandoned Tailings (Less than 3 km from Town)	800	110	11 800

Various methods have been reported to measure radon emanation (Countess, 1976; Novak, 1982). We selected a method that fulfilled the following requirements:

(1) The technique should be proven under field conditions.

(2) A time integrating detector system is essential to smooth out the large short term fluctuations of radon emanation. The measuring technique must have adequate sensitivity to measure low concentrations of radon.

(3) The detectors must be commercially available, maintenance free, passive and cost effective.

The methodology to measure radon emanation selected was developed by Fleischer (1980) using a cellulose track etch detector supplied by Terradex Corporation. These detectors, which have been calibrated in the Environmental Measurements Laboratory radon chamber, demonstrate good reproducibility and have a sensitivity of 480 mBq-d/L (Alter, 1981). A filter placed on the mouth of the detector cup prevents the detector from responding to ambient radon daughters and thoron.

The two techniques used in the study to measure radon fluxes are the "can" technique and the "mat" technique (Fleischer, 1979; 1980). In the first method the radon flux is measured just above the surface through which the radon escapes. A tube with an open end is pressed into the soil and the radon accumulates in the closed air space where the concentration is measured with the track etch detector (Figure 4.3). It was found experimentally that the concentration in the air space is an increasing function of depth below the surface to which the tube is embedded, however, at a depth of 2 metres the concentration becomes constant (Fleisher, 1982). After a long exposure, much greater than the radon half life, the flux is given by the following expression:

$$J = eC(D\lambda)^{1/2}\,(1+h/(D/\lambda)^{1/2})$$

where: e is the porosity of the medium
C is the radon concentration in the air space (mBq/m^3)
D is the bulk diffusion constant (m^2/s)
λ is the radon decay constant (sec^{-1})
h is the height of the tube above the surface (m)
J is the radon flux (mBq/m^2-s)

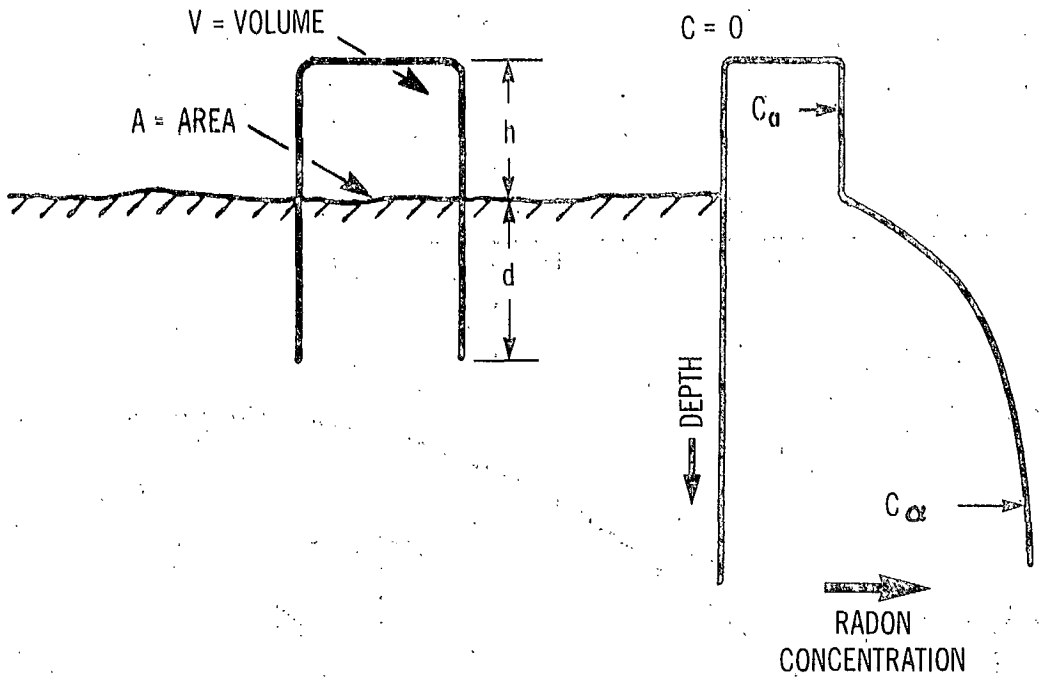


FIGURE 4.3 A CAN IMPLANTED IN THE SOIL ACCUMULATES RADON TO A CONCENTRATION THAT DEPENDS ON THE FLUX THROUGH THE AIR-SOIL INTERFACE (From Fleischer, 1980)

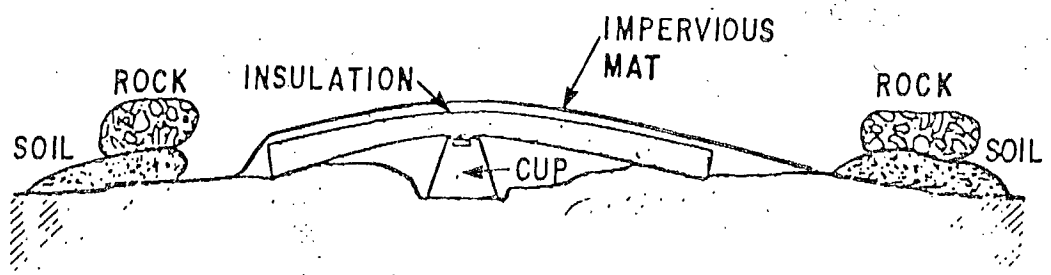
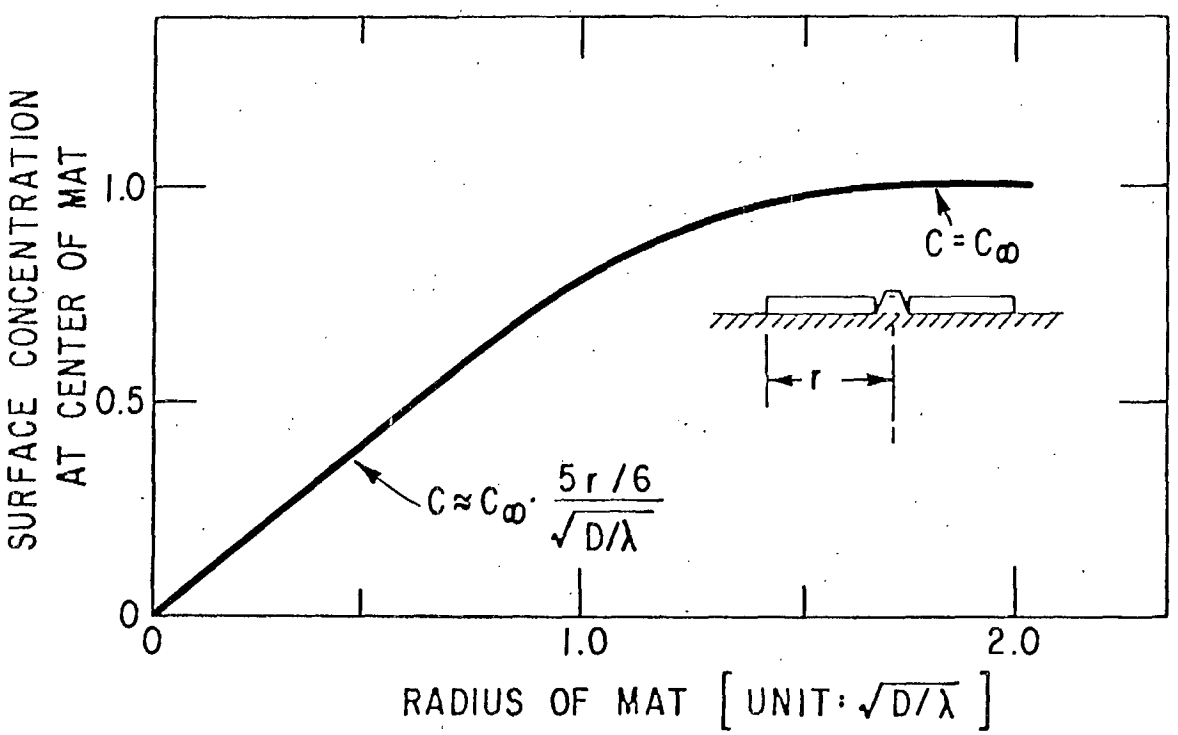


FIGURE 4.4 METHOD OF MAKING RADON MEASUREMENTS AT THE SURFACE USING THE MAT TECHNIQUE



D = Radon Diffusion Constant

C_∞ = Radon Concentration at Great Depth in Uniform Medium

λ = Decay constant

FIGURE 4.5 RADON CONCENTRATION UNDER CENTER OF IMPERVIOUS GROUND COVER (From Fleisher, 1979)

