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Exposure of the Public from Large Deposits of Mineral Residues

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FOREWORD

All minerals and raw materials contain radionuclides of natural origin. In most situations, the exposure of humans to such radionuclides is considered to be part of the normal natural radiation background and is not generally of concern. In some cases, however, the radionuclide concentrations are elevated above normal levels or become elevated as a result of mineral processing activities, and measures for protecting against exposure to the material involved may need to be considered. The mineral or raw material is then treated as radioactive material for the purposes of radiation protection and falls within the definition of naturally occurring radioactive material (NORM).

The IAEA has developed criteria for determining which materials need to be considered for regulatory control. For materials containing only radionuclides of natural origin, the criteria are an activity concentration of 1 Bq/g for ^{238}U, ^{235}U, ^{232}Th and their decay progeny and an activity concentration of 10 Bq/g for ^{40}K. These values were determined on the basis of the activity concentrations of these radionuclides in normal rocks and soil, and represent the (rounded) upper bounds of the ranges of such concentrations as determined by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR). The values are intended to apply to all solid materials except foodstuffs, material in transport and radioactive residues in the environment (for which separate criteria apply) and ^{40}K in the body (which is excluded entirely from regulatory requirements).

While the radiation dose was not a consideration in the determination of the above-mentioned regulatory criteria, the IAEA has noted that doses received by individuals as a consequence of the use of these criteria are unlikely to exceed about 1 mSv in a year, excluding the emanation of radon. However, in the case of bulk volumes of material contaminating water pathways, such as large deposits of NORM residues from mining and mineral processing, it has been suggested that case by case evaluation of the dose may be required. It was therefore decided to conduct further investigations of the doses expected to be received as a result of exposure of members of the public to a large NORM residue deposit, with consideration being given to all potentially significant exposure pathways including those involving contamination of water. The investigations were conducted using an evidence based approach involving the review of available information from real world examples of actual NORM residue deposits, as well as a calculation approach involving the modelling of radionuclide migration from a ‘representative’ large NORM residue deposit.

The investigations were carried out during 2009 under contract to the IAEA by a Canadian consulting company, SENES Consultants Limited. This report gives the results of those investigations. The IAEA Officer responsible for the preparation of this report was D.G. Wymer of the Division of Radiation, Transport and Waste Safety.

EDITORIAL NOTE

CONTENTS

1. INTRODUCTION

1.1. Background

All minerals and natural raw materials contain the so-called ‘primordial’ radionuclides ^{238}U, ^{235}U and ^{232}Th and their decay progeny, as well as ^{40}K. In the majority of situations, the radionuclide concentrations are not sufficiently elevated to pose a radiological hazard and the material is not treated as being radioactive for purposes of radiation protection. In some cases however, where the radionuclide concentrations are significantly higher than the normal range of background levels, there may be a potential for exposures that are of concern from a radiation protection point of view. In such cases, the material is treated as radioactive material and thus falls within the definition of NORM.

A wide range of activity concentrations in a wide variety of materials is reported (see, for instance, Ref. [1]). Examples of ores that have been found to be associated with elevated radionuclide concentrations include those of uranium, tin, tantalum, niobium, rare earths, aluminium, copper, gold and phosphate. The mining and processing of these ores can lead to further increases in radionuclide concentrations in the products, by-products or residues. A few examples of the activity concentrations involved are given in Table 1.

TABLE 1. EXAMPLES OF ACTIVITY CONCENTRATIONS IN NORM
(adapted from Ref. [2])

	Radionculide with highest activity concentration	Typical activity concentration (Bq/g)
Monazite sand	^{232}Th series	40–600
Metal ores, e.g., Nb, Ta, Cu, Au	^{238}U and ^{232}Th series	Up to 10
Zircon sand	^{238}U series	2–4
Phosphate rock	^{238}U series	0.03–3
TiO_2 feedstocks	^{232}Th	0.001–2
Bauxite	^{232}Th series	0.035–1.4
Red mud (alumina production)	^{238}U, ^{232}Th	0.1–3
Phosphogypsum (H_2SO_4 process)	^{226}Ra	0.015–3
Niobium extraction slag	^{232}Th	20–120
Tin melting slag	^{232}Th	0.07–15
Scale (oil and gas production)	^{226}Ra	0.1–15 000
Residue (rare earth extraction)	^{228}Ra	20–3 000
Scale (TiO_2 pigment production	^{228}Ra, ^{226}Ra	<1–1600
Scale (rare earth extraction)	^{226}Ra, ^{228}Th	1000
Sludge (oil and gas production)	^{226}Ra	0.05–800
Residue (niobium extraction)	^{228}Ra	200–500
Coal	^{238}U and ^{232}Th series	0.01–0.025
Scale (coal mines with Ra rich inflow water)	^{226}Ra, ^{228}Ra	Up to 200