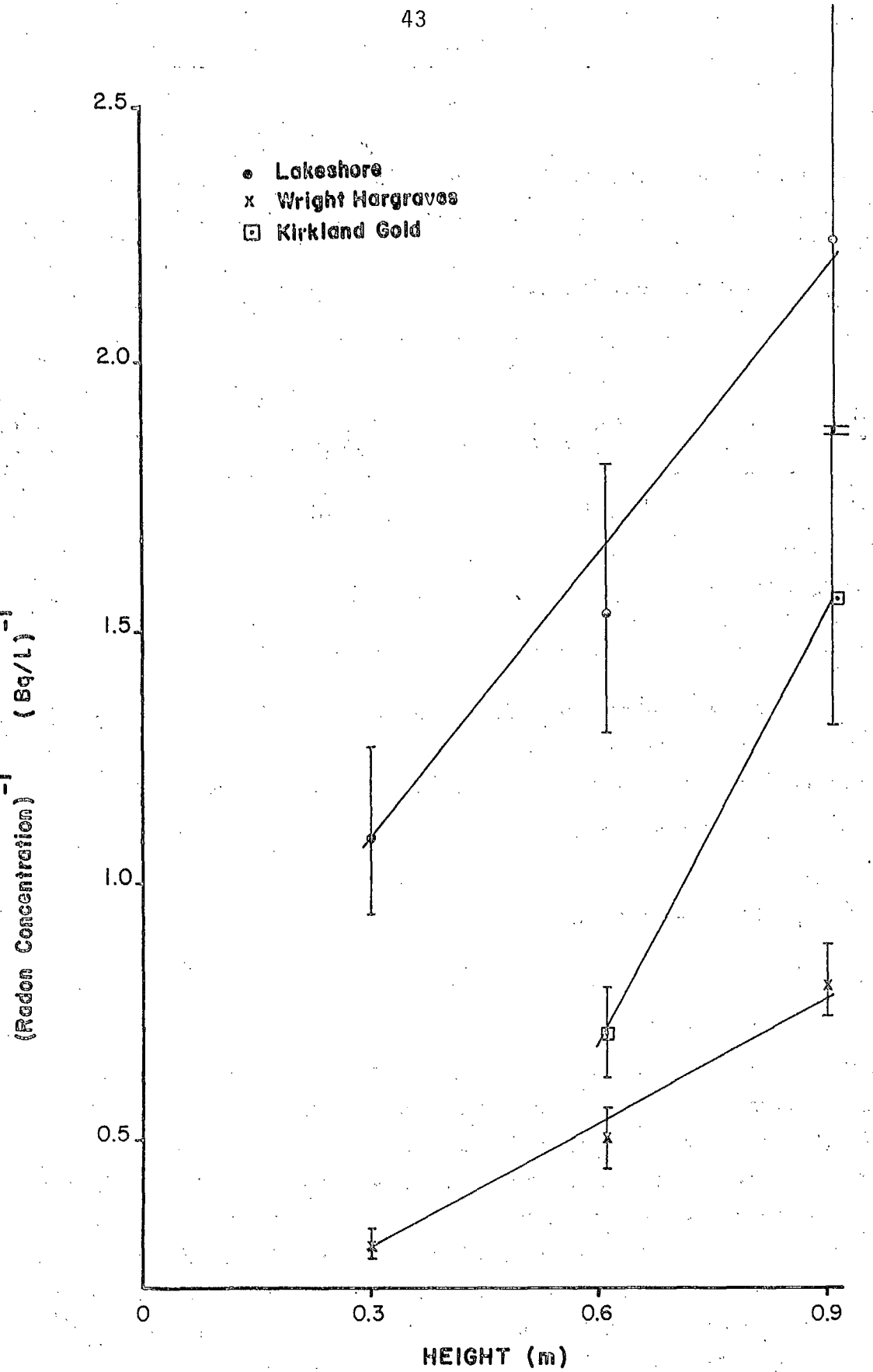


FIGURE 4.6 MEASUREMENT OF RADON EMANATION FROM KIRKLAND LAKE ABANDONED TAILINGS AREAS

From the plot of the inverse of the concentration against the height (Figure 4.6) it is possible to calculate the flux and bulk diffusion constant. These are given in Table 4.7.

TABLE 4.7 DIRECT MEASUREMENT OF RADON FLUX

	Can Method		Mat Method
Site	Radon Flux (mBq/m^2-s)	Bulk Diffusion Constant (m^2/s)	Radon Flux (mBq/m^2-s)
Lakeshore	8.9	2.7E-6	13
Kirkland Gold	5.6	1.1E-7	2
Wright Hargraves	20	5.8E-7	41
Kidd Creek			1

In the mat method, the radon concentration is measured above the surface using an impermeable mat which covers the radon detector (Figure 4.4). The radon concentration at the centre of the mat ground cover is given by (Figure 4.5):

$$C = 5/6C^*r/(D/\lambda)^{1/2}$$

where: C^* is the equilibrium radon concentration
r is the radius of the mat

The flux is given by:

$$J = -eC^*(D\lambda)^{1/2}$$

The value of $(D/\lambda)^{1/2}$ is about 30 cm for dry soil and about 10 cm for wet soil. The mat used was 1 metre square and hence much larger than the 10 cm for wet soil. Consequently, the radon concentration measured approximates C^*. The values for the diffusion constants were calculated for each tailings area as determined from the can method. The radon flux was then calculated based on this information. In the absence of site specific data for the diffusion constant, and because of the physical layout of the Kidd Creek tailings site it was not possible to use the can method to measure this parameter; the value for the Kirkland Gold tailings was assumed to apply to the Kidd Creek site. Detailed measurements would be required to determine the error introduced by this procedure; however, given the range of values for the bulk diffusion

constant it is doubtful if the estimation of the radon flux would be in error by over an order of magnitude. Given the very low value of radon emission from the Kidd Creek tailings the measurement was considered adequate.

The data in Table 4.7 show that the radon fluxes for the Kirkland Lake tailings as calculated by the can and mat methods are in good agreement. The two methods are not totally independent as the calculation of radon flux from the mat method relies on inputting the radon diffusion constant which is determined from the can measurements. The radon fluxes for the Kirkland Lake abandoned tailings areas as measured using the Fleischer techniques are on average a factor of 5 less than those predicted from the Schaiger expression. The radon fluxes for the Kidd Creek site are a factor of 50 less than predicted by the Schaiger expression. This is thought to be because the dry beach areas during the measurement period, autumn, do not contain tailings to a depth greater than 0.5 m. This will lower randon emissions. The diffusion constant measured in this study averages 1.4×10^{-6} m^2/s. This value is comparable to those reported in the literature, viz., 2×10^{-7} m^2/s (Tanner, 1964) and 2×10^{-6} m^2/s (Leach, 1982) for soils and 2.2×10^{-10} m^2/s (Leach, 1982) for mud containing 85% water.

4.4.1.2 **Randon emissions from mine ventilation exhausts.** Terradex track-etch detectors were placed in the ventilation exhausts to measure the randon concentrations over a period of one week. Although the detectors are generally used in the environment in relatively calm conditions, there is no reason to believe that the response would be altered by the turbulence in the ventilation shaft. To confirm this, the manufacturer performed a simple test comparing the detector response under calm and turbulent conditions and found no statistically significant difference. Also where feasible at the field sites the detectors were placed in the direct exhaust and in more calm locations within the ventilation system. These measurements also did not demonstrate any effect.

The randon emissions from the ventilation exhausts are summarized in Table 4.8. The radon emissions are low ranging from 110 GBq annually at the Kerr Addison mine to 590 GBq annually from the Wilroy mine. When dispersed in the environment, these low ventilation exhausts emissions represent no significant hazard to the environment.

4.4.2 Particulate Emmissions

4.4.2.1 **Particulate emissions from tailings areas.** The process by which wind resuspends fine partculate matter from the surface of the earth is complicated. A number of factors must be considered when estimating these fugitive emissions. At least

TABLE 4.8 RADON EMISSIONS FROM MINE VENTILATION EXHAUSTS

Mine	Exhaust	Flow (m^3/s)	Rn Concentration (mBq/L)	Error (%)	Radon Emission (GBq/a)
Wilroy					
	No. 1 Shaft	16.5	1020	25	590
			1270	22	
Kerr Addison	312-53 Raise	4.1	9	146	1.9
	Escape Raise	3.5	10	140	1.1
	214-2 Raise	4.9	14	129	2.2
	Manway	4.3	44	109	7.4
	216-65 Raise	2.6	68	106	7.4
	321-65 Raise	2.8	140	73	11
	175 W Pass Raise	14.9	140	73	67
				Total	98
Kidd Creek	800 NVR	184	11	149	67
	1600 SVR	154	61	54	210
				Total	277

three different wind erosion equations have been formulated to describe this phenomenon, (Cowherd 1979; Woodruff and Siddoway 1965; Travis 1974). Each model is based on a particular wind erosion problem. The parameters used to define the amount of material liberated from the surface in each model vary. Therefore one should have an understanding of the background of the models before attempting to apply them to new situations.

Woodruff and Siddoway attempted to mathematically describe agricultural field losses. Their formulation includes site specific factors that are not readily adaptable to the tailings situations being considered in this report. Somewhat simpler formulae were developed by the other authors cited. The Travis and Cowherd models are both used in this report to predict fugitive emissions. Travis used mathematical expressions to define the process by which surface soil grains start to move, collide with other particles thereby transfering momentum and cause the particles gaining the momentum. This causes the latter particles to move either horizontally or vertically.

Cowherd derived an expression which satisfies experimental data. A discussion of these models and the application of site specific input parameters is given in Appendix C.

Based on the analysis described in Appendix C, estimations using the Cowherd and Travis models of the radioactive fugitive emissions are calculated. The Cowherd model yields emissions of 1.15×10^{-5} g/m^2-s for the operating tailings and 1.2×10^{-7} g/m^2-s for abandoned tailings areas. One issue not considered in estimating these emissions is the presence of frozen ground or snow cover. In northern Ontario frost or snow cover occurs for over 120 days per year. While frost may occur only overnight, its effect is to reduce the total annual emissions. Snow cover on the surface will stop the fugitive emissions. Accordingly the total annual emissions should be reduced by 33% to account for these effects. Thus the predicted total emissions from the Cowherd model are 240 g/m^2-a for operating tailings and 2.7 g/m^2-a for abandoned ones.

After allowing for the effects of snow cover and frozen ground, the Travis model predicts emissions of 3900 g/m^2-a for the operating tailings of 51 g/m^2-a for the abandoned tailings areas. Sensitivity analysis of the Travis model (Table 4.9) demonstrates that the values of the emissions can vary by 2 orders of magnitude depending on the input parameters. Although empirical confirmation is required, the relative lack of sensitivity, particularly for soil size distribution, in the Cowherd model makes it more suitable for this study. In addition, the Travis model, developed for the U.S. southwest to predict the resuspension of fallout materials from the earth's surface, does not provide sufficient compensation for the climate and the physical nature of the tailings under construction. The concentrations of radium-226 and of thorium-230 as presented in Table 4.1 were used with the estimates of fugitive emissions from the Cowherd model to calculate the radioactive particulate emissions. One further correction was made to account for the operating tailings areas being water covered, it is assumed that only 10% of the area contributes to the emissions. The particulate emissions (Table 4.10) are less than 3.7 MBq/a for Ra-226 and less than 11 MBq/a for Th-230.