Activity concentration criteria for the regulation of materials have been established by the IAEA and are given in Ref. [3]. In developing these criteria, it was concluded that it was not practical to derive criteria for radionuclides of natural origin on the basis of dosimetric considerations (as was done for radionuclides of artificial origin), since in many cases such an approach would produce activity concentrations lower than those occurring in the natural environment. Therefore, the activity concentration criteria for radionuclides of natural origin were based on the upper bound of the worldwide distribution of natural radionuclides as given, for example, in Ref. [1]. Consequently, it is stated in Ref. [3] that "It is usually unnecessary to regulate…" material containing radionuclides of natural origin at activity concentrations below 1 Bq/g for radionuclides in the uranium and thorium decay series and below 10 Bq/g for ^{40}K.

1.2. Objective

While dose was not the key consideration in deriving the activity concentration criteria for radionuclides of natural origin, it is stated in Ref. [3] that "Doses to individuals as a consequence of these activity concentrations would be unlikely to exceed about 1 mSv in a year, excluding the contribution from the emanation of radon, which is dealt with separately in the BSS". It is generally recognized that the main potential for exceeding an annual dose of 1 mSv per unit activity concentration of 1 Bq/g arises from a scenario involving exposure of members of the public to large mineral residue deposits, such as mine tailings and 'waste' rock (rock with low levels of mineralization that does not necessarily fall within the IAEA's definition of waste because of the potential for further use).

The objective of this report is, firstly, to present the findings of an investigation to determine the doses expected to be received by members of the public exposed to large NORM residue deposits, with consideration being given to all potentially significant exposure pathways, including those involving contamination of water, and to the radionuclide activity concentrations in the residue material. The investigation was carried out under contract to the IAEA by SENES Consultants Limited [4] using an evidence based approach involving the review of available information from real world examples of actual NORM residue deposits, as well as a calculation approach involving the modelling of radionuclide migration from a 'representative' large NORM residue deposit.

Secondly, the objective of this report is to establish, through the findings of the investigation, the consequences of applying the recommended activity concentration criterion of 1 Bq/g in situations where individuals are exposed to large mine residue deposits. Although, as stated above, the derivation of the activity concentration criterion was not based on dose considerations, a knowledge of the doses likely to be received in such situations would help to establish whether or not the use of the 1 Bq/g criterion for determining the scope of regulatory control could, in any reasonable circumstances, lead to an exposure situation that would be regarded as unacceptable.

1.3. Scope

The scope of the investigation was to assess the doses expected to be received by members of the public living near a large mineral residue deposit using, to the extent possible, an evidence based approach based on data gathered from actual mineral residue deposits. The key assumptions concerning the characteristics of the residue deposit were:

- A nominal deposit volume of 2 million m^3 covering 10 ha;
- The presence of radionuclides in the ^{238}U and/or ^{232}Th decay series, each at a concentration of 1 Bq/g;
- The possibility of the residue material being acid generating.

The dose calculations were done using, where appropriate, the same modelling approach and assumptions used for deriving the activity concentration criteria reported in Ref. [3] for radionuclides of artificial origin. The modelling approach and assumptions are described in Ref. [5].

The key elements of the study were:

- To consider the available information from real world examples of actual NORM residue deposits that could be used to define the characteristics of a representative NORM residue deposit and the parameters controlling the migration of radionuclides from the deposit;
- To define the relevant exposure pathways and conduct a dose assessment for the most highly exposed members of the public.

For this assessment, the exposed individuals were assumed to live within a few metres of the NORM residue deposit and, for consistency with the modelling approach used in Ref. [5], an adult and a 1–2 year old child were chosen as hypothetical receptors. The exposure pathways considered in this assessment were:

- Inhalation of airborne dust;
- External exposure from dust deposited on the ground at the residence;
- Ingestion of dust from dust deposited on the ground at the residence;
- Ingestion of garden and agricultural products from irrigation with contaminated groundwater;
- Ingestion of contaminated groundwater;
- Ingestion of fish obtained from contaminated surface water.