4.4.2.2 **Particulate emissions from mine ventilation exhausts.** The emission rates were calculated by multiplying the dust concentration in the exhaust by the specific activity radionuclides in the host rock and by the ventilation flow. The operators at the Kidd Creek mine and at the Kerr Addison mine report that the dust concentration in the ventilation exhausts is 1.2 mg/m^3. Data from the Wilroy mine was not available so it was assumed that the dust concentration would be similar to the other mines. The emissions reported in Table 4.11 are less than 1.2 MBq/a for Ra-226 and less than 40 MBq/a for Th-230. Hence they constitute a minor source.

TABLE 4.9 SENSITIVITY TESTS OF THE TRAVIS MODEL

Variable Tested	Test Conditions	Effect on Flux
Percentage of particles Less than 20 um 3% - 12%	0.1% Moisture	Increases 2 orders of magnitude
	10% Moisture	Increases 2 orders of magnitude
Moisture content of soil	Standard	
1% - 17%		Decreases 2 orders
17% - 40%		Decreases by 30%
Surface Roughness	Particle diameter	
Zo 1 cm - 10 cm	100 um	Increases by 7 times
Particle diameter		
60 um - 300 mm	Standard	Decreases 40 times

TABLE 4.10 PARTICULATE EMISSIONS FROM TAILINGS AREAS

Site	Area (ha)	Ra-226 (MBq/a)	Th-230 (MBq/a)
Wilroy	29	0.26	2.7
Kerr Addison	36	0.15	
Kidd Creek	1200	2.6	9.3
Kirkland Lake Abandoned Tailings (Within 3 km of Town)	800	2.4	5.6

TABLE 4.11 PARTICULATE EMISSIONS FROM MINE VENTILATION EXHAUSTS

Site	Dust Concentration (mg/m^3)	Flow (m^3/s)	Ra-226		Th-230	
			(Bq/kg)	(kBq/a)	(Bq/kg)	(kBq/a)
Wilroy	1.2	16.5	37	23	370	230
Kerr Addison	1.2	37.1	19	26	-	-
Kidd Creek	1.2	338	93	1200	320	4000

5.0 EVALUATION OF SOURCE TERMS

The magnitude of source terms can be evaluated by:

(1) Comparison with values specified in regulations, standards and guidelines.
(2) Comparison with the source terms of a nuclear facility, specifically a uranium mining operation.
(3) Comparison with concentrations of radioactivity found in similar natural environments but unaltered by technological changes.
(4) Consideration of the concept of ALARA.

The average concentration of radium-226 was found to be 85 $\pm$ 56 mBq/g in non-uranium solid tailings and 85 $\pm$ 48 mBq/L in liquid effluents from tailings areas and from mine dewatering systems, although it must be identified that there is, as expected, significant variation in the data as demonstrated by the relatively large standard deviations. As baseline radiological data are not available for the non-uranium mining areas in Kirkland Lake and Timmins regions, the source terms must be compared to levels measured elsewhere in order to evaluate the impact of the mining on the radiological environment. The typical concentration of radium-226 in the earth's crust is 30 mBq/g world wide and 41 mBq/g in the United States (Myrick, 1981) and the average concentration of radium-226 in surface waters is 3.7 mBq/L. Perhaps a more appropriate basis for comparison is the radionuclide concentrations in surface waters of other areas of northern Ontario, where it is expected that the level of natural radioactivity may be elevated. Water quality data of the Mississagi and Blind River basins as documented in the Elliot Lake Uranium Mines Expansion Environmental Assessment (MacLaren, 1978a) indicates that the average dissolved radium-226 in these waters is <52 mBq/L; the measurements range from <7.4 to 120 mBq/L. Also the average concentration in Dunlop Lake, a body of water traditionally taken as the reference point for evaluating Elliot Lake water quality has been found to be less than 37 mBq/L. Although, in the absence of baseline data, it is not possible to conclude definitively that the levels observed are as a result of non-uranium mining, it appears probable that non-uranium mining operations produce technologically enhanced natural radioactivity particularly with respect to the aquatic environment.

5.1 Regulations, Standards and Guidelines

The regulations, standards and guidelines which prescribe limits on the radiation exposure of the public and on radioactive releases are summarized in Table 5.1.

On average, liquid effluent concentration limits will not be exceeded due to discharges from the non-uranium mining sites. As shown in Figure 4.2 there is less than 5% probability that sample will have a radium-226 concentration above the 370 mBq/L federal standard prescribed by the Metal Mining Liquid Effluent Regulations and Standards (DFE, 1977). In fact, in none of the samples analyzed was the concentration of radium-226 above 370 mBq/L. Also, there is only a 20% probability that any sample will have a radium-226 concentration in excess of 110 mBq/L. Also, in view of the relatively low levels of radioactivity observed in the effluents from non-uranium mine sites, changes in waste management practices to satisfy ALARA would not appear to be warranted.

TABLE 5.1 RADIATION STANDARDS AND GUIDELINES FOR MEMBERS OF THE PUBLIC

Radiation/Radioactivity	Limit		Agency
Internal radioactivity or penetrating radiation	Whole Body	5 mSv/a	AECB (1974)
	Gonads	5 mSv/a	
	Bone Marrow	5 mSv/a	
Radon Daughters from nuclear fuel facilities		1.4×10^{-3} J.h.m^{-3}	AECB (1978)
Clean-up criteria		4.1×10^{-6} Jm^{-3}	
Dissolved Ra-226 in effluents	Maximum monthly mean	0.37 Bq/L	Can. DFE (1977)
	Maximum composite	0.74 Bq/L	
	Maximum grab sample	1.1 Bq/L	
Ra-226 in drinking waters	Objective	0.1 Bq/L	Ont. MOE (1978)
Ra-226 in drinking waters (Total)	Maximum permitted	1 Bq/L	Can. NH&W (1978)
	Target	0.1 Bq/L	
Ra-226 in receiving waters (Total)	Objective	1 Bq/L	Ont. MOE (1982)

Many national and international agencies such as the Atomic Energy Control Board, the Ontario Ministry of the Environment are actively working on the concept of de Minimis dose and de Minimis levels. The radionuclide content of the waste from non-uranium mining provides an additional perspective upon which to establish levels for nuclear fuel facilities.

5.2 Comparison of Source Terms from Uranium and Non-uranium Mining

Tables 5.2 and 5.3 list the source terms for air emissions and releases to water respectively for uranium mines and mining camps in Ontario and Saskatchewan. Available data for all major uranium mine sites both currently operational or shut down are summarized in order to make the comparison between non-uranium and uranium mine sites as comprehensive as possible.

TABLE 5.2 EMISSIONS TO ATMOSPHERE FROM URANIUM MINE/MILL SITES IN SASKATCHEWAN AND ONTARIO*

Mine/Emission	Mining Operations	Milling Operations	Tailings Management
Beaverlodge--(a)+			
Uranium	6.3E-1--(b)	9.6E+0--(b)	
Ra-226	3.2E-1	2.0E+0	
Th-230	3.2E-1	2.0E+0	
Pb-210	3.2E-1	2.0E+0	
Rn-222	8.8E+4	1.3E+4	6.7E+3--(c)
Rabbit Lake--(a)			
Uranium	neg.--(d)	3.3E+1	
Ra-226	neg.	1.3E+1	
Th-230	neg.	1.3E+1	
Pb-210	neg.	1.3E+1	
Rn-222	4.1E+4	2.9E+4	8.8E+4
Cluff Lake			
Uranium	neg.	3.1E+1	
Ra-226	neg.	4.1E-1	
Th-230	neg.	4.1E-1	
Pb-210	neg.	4.1E-1	
Rn-222	2.6E+4	5.2E+4	1.3E+5
Agnew Lake			
Rn-222	1.9E+4--(g)		
Denison--(f)			
Uranium	0.74--3.7E+0	0.37--1.9E+2	
Rn-222	4.1E+5		2.1E+5
Rio Algom--(f)			
Rn-222	6.3E+4	1.9E+4	7.8E+5

* Emissions are expressed in GBq/a
neg.: negligible

\+ For footnotes, see page 54
6.3E-1 = 6.3×10^{-1}

TABLE 5.3 RELEASES TO WATER FROM URANIUM MINE/MILL SITES IN SASKATCHEWAN AND ONTARIO*

Mine/ Release	Mine Water		Tailings Decant		
	Flow (m^3/a)	Total	Flow (m^3/a)	Dissolved	Total
Beaverlodge--(a)+	3.3E+6		1.1E+7--(g) 2.6E+6--(h)		
Uranium		2.1E+2--(b)		2.7E+2	2.7E+2
				2.1E+2	2.1E+2
Ra-226		8.1E-1		2.8E+0	8.5E+0
				3.6E-1	9.6E-1
Th-230		8.1E-1		2.8E+0	8.5E+0
Pb-210		8.1E-1		2.8E+0	8.5E+0
Rabbit Lake--(a)	4.8E+5		1.9E+6 2.2E+6--(h)		
Uranium		4.8E+0		8.5E+1	8.5E+1--(h)
Ra-226		8.5E-2		3.7E-1	2.4E-2
				1.4E-1	2.1E-1
Th-230		8.5E-2		1.4E-1	2.1E-1
Pb-210		8.5E-2		1.4E-1	2.1E-1
Cluff Lake			2.0E+6 1.5E+6--(h)		
Uranium					4.4E-1
Ra-226				7.4E-1	3.0E+0
Th-230				7.4E-1	3.0E+0
Pb-210				7.4E-1	3.0E+0
Agnew Lake--(j)	1.4E+6--(i)		2.7E+5		
Ra-226		1.9E+0		7.0E-3	7.0E-3
Denison--(f)	1.1E+6		8.E+6		
Uranium		5.6E+2		7.4E+0	
Ra-226		3.1E+1		8.1E-1	7.0E+0--(k)
Th-230				3.0E+0	
Pb-210				7.4E+0	
Th-228, 232				1.5E+0	

* Releases are expressed in GBq/a

+ For footnotes, see page 54

3.3E+6 = $3.3+10^6$

TABLE 5.3 RELEASES TO WATER FROM URANIUM MINE/MILL SITES IN SASKATCHEWAN AND ONTARIO* (CONT'D)

Mine/ Release	Mine Water		Tailings Decant		
	Flow (m^3/a)	Total	Flow (m^3/a)	Dissolved	Total
Rio Algom--(f)			6.0E+6		
Ra-226		3.0E+1		5.9E-1--(k)	
Th-230				7.4E-1	
Th-228, 232				1.1E+0	
Madawaska--(k)				1.7E-1	1.1E+1
a-226					

* Releases are expressed in GBq/a
3.3E+6 = 3.3×10^6

FOOTNOTES FOR TABLES 5.2 & 5.3

a) Taken or derived from Beak, 1980.

b) Uranium-238 daughters assumed to be in equilibrium.

c) Radon flux calculated from modified Schaiger expression. Constant equals 0.65 for summer and 0.065 for winter.

d) Calculated from radon daughter measurements and ventilation rates as reported in Agnew Lake Annual Reports, 1978, 1979, 1980.

f) Taken or derived from MacLaren, 1977, except for Ra-226 data for liquid effluents which were taken from reference (k).

g) Total flows of Ace Creek and Tailings Creek.

h) Personal Communication; Atomic Energy Control Board.

i) Mine water discharged to tailings area.

j) Taken from Agnew Lake Annual Reports, 1978, 1979, 1980.

k) Average annual releases in period 1979 - 1981. Personal communication, Atomic Energy Control Board, Waste Management Division.