For consistency with the approach taken in Refs [3, 5], inhalation of radon was not formally included in the scope of the investigation. However, the results of separate calculations for airborne radon concentrations are given.

1.4. Structure

Following this introductory section, Section 2 provides a summary of the empirical evidence collected for this study pertaining to the characteristics of the various NORM residue deposits and their potential leachate characteristics. Section 3 provides details of the models and data used to perform the exposure pathway assessment and presents the results of the calculations. Section 4 discusses the results of the dose assessment and draws some conclusions on the key exposure pathways and radionuclides involved. Details of the dose calculations, the relevant air dispersion considerations and the sensitivity of the results to the soil–groundwater distribution coefficients are given in three appendices.

2. EMPIRICAL EVIDENCE

2.1. Types of residue considered

Many bulk NORM residues contain elevated levels of natural radionuclides (see Table 1, for instance). Such materials include waste rock from many different types of mining operations and mineral processing residues such as ‘red mud’ from bauxite production, gold tailings, copper slag, tin slag and phosphogypsum (a by-product of fertilizer production). Uranium and thorium series radionuclides are commonly found in a number of mining operations including amongst others, uranium mining (e.g. Canada, Germany), mining of rare earth and niobium deposits (many sites internationally), gold mining (e.g. South Africa), tin mining (e.g. Canada, Brazil) and copper mining (e.g. Australia). Mining of other non-metallic ores, along with ore processing, also generates large quantities of NORM, examples being the production of phosphate fertilizer and abrasives and the generation of fly ash and coal ash residues from coal-fired power generation.

The chemical and mineralogical properties of mining residues and metallurgical products and the presence of other chemicals such as organic matter or iron oxide can affect the availability of the radionuclides to the environment. Also, mining or metallurgical processes can affect the mobility of radionuclides in the natural environment. In this regard, there are few differences (other than differences in radionuclide activity concentration) between residues from the mining and processing of uranium ore and those from other types of mining and mineral processing operations. The resulting radiological impacts per unit activity concentration are likely to be similar for both cases.

Residues from uranium mining include waste rock, tailings, treatment sludges and so-called ‘clean waste’. Clean waste has no specific definition but based on precedent, as for example in Canada, would include materials such as waste rock with less than 0.03% U_3O_8 (implying a ^{238}U concentration of about 3 Bq/g) and would be non-acid-generating. Other elements such as arsenic, selenium, sulphur (as sulphide), nickel, molybdenum and vanadium may also be present and may be used as a marker to define clean waste. Clean waste from uranium mining is likely to be similar to NORM residues with low mineralization that may be generated at other mining operations.

Potential issues with waste rock include elevated levels of radioactivity (radon, elevated gamma fields, and uranium/radium in leachates). In some cases, such materials also have the potential to generate acid drainage as a consequence of the oxidation of sulphide mineral present in the rock. In most jurisdictions, waste rock is closely regulated using various measures including monitoring, closure management and restrictions on its use as a product. There are several approaches to the management of residues with acid generating and/or metal leaching potential. Short term management can include temporary storage on lined pads with all seepages intercepted and treated. Long term management includes isolation under engineered covers and disposal in mined out cavities. In some locations where local conditions permit, water barriers are used to prevent oxidation of sulphides and to reduce metal mobility. There is a reasonably extensive database on leachate chemistry from uranium mine residue deposits, but considerably less information on the quality of leachates from other types of NORM residue deposit.

2.2. Sources of data

The information regarding the characteristics of NORM residue deposits was obtained from the following sources:

- IAEA publications;
- Open literature and journal papers;
- Proceedings of workshops and conferences;
- SENES in-house reports and information from uranium mining and milling operations.

2.3. Characteristics of residues

The characteristics of NORM residue deposits are quite variable and are dependent in large part on the geological setting of the source material, the specific uranium/thorium content of the material and the effects of processing (concentration/mobilization of radionuclides in some cases). For most unaltered residue materials, radionuclides are mobilized with weathering and the passage of water and air through the pile. Some of the radionuclides will migrate more readily than others which may then be precipitated or adsorbed on the surfaces of residue materials and local subsurface soils. Retardation of the movement of radionuclides such as radium and uranium has been demonstrated at many locations and retention coefficients, although highly variable, have been reported for many soils and rock types.

2.3.1. Physical and chemical characteristics

The physical, chemical, and geochemical characteristics of mining and industrial mineral based residues vary greatly. A large amount of information is available from various international studies on such materials. Examples of important properties include geochemical components, the presence of chemical contaminants, acid generating potential, carbonate content, pH, water content, porosity, organic matter content, clay content and cation exchange capacity.

Data from a waste rock pile at the Rabbit Lake uranium mine in northern Saskatchewan, Canada indicates that this rock is comprised primarily of sandstones (with a smaller fraction of chloritic gneiss and graphitic gneiss) and approximately 50% of the material in the pile is greater than 10 cm in diameter [6]. Since the coarse fractions will retain much less moisture than the fines, the volumetric moisture content is expected to be substantially lower than that of the fines. For this waste rock pile, the average gravimetric moisture content was 8.7%, similar to the volumetric moisture contents in the range of 8% for another site in Saskatchewan (Key Lake). Moisture data from other waste rock sites are typically in the range 7–10% [6].

Waste rock from a former uranium mining area in Ronneburg, Germany contains carbonates, ochre limestones, and calcareous slates. These constituents are lumped together with slates and alum slates containing pyrite and carbon creating geochemical variabilities in the waste rock piles [7].

The waste rock pile at Barzava uranium mine in western Romania contains grey sandstone; micro-conglomerates; clay with organic substances and crystalline schistose, with a particle size range of 10–100 mm [8].

The residues in the vicinity of the Kalna abandoned uranium mines in Serbia are made up of varied lithological complexes that differ in age, origin, mineral and petrographic compositions. The oldest rocks are amphibolite, gneiss and greenstone of Riphean/Cambrian age (diabase phyllite formation) [9].

A large variety of NORM residues arising from the processing of ores includes tailings, treatment sludges, various slags from the production of tin, niobium and tantalum ores, as well as phosphogypsum, a by-product of fertilizer production.

2.3.2. Radioactivity content

In addition to the uranium, thorium and radium isotopes included in the uranium and thorium decay series, other radionuclides of potential concern in NORM residue deposit seepage include ^{210}Pb and ^{210}Po. Monitoring of groundwater around tailings basins has demonstrated that ^{210}Pb and ^{210}Po tend to be present at concentrations below that of ^{226}Ra. Typically, ^{210}Po appears immobile as a result of precipitation or adsorption on solid surfaces.

2.3.2.1. Residues from the mining and processing of uranium ore

A range of uranium and thorium concentrations for waste rock at Key Lake uranium mine, Canada are shown in Table 2. Similar results for waste rock at Rabbit Lake uranium mine, Canada are shown in Table 3 [6].

TABLE 2. TYPICAL RADIOACTIVITY LEVELS IN VARIOUS FORMS OF WASTE ROCK AT KEY LAKE URANIUM MINE, CANADA

	Activity concentration (Bq/g)			
	Low grade ore	'Special waste'	'Clean waste'	'Acidic waste'
U-238	>12	3.7–12	<3.7 (mean 1.2)	1.2
Th-232	>1.2	0.41–1.2	0.12	0.12

Note: These data were reported in the January 2005 waste rock management plan for Key Lake uranium mine. The data are similar to those for other uranium mine waste rock piles.

TABLE 3. TYPICAL RADIOACTIVITY LEVELS IN WASTE ROCK AT RABBIT LAKE URANIUM MINE, CANADA

	Unit	Concentration
Uranium	% U	0.0033-0.14
U-238	Bq/g	0.41-17
Ra-226	Bq/g	0.6-15

In Ref. [10], it is reported that test samples from waste rock piles from Midnite mine (a former uranium mine in the USA) indicated that the uranium content of the waste rock material ranged from 7 to 229 ppm (^{238}U: 0.09–2.8 Bq/g).

In Germany, at the former Gessenhalde uranium heap leaching facility, the average residual soil concentrations of uranium, thorium, and lead were about 8.5, 12 and 20 ppm, respectively (^{238}U: 0.1 Bq/g; ^{232}Th: 0.04 Bg/g) [11].

The analytical results from residues in the vicinity of the Kalna abandoned uranium mines in Serbia indicated that uranium concentrations ranged from 1.4 to 4.3 ppm while thorium concentrations ranged from 3.4 to 12.3 ppm. The results of the analysis indicated that the primary source of uranium in the area is the fragmented granitic rocks [9].