Atmospheric and aquatic source terms are given for the mining operations, milling processes and tailings management areas. Available data are incomplete particularly with respect to nuclides other than radium-226. The comparison of the source terms from uranium mines with those developed in this work for non-uranium mines must be interpreted carefully as the measurements and analyses were performed by different laboratories and the methodologies for calculating the source terms from the sampling data may differ. However, it is judged that the derivation of the information is of sufficient consistency to permit a first order comparison.

The major source terms for a uranium mine/mill site are discussed very briefly below:

Mining Operations: Atmospheric emissions from the ventilation exhausts originate primarily from blasting, ore handling and where practiced underground ore crushing. The main emissions are radon gas and particulate radioactivity. Aquatic releases arise almost entirely from the mine water discharge. Generally, the mine water is directed to the mill or tailings area where it is treated with barium chloride, however, at some locations it is treated directly with barium chloride before release to the environment.

Milling Operations: The major sources of radon are the crushing and grinding stages while uranium and polonium-210 are emitted from the drying and packaging stages. The liquified sludge from the concentrator stages are directed to the tailings areas.

Waste Management Operations: Radon emissions from tailings areas generally result in significant radon source terms. Fugitive dust incorporating particulate radionuclides are also released to the atmosphere. Liquid releases arise from the treated decant overflow and the seepage through dykes. Generally, however, the flow from the seepage is small in comparison to the decant flow.

Originally, at the Beaverlodge site and in the Elliot Lake area waste management practices were not as stringent as currently exercised and the magnitudes of the current releases reflect this. Also, it is not always clear where to designate the point at which to calculate the discharge to the environment and hence to specify the particular source term. In this report the final control point was considered to be the location at which to specify the source term. The environmental loading for a mining company, e.g. Rio Algom was calculated as the sum of the releases from each individual site belonging to the company in the area.

The comparison of source terms from uranium and non-uranium facilities is not straight forward. The reason for the difficulty is that the magnitude of a source term is dependent on many factors such as the size of a facility and, in the case of liquid effluents, the design of the tailings area and the effluent flow. Accordingly, in this report, the comparison is performed in three ways. In Table 5.4, the collective source terms of the Kirkland Lake area facilities are compared, with the exception of Elliot Lake, to the source terms of the individual uranium mines in Ontario and Saskatchewan. As the Elliot Lake facilities are situated within a 50 km radius, the comparison with Elliot Lake are made on an area basis. From Table 5.4, it can be observed that radon emissions from the Kirkland Lake area facilities are only a few percent of the nuclear facilities. The exceptions to this are the radon emissions from the Beaverlodge and Rabbit Lake tailings, where the emissions are of the same order as the Kirkland Lake emissions.

As identified in section 4.4.1, the direct measurements of the radon fluxes from the Kirkland Lake tailings areas indicated that the Schaiger relationship predicted radon emissions were a factor of 5 higher. However, as the radon fluxes from the uranium mine tailings were calculated using the Schaiger relationship, the comparison was made on this basis. Table 5.4 demonstrates that the annual loading of the aquatic environment of radium-226 from the Kirkland Lake non-uranium effluents ranges from approximately 1% of the Beaverlodge and Elliot Lake releases to values similar to those of the Madawaska and Agnew Lake mines. The data for Th-230 suggest that the ratio of the loading from the Kirkland Lake facilities to the Uranium mining facilities is larger than the ratio for Ra-226. Any inference from this observation must be made with caution as the data for Th-230 is not as complete as that for Ra-226 for both the uranium and non-uranium facilities.

The comparison (Table 5.5) of the source terms of the Kidd Creek mine sites with the uranium facilities is analogous to that in Table 5.4. At Kidd Creek the radon emissions to air are 2% or less of the nuclear facilities emissions. The exception to this is Beaverlodge where the emission of radon is only a factor of 5 greater than that from Kidd Creek. The loading of the aquatic environment from Ra-226 in the liquid effluents ranges from approximately 5% of the Elliot Lake cumulative loading, to a factor of 46 larger than that from Agnew Lake. It should be noted that the Agnew Lake heap leaching process produces minimal liquid effluents. The process is unique in Canada and therefore the Agnew Lake data cannot be taken to pertain to uranium mills in general.

TABLE 5.4 COMPARISON OF SOURCE TERMS OF THE KIRKLAND LAKE MINING REGION TO URANIUM MINING REGIONS/SITES*

	Airborne Emissions		Liquid Effluent Releases			
			Mine Water		Tailings Decant	
	Mine Exhausts (Rn-222)	Tailings (Rn-222)	(Ra-226)	(Th-230)	(Ra-116)	(Th-230)
Elliot Lake	0.002	0.012	N/A		0.017	0.006
Madawaska	-	-	N/A		0.13	-
Agnew Lake	0.037				3.3	-
Beaverlodge	0.008	1.8	0.12	0.64	0.008	0.076
Cluff Lake	0.026	0.089	-	-	0.03	0.3
Rabbit Lake	0.017	0.13	1.1	6.5	0.063	1.6

* The values shown in this Table are the ratios of annual releases from Kirkland Lake region to that of uranium mining regions.

N/A: not available

TABLE 5.5 COMPARISON OF SOURCE TERMS OF THE KIDD CREEK SITE TO URANIUM MINING REGIONS/SITES*

	Airborne Emissions		Liquid Effluent Releases			
			Mine Water		Tailings Decant	
	Mine (Rn-222)	Tailings (Rn-222)	(Ra-226)	(Th-230)	(Ra-226)	(Th-230)
Elliot Lake	0.0006	0.0015	-	-	0.049	0.95
Madawaska	-	-	-	-	1.9	-
Agnew Lake	0.015	-	-	-	46	
Beaverlodge	0.003	0.22	0.27	0.77	0.11	0.81
Cluff Lake	0.011	0.011	-	-	0.44	4.7
Rabbit Lake	0.007	0.017	2.6	7.4	0.88	26

* The values shown in this Table are the ratios of annual releases from Kidd Creek site to that of uranium mining regions.

It is also instructive to compare the concentrations of radionuclides in uranium and non-uranium effluents. From Table 5.6, it may be observed that the Ra-226 concentrations tend to fall near the low end of the range of concentrations in nuclear facility effluents. However, the range of concentrations from uranium and non-uranium facilities are not too dissimilar.

TABLE 5.6 CONCENTRATIONS OF DISSOLVED RADIONUCLIDES IN URANIUM AND NON-URANIUM MINE LIQUID EFFLUENTS

	Mine Water (a)		Tailings Decant	
	Ra-226 (mBq/L)	Th-230 (mBq/L)	Ra-226 (mBq/L)	Th-230 (mBq/L)
Beaverlodge	300		260	
Rabbit Lake	110-190		110	
Cluff Lake			300 (total)	
Agnew Lake			19-52	
Madawaska			70	
Dennison			19-440 (b)	
Rio Algom			37-1500 (b)	11-190 (c)
Kidd Creek	11-220	370	26	280
Kirkland Lake Mines	22-63	380	26-33	240-270

(a) Values sited only when mine water released directly to environment.
(b) Most results indicate low concentrations < 190 mBq/L.
(c) Most results less than 110 mBq/L.

Table 5.7 compares the annual loading of radionuclides from the Kidd Creek sites to individual facilities in Elliot Lake from 1979 to 1981 (AECB, 1983), when current stringent waste management procedures are being practiced. From this table, we can observe that the Kidd Creek site annual release of Ra-226 ranges from approximately 20% of that of the Quirke mine to a factor of 8 larger than the Williams Lake and Stanrock facilities. The average is 0.8. From the data in Tables 5.4 to 5.7, we may infer that the loadings of the aquatic environment from both uranium and non-uranium mining sites are generally low and that the effectiveness of the waste management practices in uranium mines may be demonstrated by the comparison between the aquatic source terms for uranium and non-uranium mining facilities.

TABLE 5.7 LIQUID RELEASES OF RADIUM-226 FROM ELLIOT LAKE URANIUM FACILITIES AND RATIO OF RELEASES TO THOSE FROM KIDD CREEK MINES

Facility	Dissolved Ra-226 (mBq/L)	Annual Loading (MBq/a)	Ratio of Releases Elliot Lake/Kidd Creek
Long Lake	81	170	0.32
Williams Lake	210	70	0.13
Stanrock	30	63	0.12
Panel	120	340	0.62
Quirke	230	2300	4.2
Stanleigh	310	2800	5.1
Nordic	100	210	0.39
Pronto	160	370	0.68

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APPENDIX A

RADIOACTIVITY IN NON-URANIUM METAL MINES

The hazard to non-uranium miners from the inhalation of radon daughters has been recognized over the last 15-20 years. This awareness has resulted in radon daughter surveys being performed in many countries. In Canada, there were high levels recorded in the fluorspar mine in St. Lawrence, Newfoundland, where only 5 samples out of 80 showed levels less than 6×10^{-6} Jm^{-3} (British Columbia Royal Commission, 1980). High levels of radon daughters have also been found in a British Columbia silver/lead/zinc operation. A survey in a British tin mine in 1972 revealed radon daughter concentrations ranging from $(0.25 \text{ to } 5) \times 10^{-4}$ Jm^{-3} (Dungey, 1981). A recent survey of selected British mines determined that the radon daughter concentrations in 21% of accessible locations were greater than 6×10^{-6} Jm^{-3} (O'Riordon, 1981). Elevated radon daughter levels also have been reported in Swedish (Snihs, 1978), Italian (Sciochetti, 1981), and Polish mines (Domanski, 1981). Measurements by the U.S. Mine Enforcement and Safety Administration summarized by the U.S. Environmental Protection Agency (Bliss, 1978) show that the maximum radon daughter concentrations observed in underground mines ranged from 4×10^{-6} Jm^{-3} in gold mines to 1.9×10^{-5} Jm^{-3} in antimony mines (Table A.1). Radon emissions can be estimated from this data from the ventilation exhaust flow and selecting an appropriate equilibrium factor (Andrews, 1980). Equilibrium factors of 0.4-0.5 are consistent with measured values in non-uranium mines (Berteig, 1981). Radon emissions from selected non-uranium mines in the United States measured directly by Andrews (1980, 1981) range from negligible for open pit mines to 8.5 TBq/a for a zinc mine (Table A.2).

There are few data on the release of radioactivity from non-uranium mines into the aquatic environment, and it was not possible to find references to any Canadian data. In Brazil discharges from an iron niobium mine had a radium-226 concentration of 1.8 Bq/L (Paschoa, 1981). In the United States, the U.S. Environmental Protection Agency commissioned a study to evaluate the radioactivity released in non-uranium metal mining liquid discharges (Pedco, 1981). The graph of radium-226 concentrations in the waste waters plotted on cumulative probability paper demonstrate that the points follow a lognormal distribution (Figure A.1). The median concentration is approximately 0.15 Bq/L and 30% of the samples are expected to have radium-226 concentrations in excess of 0.37 Bq/L. The measurements summarized by Bliss (1978) show similar values. The concentrations of radium-226 in the solid and sludge wastes as determined by the Pedco study also follow a lognormal distribution (Figure A.2). The median value is 15 mBq/g and

the 90 percentile value 11 mBq/g. In view of the low levels of radioactivity in the solid wastes, the concentration of radionuclides in the liquid wastes is relatively high.

TABLE A.1 RADON DAUGHTER LEVELS IN U.S. NON-URANIUM METAL MINES

Type of Mine	Number of Observations	Mean Concentration (Jm^{-3})
Antimony	1	1.9E-5
Molybdenum	3	1.7E-5
Copper	2	1.5E-5
Tungsten	2	1.3E-5
Iron	5	1.0E-5
Lead/Zinc	12	5.6E-6
Silver	2	5.0E-6
Gold	4	4.2E-6

Note: The average radon daughter concentration is obtained from mines where at least 2×10^{-6} Jm^{-3} has been detected; the value being the mean of the maximum radon daughter concentration for the particular type of mine.

TABLE A.2 SUMMARY - RADON-222 AND RADIUM-226 RELEASES FROM UNITED STATES MINES

Mine Type[a]	Rn-222 Release		Ra-226 Concentration (mBq/g)		
	MBq/tonne ore	TBq/a	Ore	Concentrate	Tailings
U.G. Limestone	0.03	0.03	15	-	-
U.G. Fluorspar	-	0.07	35[b]	5.6[b]	28[b]
U.G. Copper	0.63	0.21	30	32 (Cu) 13 (Zn) 19 (Fe)	32
O.P. Copper	High background area Not measurable above background	0.07[b]		N.C.[c]	59[b]
U.G. Iron (Magnetite)	Particulate concentrations 4-6 times ambient.	2.6	110	96	230[d]
O.P. Iron (Hematite)	Not measurably above background		13	14	9.3
U.G. Phosphate		3.7	850	-	-
U.G. Fire Clay	3.8	0.59	52	-	-
U.G. Zinc	Particulate Concentrations about twice background	8.5	4.1	2.2	8.1

a) U.G. = Underground; O.P. = Open Pit
b) Ra-226 analyses not completed, U-238 reported
c) No analyses complete yet
d) Waste sand - separated magnetically