The compositions of the waste rock pile and other residues in the close vicinity of the Barzava uranium mine located in the west of Romania were measured. The analytical results indicated that the uranium concentration ranged from 53 to 175 ppm (^{238}U: 0.65–2.2 Bq/g) while ^{226}Ra concentrations were between 0.15 and 0.9 Bq/g [8].

Overall, a review of the information shows that the uranium activity concentrations in residues associated with the mining and processing of uranium ore generally vary between 0.1 and 17 Bq/g while the corresponding range for thorium is between 0.04 and 1.2 Bq/g. Thus, the uranium activity concentrations vary by more than two orders of magnitude, while the range of thorium concentrations is narrower. In both cases, the ranges extend above and below the regulatory criterion of 1 Bq/g (see Section 1.1).

2.3.2.2. Residues from the mining and processing of ores other than uranium ore

Significant concentrations of uranium and/or thorium are commonly associated with a number of other mining operations, including rare earth and niobium mining (many sites internationally), tin mining (several locations), copper mining (e.g. Olympic Dam, Australia) and gold mining (e.g. South Africa). Depending on the nature of the residues, the facilities containing the residues may or may not have engineered containment. Typical data on selected tailings and waste rock sites with NORM are provided in Table 4.

TABLE 4. TYPICAL RADIOACTIVITY LEVELS IN NORM RESIDUES OTHER THAN THOSE ASSOCIATED WITH THE MINING AND PROCESSING OF URANIUM ORE

	Activity concentration (Bq/g)					
	Waste rock, copper mining	Phospho-gypsum [12]	Gold tailings, South Africa [13]	Red mud [14]	Coal ash	Waste rock, niobium mining
U-238	1.0 (0.11–2.1)	~ 0.1 (0.005–0.5)	1.0 (0.1–5)	<0.4	0.10–1.0	0.34
Th-232	–	0.004–0.6[a]	–	0.41	–	1.2
Ra-226	–	~1 (0.015–5.1)	1.4 (0.1–5)	<0.18	–	–

[a] Highly dependent on the deposit characteristics, with phosphogypsum from igneous sources containing much more thorium than phosphogypsum from sedimentary sources.

During the phosphoric acid production process, uranium, thorium and lead distribute primarily in the phosphoric acid, while most of radium, polonium, and traces of uranium appear in the phosphogypsum [15]. An analysis of ^{238}U, ^{230}Th and ^{226}Ra in samples taken from three phosphogypsum stacks in Florida, USA and two in Canada is reported in Ref. [16]. The ^{238}U activity concentrations were in the range of 0.092–0.530 Bq/g, while the activity concentrations for ^{230}Th were in the range 0.072–0.150 Bq/g. The range of activity concentrations for ^{226}Ra was 0.310–0.930 Bq/g. These data confirm that some preferential distribution to the fertilizer product of uranium occurs compared with radium, most of which ends up in the phosphogypsum by-product.

An analysis of phosphogypsum derived from phosphate rock from Florida Togo, and Idaho indicated that the average activity concentrations of ^{226}Ra ranged from 0.43 to 0.93 Bq/g, while the average activity concentrations of ^{210}Pb ranged from 0.34 to 0.840 Bq/g [17]. The average activity concentrations of ^{228}Th were between 0.004 and 0.009 Bq/g.

For NORM residues other than those associated with the mining and processing of uranium ore, the range of uranium activity concentrations is generally from 0.1 to 5 Bq/g, while the corresponding range for thorium activity concentrations is 0.004 to 1.2 Bq/g. The activity concentrations depend on the type of mining and processing activity and type of residue. As with uranium mining residues, the ranges of concentrations extend above and below the recommended regulatory criterion of 1 Bq/g.

2.3.3. Leachate

There is often a potential to produce contaminated drainage even though the residue piles are believed to contain only low levels of contaminants. A thorough understanding of the chemical and mineralogical characteristics of the residue is necessary to understand and predict leachate chemistry.