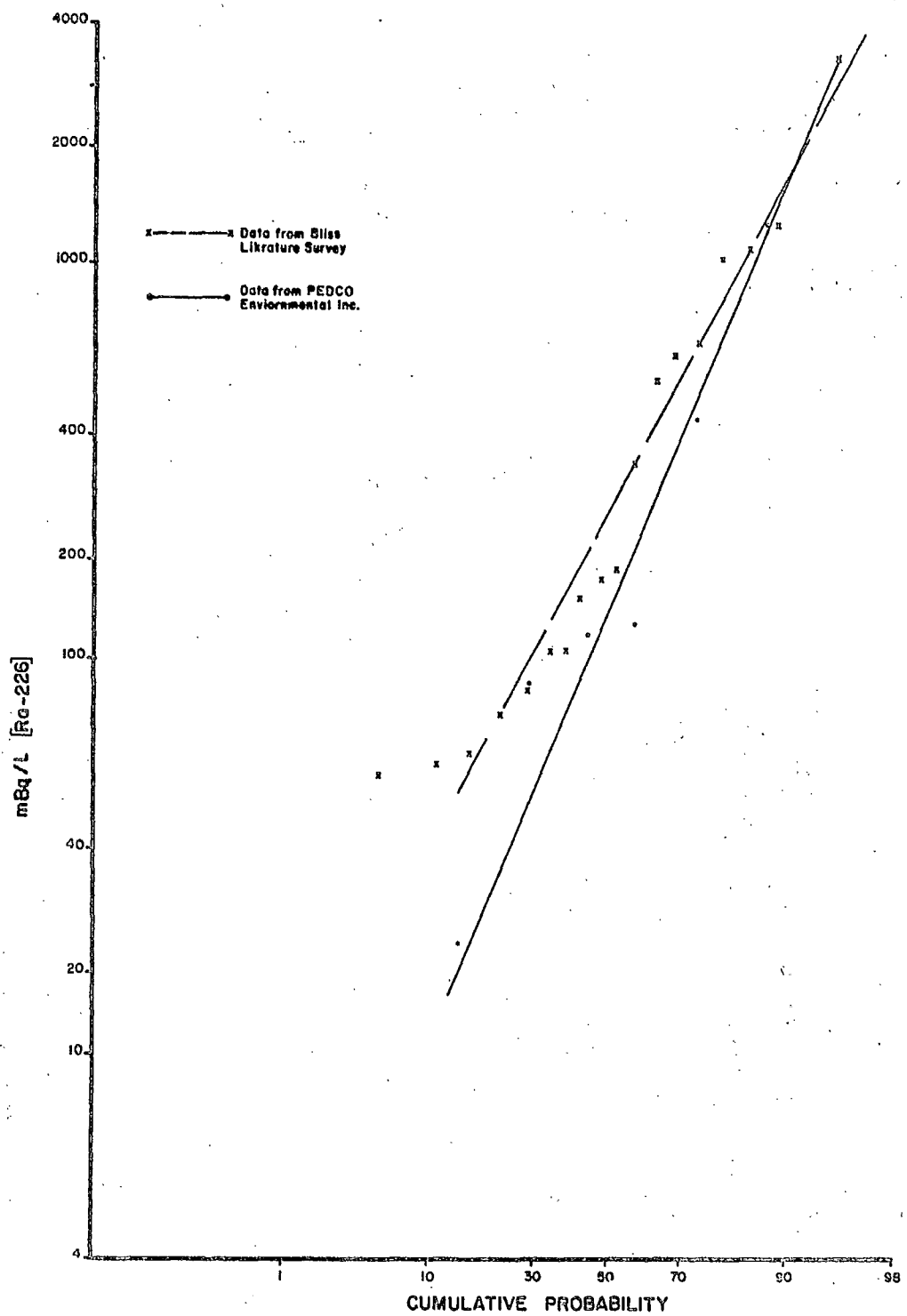

FIGURE A.1 CONCENTRATIONS OF RADIUM-226 IN U.S. NON-URANIUM MINE/MILL LIQUID EFFLUENTS

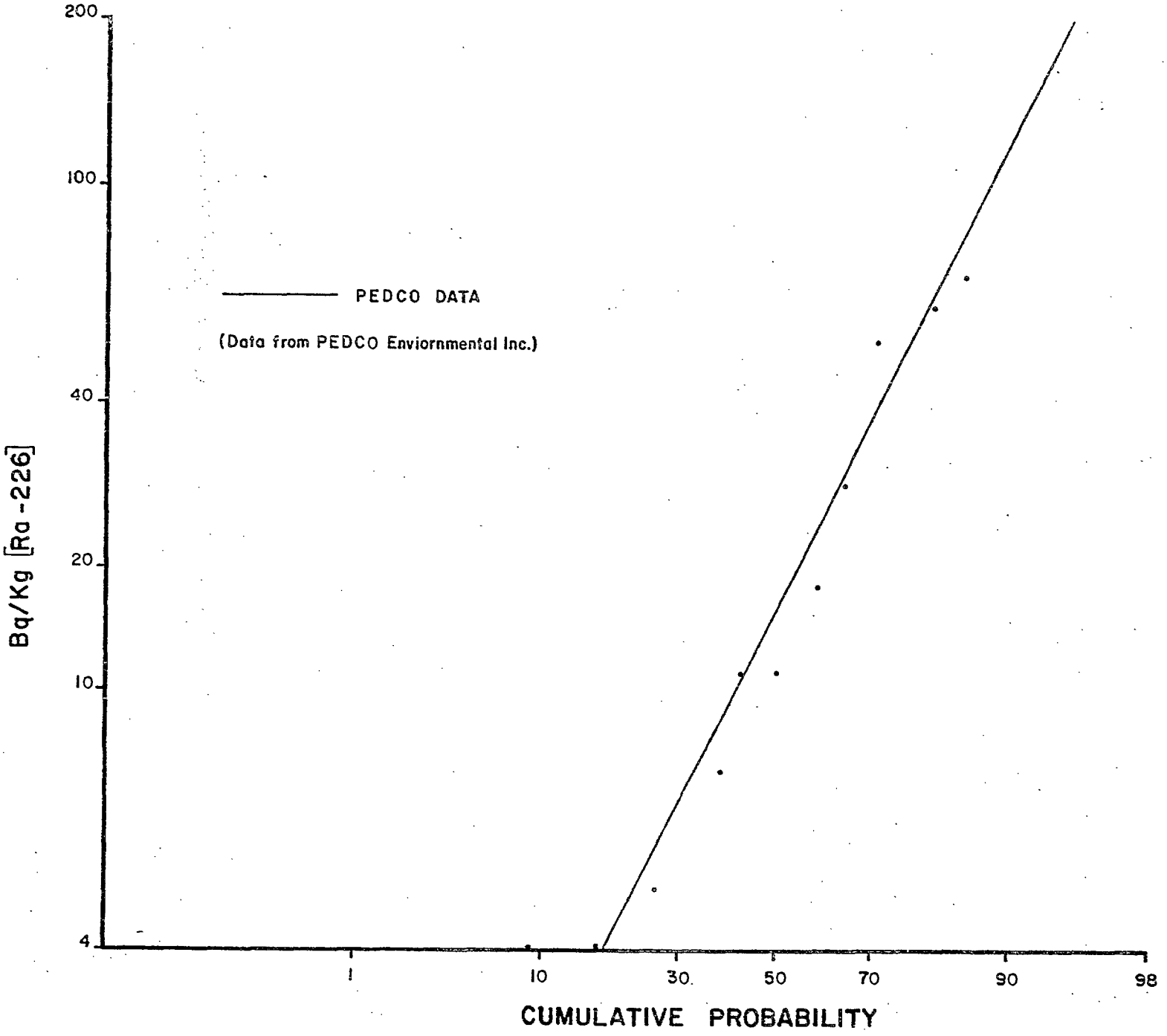

FIGURE A.2 CONCENTRATION OF RADIUM-226 IN U.S. METAL MINE/MILL SOLID WASTES

APPENDIX B

B.0 MINING, MILLING AND WASTE MANAGEMENT OPERATIONS IN THE KIRKLAND LAKE AND TIMMINS MINING REGIONS AND THEIR EFFECT ON RADIONUCLIDE BEHAVIOUR

B.1 Introduction

In this section, the influence of mining and milling operations on the chemical nature and on the behaviour of the radionuclides is discussed. Simplified descriptions of the mining, milling and waste management operations of the precious and base metal mines in the Kirkland Lake and Timmins regions are presented. The chemical and physical behaviour of selected nuclides is discussed in the context of the processes and the waste management environment. As the scope of the study includes both operating and abandoned sites, the differences between the two situations are analyzed, including the influence on potential long term problems.

B.2 Mining Operations

All gold, silver, copper and zinc ores in the Kirkland Lake and Timmins regions are extracted from underground mines. Shafts may reach depths greater than 2000 metres and the crosscuts and drifts cover hundreds of acres in the horizontal plane. There is also a large operating open pit iron mine in the Kirkland Lake camp. The well known Kidd Creek open pit copper-zinc mine near Timmins is no longer operational and the ore is now being extracted from beneath the pit by underground mining.

The techniques used in underground mining depend somewhat on the nature and physical characteristics of the ore body. In the stope the ore is generally blasted away from the walls. Ore separated from the waste rock may undergo preliminary crushing underground with the waste rock being used for stope backfilling or brought to the surface, where it is disposed.

Sources of airborne particulate radioactivity from mining operations arise primarily from drilling, blasting and crushing operations. These dust particles, having the same specific radioactivity as the ore and host rock, are exhausted to the environment by the mine ventilation system. Radon emissions from the mine ventilation exhausts may be significant as:

(1) There would be enhanced radon emanation from the large surface areas in the underground mine.

(2) Dissolved radon in mine water may be released to the mine atmosphere. This has been demonstrated to be a major source of radon in non-uranium mines.

Mine water may also be a significant source of radioactive particulate releases to the aquatic environment. At non-uranium mining studied in the Kirkland Lake and Timmins regions, the mine water is generally discharged to a receiving body of water and not to the tailings ponds. In the mining camps studied no treatment is provided before discharge to the environment.

B.3 Milling Operations

The copper-zinc and gold mills employ a complex physical-chemical process to concentrate the ore which have many unit operations similar to those of the uranium mill. The major non-uranium mill operations are crushing/grinding, flotation, thickening, separation, drying and roasting. As this study involves mainly gold mining, the processes in the gold concentrator will be described and differences in the iron and base metal mills identified. Details of the operations of the Kidd Creek (Pabst, 1981) and the Adams (1981) mills are discussed in the references. A general outline of the physical and chemical processes in the gold concentrators is given in the following sub-section.

B.3.1 Gold Mills. The processes in the Dome Mines Concentrator (Pabst, 1981b) may be considered illustrative of gold mills generally and are shown schematically in Figure B.1. In general, ore brought to the mill is washed and ground to approximately 200 mesh. The ore then undergoes mechanical separation to remove high specific gravity metal from the relatively less dense rock grains. At locations such as Kerr Addison, where the ore contains significant quantities of pyrite, which inhibits the leaching process, the operators roast the ore to remove the sulphur (Kerr Addison, 1980). Slurry from the washing stage is discharged to the tailings areas. In the flotation stage, separation of the sulphides from the gold is achieved. Gold is released from the ore by amalgamation which is followed by the cyanidation process. The chemical equations governing the major processes are given in Figure B.2. During cyanidation, the ore is agitated for a number of hours in large reactors in approximately 1% NaCN or KCN at a pH of about 11. The slurry is then separated by filtration. The pregnant solution with dissolved gold is transferred into a reaction tank where either zinc dust is added to precipitate gold by reduction or it is recovered by electro-winning (Pabst, 1981b). The mines in the Kirkland Lake camp do not use the electrolysis process. Gold sludge recovered from the precipitation phase is separated from the barren solution. The gold is refined to produce gold bullion. The wastes from the cyanidation and precipitation stages are discharged to the tailings areas (Kerr Addison, 1980; Habashi, 1967).

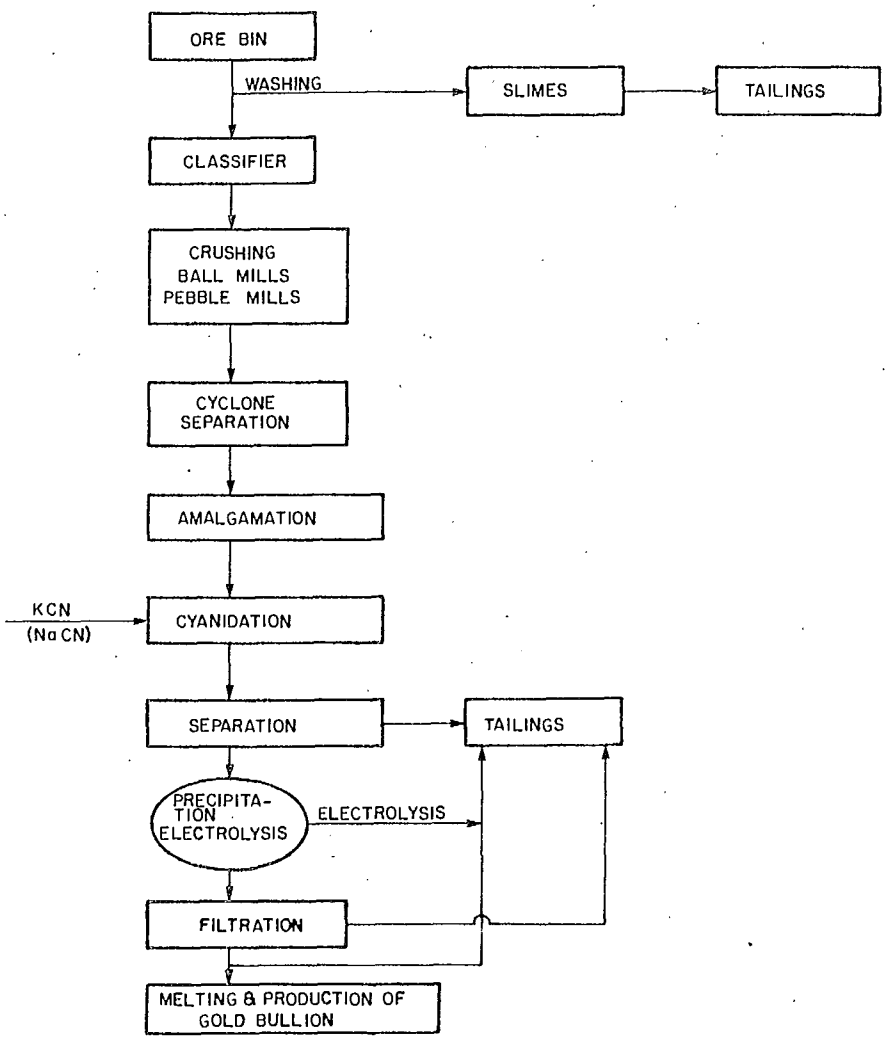

FIGURE B.1 DOME GOLD CONCENTRATOR

1. Amalgamation:

$$Au(Ag) + Hg_x = Au(Ag)\,Hg_x$$

2. Cyanidation: (Au with NaCN)

(a) $2Au(s) + 2H_2O + 4NaCN + O_2 = 2NaAu(CN)_2(l) + 2NaOH(l) + H_2O\ (l)$

(b) $4NaCN(l) + 2Au(s) + H_2O(l) = 2NaAu(CN)_2(l) + 2\ NaOH\ (l)$

3. Reduction: $(NaAu\ (CN)_2)$

$$2NaAu(CN)_2(l) + Zn(s) = 2Au(s) + Na_2Zn(CN)_4(l)$$

4. Electrolysis: e.g. for Au^+

$Au^+(e) + e^- = Au(s)$

(l) = liquid
(s) = solid

5. Roasting:

Process with oxidizing agent to remove sulphur and arsenic.

FIGURE B.2 MAJOR CHEMICAL PROCESSES IN GOLD MILLS

Because of the absence of empirical information and the complexities of the processes, the behaviour of the radionuclides in the mill circuits is uncertain; however, as only gold is recovered it is expected that other elements would be discarded. The amalgamation process does not involve uranium, thorium or radium. During cyanidation, because of the strong complexing properties of the cyanide ions and the uranium ion, a uranium cyanide based complex may be formed. Sodium hydroxide as a product of the process will tend to increase the pH and to hydrolyze the thorium ions with subsequent precipitation. Some operators add lime to the mill process. The sulphides in the ore when released to the tailings may be oxidized to sulphates. The lime will partly precipitate as calcium sulphate and radium can be expected to coprecipitate with the calcium sulphate.

As in the uranium mill operation, radon releases should be principally from the ore grinding and crushing stages. Particulate emissions should be low as there are generally dust collectors over the grinding and crushing equipment. Airborne releases from the intermediate amalgamation, cyanidation and precipitation stages should be minimal.

B.4 Waste Management Operations and Their Effect on Radionuclide Behaviour

B.4.1 Characteristics of Waste Management Areas. Radionuclide behaviour in a tailings area is complex and difficult to model. The kinetics of the various reactions are governed by the physical and chemical environment. The factors that are prominent in influencing radionuclide behaviour are presented in Table B.1. The analytical results for a number of elements in selected Ontario uranium and non-uranium tailings areas are summarized in Table B.2 (Hawley, 1979). Table B.2A lists the elements which are found in greater abundance than in the earth's crust while Table B.2B lists the elements that are depleted. The elevated concentrations of selected elements are not surprising given the mineralogy of the ores. Generally very high concentrations are found for sulphur and arsenic. Mercury content is higher in the gold camp due to the use of this element in the milling process. The concentrations of sulphur, vanadium, lead, calcium, barium and phosphorus are similar in the uranium and non-uranium mine tailings areas, whereas arsenic, sodium, potassium, and fluorine are more abundant in the uranium camp.

B.4.2 Behaviour of Radionuclides in Tailings Areas.

Uranium

In nature, uranium occurs in oxidation states III, IV, V or VI typically as cumulative uranium oxide or as a complex with other metals. Only oxidation states IV and V are relatively stable in aqueous solutions as lower states are rapidly oxidized to the higher forms. U-VI is the thermodynamically more stable form. It is the final oxidation product in the hydrochemical cycle in nature. Equations which illustrate the oxidation/reduction processes are as follows (Langmuir, 1978):

$$UO_2(s) + H_2O(l) + 1/2\ O_2(g) = UO_2^{2+} + 2OH^-$$

or

$$U^{4+} + 2H_2O = UO_2^{2+} + 4H^+ + 2e^-$$

The presence of oxygen from the atmosphere or dissolved in the tailings liquid shifts the balance of the equations to the right. Hence the predominance of U-VI in solid surface deposits and liquids is expected at most locations. However, in the presence of organic matter, either pure or associated with sulphides, as occurs in ponds near wooded lands, the carbon would act as a reducing agent on uranyl ions converting them to insoluble uranous ions (Bliss, 1978). This process may be shown as follows:

TABLE B.1 FACTORS INFLUENCING RADIONUCLIDE BEHAVIOUR IN TAILINGS AREAS

Factor	Effect	Comments
pH	- Reduced stability of non-sulphur compounds - Enhanced mobility of uranium, radium and thorium - favourable conditions for radionuclide reaction	- Operators indicate that pH is about 7. Study by Scott Envir. Canada (1975) states that the pH of base metal untreated operating tailings is 4 and that of abandoned ones, less than 3.
Size	- Leachability and solubility of radionuclides increased with finer sizes	- Slimes which are treated ore materials are approx. 200 mesh.
Climate	- Solubility of radionuclides in liquid phase reduced in low temperatures	- Higher levels of radioactivity in effluent streams have been observed for uranium mines during spring breakup.
Effluent Treatment	- Efficiency of radionuclide removal low from lime addition in effluent - lime added in mill circuit will tend to concentrate radioactivity in tailings areas	- Some non-uranium operators add lime to neutralize liquid overflow. Generally, no settling pond following lime addition station. Barium chloride addition not performed.
Long Term Weathering	- Long term enhancement of radium and thorium mobility	- Buffering from chemical additions in operating sites is gradually reduced and tailings areas containing significant quantities of acidifying elements such as sulphur.

$$4UO_2(CO_3)_3^{4-} + HS^- + 15H^+ = 4UO_2(s) + SO_4^{2-} + 8H_2(g) + 12CO_2(g)$$

(Hydrogen sulphide is formed during the decay process of organic materials.)

A redox reaction takes place in the presence of ferric cations:

$$2Fe^{2+} + UO_2(CO_3)_3^{4-} + 3H_2O = UO_2(s) + 2Fe(OH)_3(s) + CO_2$$

TABLE B.2 PROPORTIONS OF SELECTED CHEMICALS IN ONTARIO TAILINGS

A) Elements Generally More Abundant in Tailings Than in Earth Crust*

Element	Ontario Average	Gold Camp		Base Metals	Iron Camp	
		Timmins	Kirkland Lake	Timmins	Kirkland Lake	Bancroft
Sulphur	57	50	59	181	48	25
Arsenic	28	15	2.2-58.0	6.2	1.8	205
Iron	17.5	10	9	30	9.0	0.72
Vanadium	1.5	1.1	5.5	0.7	N.A.	0.9
Mercury	8	2	13	3.3	0.6	0.5
Lead	0.5	up to 160	1.8	3.4	1.1	5.5

* Values shown are the ratios of content in Designated Area to average content of Earth Crust.

B) Elements Generally Depleted in Tailings Compared to Earth Crust*

Element	Ontario Average	Gold Camp		Base Metals	Iron Camp	
		Timmins	Kirkland Lake	Timmins	Kirkland Lake	Bancroft
Sodium	49	18	63.0	8	N.A.	110
Potassium	57	48	26 to 16 000	31	N.A.	122
Calcium	11	15	8.9	8	N.A.	5
Fluorine	42	38	48.8	36	20.0	207
Barium	6	3	3.0	8	7.0	5
Phosphorus	70	44	91.0	21	N.A.	60

* Values shown are percentage of content in Designated Areas based on average content of Earth Crust.

Source: Hawley, 1979.

Uranium oxides form complexes with a variety of ligands such as hydroxides, fluorides, sulphates, silicates and cyanides. The formation of a particular complex compound is related to the specific ligand concentration, pH, temperature and the oxidation state. In the tailings ponds, uranium will tend to oxidize to the uranyl ion from bacterial and atmospheric actions. If the tailings are treated with lime, the pH of the material will be neutral to alkaline. Langmuir (1978) has reported that based on solubility and sorption data, the uranyl ion is least mobile in the pH range of 5-8.5. Langmuir's work based on computer modelling has recently been extended to other nuclides such as radium and to tailings ponds environment in the United States (Langmuir, 1982).

Uranium behaviour may be related to the type of adsorbent material and to the pH of the solution (Ames, 1978; Langmuir, 1978). If the solution in the tailings pond is in the neutral range and the bottom consists of clay and organic sediments, a significant fraction of uranium in the form of UO_2^{2+} ions will desorb from the clay adsorbent bed into solution (Langmuir, 1978) with eventual discharge to the environment. Temperature also influences the adsorption process as solubility increases with temperature. The sludge discharged from the mill is typically at 20°C. The lower tailings pond water temperature, particularly in winter, will tend to increase the adsorption to the clay and organic materials.

As non-operating, non-uranium tailings areas receive no neutralizing treatment, the pH tends to be low, and the U-IV ion is present along with the U-VI ion. Separation of the two forms will take place as the relatively soluble U-VI will be carried downstream while the U-IV will be hydrolyzed and/or adsorbed on to adsorbent grains such as ferric hydroxide which remain in the vicinity of the source (Dyck, 1978).

Thorium

Dyck (1978) has estimated that the earth's crust contains approximately three times as much thorium as uranium. Aerial surveys in the regions of interest have measured the thorium:uranium ratio to be approximately 2.5:1 (Ont. MNR, 1977). In nature thorium does not occur as a free element but mostly as the mineral monazite.

The behaviour of thorium in the local environment depends on the following factors:

(1) pH
(2) the specific ligands present and their concentration
(3) temperature
(4) the distribution of particle sizes of the tailings material

Thorium in solution produces an ion which is highly complexing. Ligands which form complexes include fluorides, chlorides, nitrates, and sulphates. Soluble compounds in the tailings pond are thorium chlorides, thorium nitrates and thorium sulphates.

Because of weathering conditions in the regions of interest, the pH of the tailings material strongly influence the behaviour of the compounds associated with thorium cations. Generally, the concentration of dissolved thorium decreases with increasing pH up to a pH value of 5. Above this value the concentration is not significantly affected due to the formation of $Th(OH)_4$ (Ames, 1978). Thorium compounds tend to adsorb on to clay particles. Three fundamental mechanisms of adsorption are:

(1) Thorium hydroxide precipitation as a result of soil calcite buffering
(2) Adsorption of dilute thorium solutions on clay soils at pH greater than 2
(3) Thorium adsorption on organic soils increases with pH above 5.

Thorium can become a long term hazard. In the non-operating tailings areas as the materials become more acidic, the thorium will tend to be transferred into solution and to be mobilized as Th^{4+}.

Radium

The two main isotypes of radium are Ra-226 and Ra-228 in the uranium and thorium series. Radium occurs in nature as Ra^{2+} typically crystallographically bound to bed rock material. It may also be found a highly insoluble compounds such as $RaSO_4$ and as coprecipitates such as $Ca(Ra)SO_4$ or $Ba(Ra)SO_4$.

Radium compounds for which, thermodynamic data exist, are $RaCl_2$, $Ra(NO_3)_2$, RaI_2 and $RaSO_4$. These are with the exception of radium sulphate considered soluble. In the tailings ponds it is expected that the radium will primarily be in the form of radium sulphate although some radium nitrate might be present as a result of the cyanide ion degredation and oxidation arising from the gold milling process.

Chemically, radium is a homolog of the alkaline-earth group elements and demonstrates little tendency to form complexes. In aqueous environments such as underground mine waters with relatively elevated Ca^{2+}, Mg^{2+} and high partial pressure of carbon dioxide the Ra^{2+} tends to remain in soluble form. However, once the water is brought to the surface and the CO_2 is purged, the radium tends to precipitate as $Ca(Ra)CO_3$ or $Ca(Ra)CO_3$ and/or adsorb on to the surrounding porous material, with the calcium and magnesium ions precipitating or coprecipitating in the same process (Dyck, 1978).

APPENDIX C

THE TRAVIS AND COWHERD FUGITIVE EMISSION MODELS AND THE ESTIMATION OF PARTICULATE EMISSIONS FROM TAILINGS AREAS

C.1 Travis Model

This model is discussed in NUREG-0706 (NRC, 1980) where it is used as the basis for the UDAD wind erosion calculations. Four equations (Table C.1) are used to develop an expression for the vertical flux from the surface. This vertical flux is by definition only concerned with the amount of material smaller than 20 um in diameter.

The four equations define:

- the shear velocity at the surface in (m/s)
- the threshold velocity for a saltation particle diameter of a specific size in (m/s)
- the horizontal flux in (g/m.s), and
- the vertical flux in (g/m^2-s).

The wind velocity profile has a logarithmic form. As the surface roughness increases so does the shear velocity.

The threshold velocity equation is defined in terms of:

- percent moisture
- saltation particle diameter (cm) and
- particle density (g/cm^3)

It provides a minimum velocity for saltation or the initial movement of particles across the surface.

The horizontal flux equation relates the shear velocity and the threshold velocity in an expression that dictates no erosion will occur if the shear velocity is less than the threshold velocity.

The vertical flux equation is a function of:

- shear velocity
- threshold velocity
- horizontal flux, and
- proportion of material less than 20 um.

While these equations account for many of the meteorological factors. The power relationship of P, the portion of fine material <20 um, produces significant variations in the quantity of material that is eroded if considered for different tailings

Shear Velocity

$$U_t = U^* \, 2.5 \, \ln \frac{Z}{Z_o} \qquad (1)$$

U_t = wind velocity at height t(m/s)
U^* = shear velocity (m/s)
Z_o = surface roughness (cm)
Z = height (cm)

Threshold Velocity

$$U_t^* = 0.1 \sqrt{\frac{P_s - P \, gd}{P}} \, (1.8 + 0.6 \log_{10} W) \qquad (2)$$

U_t^* = threshold velocity (m/s)
d = diameter of saltation particle (cm)
p = air density (g/cm^3)
p_s = particle density (g/cm^3)
g = gravitational constant (cm/s^2)
W = percent moisture (%)

Horizontal Flux

$$q_h = 10^2 \, U^{*2} \, (U^* - U_t^*) \qquad (3)$$

q_h = horizontal flux (g/m.s)

Vertical Flux

$$q_v = 2 \times 10^{-8} \, (q_h / U^{*3}_t) \, ((U^*/U^*_t)^{P/3} - 1) \qquad (4)$$

q_v = vertical flux ($g/m^2.s$)
p = mass percentage of particles of less than 20 um in diameter

TABLE C.1 EQUATIONS USED IN THE TRAVIS MODEL

masses. The particle size distribution of tailings can vary between 3% and 90% fine material. With dry material (0.1% moisture) a shift from 3% to 24% fines can change the flux rates from 10^{-4} to 10 g/m^2-s. This increase of emissions to 10 000 times higher levels may not be realistic.

C.2 Cowherd Model

This model was formulated from a fit of experimental data generated for representative open sources. The equation is:

$$E = 0.38\ (e/1123)\ (s/15)\ (f/25)\ (PE/50)^2$$

where:

E is the emission rate (kg/m^2-a)
e is the soil erodibility (t/km^2-a)
s is the percentage of material with diameter <74 um
f is the frequency of wind >5.3 m/s at 0.3 m above ground
PE is the Thornthwaites precipitation-evaporation index

It can be seen from this equation that physical data on the tails is required. The soil erodibility can be assessed from a sieve analysis. Woodruff and Siddoway (1965) provide a relationship between sieve analysis and erodibility on the basis of the percentage of material greater than 0.84 cm in size. The information is summarized in Table C.2. The climatic factors can be obtained from historical records. Wind speed can be adjusted using the logarithmic profile.

TABLE C.2 SOIL ERODIBILITY "e"

Percentage of Dry Soil Fractions > 0.84 cm	e = Tons/acre	e = t/ha
1	310	694
10	134	300
20	98	220
30	74	166
40	56	126
50	38	85
60	21	47
70	12	27
80	2	5

(Taken from Woodruff and Siddoway, 1965)

C.3 Comparison of Travis and Cowherd Models

A major difference between the two models is the time frame over which the data are generated. The Travis uses instantaneous wind speed data to provide a set of emission values. Superimposing a yearly wind speed frequency analysis provides generalized annual emission data. The Travis model therefore lends itself readily to use with conventional dispersion models that use wind speed data for dispersion estimates. On the other hand the Cowherd model is limited to the interpretation of wind speed data using an assumed wind profile and frequency pattern. No instantaneous values are produced by the Cowherd equation and the sensitivity may be poor.

The Travis model makes no simple allowance for changing moisture on the surface of the material. The climatological parameter, P-E (Thornthwaites Evaporation Index) of the Cowherd model does provide some recognition of changing moisture levels. It calculates these as a function of precipitation and temperature.

Both models have significant limitations in estimating emissions for very fine tailings. The Travis model "P" causes the values to rise very quickly for small changes in the amount of <20 um material. The Cowherd "e" erodibility factor provides a less sensitive measure of generation potential but is still difficult to use.

Both models use the logarithmic relationship for wind speed. This is satisfactory for neutral stable conditions. However, data have shown that this is not strictly applicable under stable and unstable conditions.

Cowherd (1979) states that the best way to develop wind erosion emission factors for a specific source is to test the actual surface. Woodruff and Siddoway (1965) state that more information on cloddiness, soil moisture variation and seasonal/annual erodibility is required. Keeping the foregoing limitations in mind the emission factors for tailings areas can be calculated.

C.4 DERIVATION OF SITE SPECIFIC PARAMETERS FOR ESTIMATING FUGITIVE EMISSIONS FROM TAILINGS AREAS IN THE KIRKLAND LAKE AND TIMMINS MINING REGIONS

The regions of interest are located northeast of Sudbury. For purposes of this study, meteorological data from Sudbury will be used. The STAR analysis of Sudbury data (joint wind speed-wind direction-stability frequency of occurrence) was supplied by AES (Atmospheric Environment Service).

The percentage of different wind classes is listed in Table C.3. North of Lake Huron the Thornthwaites P-E value decreases from 115 at Expanola to 100 at Cochrane (Beak, 1981). The value of 115 was selected as input to the Cowherd expression.

TABLE C.3 FREQUENCY OF WIND SPEED CLASSES IN SUDBURY AREA (STAR DATA)

Wind Speed at 10 M (m/s)	Frequency of Occurrence (%)
0------1.54	8.2
1.54---3.09	18.3
3.09---5.15	31.7
5.15---8.24	30.5
8.24--10.81	8.7
>10.81	2.5

The Cowherd model requires a frequency of occurrence for winds greater than 5.3 m/s at 0.3 m above the surface. This translates to 10 m levels of 46 m/s if the logarithmic profile is applied. A maximum frequency of 25% was selected on the basis that 2.5% of the annual winds are greater than 10.8 m/s. It is recognized that this yields a very conservative emission rate and that the actual release rate is probably a small portion of the calculated value. However, the freeze/thaw cycles influence the surface material degradation thereby giving rise to significant emissions during periods of high wind velocities. Under normal conditions, the emission rate would be minimal.

Surface winds in the Travis model are a function of Zo, the surface roughness. This roughness can be approximated to 1 cm for "clay-type" tailings (Beak, 1981) and a value of 5 cm for "rock" tailings. The shear velocity is calculated by adjusting the 10 m wind speeds to representative heights, 1 and 5 cm. The mid-points of the classes shown in Table C.2 were selected as the representative Ut values.

The values for erodibility "e" and the fraction <20 um "P" are based on laboratory analyses for the size distribution of two samples; a sample from the clay-like tailings at Kidd Creek and a sample from an abandoned Kirkland Lake tailings area. The results shown in Table C.4, demonstrate that there are two distinct types of tailings materials, the clay tailings where 99.6% of the material is less than 46 um diameter and the rock like tailings where 83% of the material is greater than 1500 um diameter.

TABLE C.4 PARTICLE SIZE DISTRIBUTION FOR KIDD CREEK AND SYLVANITE TAILINGS

Size	Equivalent Diameter (um)	Kidd Creek Percent Passing	Sylvanite Percent Passing
10	1500	100.0	16.8
100	150	99.9	-
200	75	99.8	3.5
325	46	99.6	3.1

With the amount of fines in the clay sample, the erodibility is greater than 694 t/ha/a and P would be at least 50 assuming 1/2 of the material is <20 um. The resulting emissions do not agree with the theory discussed earlier. The clay sample will be assumed to have a "P" of 25 and thus "e" will be selected at 86 t/ha/a. The silt content is selected at 99%.

The rock-like tailings sample has a high proportion of material greater than 0.84 um. Selecting "e" corresponding to 70% greater than 0.84 um yields 27 t/ha/yr. The value of "P", assuming that the amount of material is less than 20 um in size is 50% of the <46 um material, is 1.5. The silt content, defined as the amount passing a 200 mesh sieve, is 3.5%.

As Cowherd (1979) indicates that the top 2-3 cm of material are effected by wind erosion, only surface moisture is of importance to the model. A diurnal variation in moisture for storage piles is also reported by Cowherd. Seasonal variations can also be expected based on precipitation. Given that the emissions decline to zero at moisture levels of 1.3% for dune sand, and 20.7% for silty clay, values in this range should be selected. This will provide emission estimates although their exact magnitude may be in error. The clay moisture was selected at 10%, the rock at 3%.

The value for the saltation particle diameter in the Travis model was selected on the basis of literature data. Most literature reports that the size of the saltation particles generally range from 100-500 um. Since the larger the diameter, the higher the threshold wind velocity, it was decided to use the minimum diameter (100 um). In this way the model is possibly conservative in terms of emission levels.