NORM residues from mineral processing operations typically contain meta-stable components such as precipitates that can release contaminants including heavy metals and radionuclides. Also, chemical oxidation can have a large impact on the mobility of radionuclides in tailings and other residues. Uranium, for example, may be converted from U(IV) to U(VI) and become more mobile especially in the presence of carbonate. A fraction of the radium content may be mobilized in hydrometallurgical processes and co-precipitated with other metal sulphates. The mobility of this co-precipitated radium becomes controlled by sulphate levels. Dissolution is assumed to be the dominant mechanism of release of uranium and ^{226}Ra from oxide, sulphate and carbonate minerals. However, release from these minerals can also be influenced by the pH and redox changes associated with oxidation. The presence of carbonate, iron oxides, clay minerals and organic matter can greatly affect the mobilization of radionuclides in the residues. Because of the stability of the uranyl carbonate complexes, uranium-rich residues with naturally high carbonate contents tend to produce higher levels of uranium in leachates. High levels of chloride also tend to mobilize radium.

Fluctuations in pH can result in a relatively rapid release of radionuclides and this is likely to be attributable to changes in both the solubility of the host mineral and the retention coefficients. For example, uranium and thorium are more soluble under acid conditions while radium solubility may be reduced (in the presence of high sulphate concentrations). Should a residue deposit be acid generating, uranium levels may increase, thorium will be mobilized if the residue deposit is very acidic, and radium levels will be at similar or possibly lower levels owing to secondary precipitation of radium–sulphate complexes.

Overall, the factors controlling the solubility of key radionuclides in NORM residues are reasonably well known but the actual characteristics of leachate vary greatly. Laboratory and field data from various sites show that the rates of contaminant release from waste rock change over time. In cases where there is no sharp change in pH, that is, where the system remains neutral, contaminant release rates generally decrease over the long term [6].

In addition to the above factors, water flow within NORM residue deposits is another important factor influencing the mobility of radionuclides from these deposits. The average velocity at which water moves through the deposit is a function of various parameters, including the infiltration rate and the overall volumetric water content of the deposit.

Typical leachate characteristics for residues at the Key Lake uranium mine in Canada are shown in Table 5. These residues produce neutral drainage with elevated levels of uranium isotopes and ^{226}Ra and negligible levels of thorium. Uranium levels show the greatest variability.

TABLE 5. TYPICAL CHARACTERISTICS OF LEACHATES FROM URANIUM MINING

	Leachate from ‘clean waste’	Leachate from ‘acidic waste’
U (mg/L)	0.5 (0.1–5)	5
U-238 (Bq/L)	6 (1.2–60)	60
Ra-226 (Bq/L)	0.5 (0.1–1.5)	0.5
Th (μg/L)	<1	100
Th-232 (Bq/L)	<0.004	0.4

Note: The values are based on data reported in the January 2005 waste rock management plan for Key Lake uranium mine. These data are similar to data for other mines.

The leachate from waste rock piles from the Midnite mine in the USA exhibited alkaline characteristics with leachate pH values near to 8.5 [10]. The tests indicated that the oxidation during the dry period increases the uranium leaching during subsequent rainfalls. The uranium leach concentrations were mostly less than 5 mg/L (^{238}U: 60 Bq/L) and generally about 2 mg/L (^{238}U: 25 Bq/L). On the other hand, the samples from different layers of the waste rock pile had pH ranges of 3 to 7. The uranium concentrations of leachate were consistently below 0.2 mg/L (^{238}U: 2.5 Bq/L) with initial leachate concentrations as high as 2.9 mg/L (^{238}U: 36 Bq/L) [10]. This part of the waste rock pile is not as reactive with respect to uranium.

The test results for leach samples from the Rabbit Lake waste rock pile in Canada indicate that arsenic, molybdenum, nickel and uranium dominate the chemistry in several of the samples [6]. The alkalinity was typically very low, and sulphate was the dominant anion. The pH values tended to be weakly acidic. Table 6 shows the test results for leach concentrations in samples of Rabbit Lake waste rock.

TABLE 6. TYPICAL CHARACTERISTICS OF LEACHATE FROM RABBIT LAKE URANIUM MINE [6]

	Unit	Concentration
U	µg/L	40–13 800
U-238	Bq/L	0.5–170
Pb	mg/L	<0.002–0.022
Ra-226	Bq/L	0.33–5.5

Lead that is leached from NORM residue deposits may adsorb on underlying soils. Lead may precipitate in soils if soluble concentrations exceed about 4 mg/L at pH 4 and about 0.2 mg/L at pH 8. In the presence of phosphate and chloride, these solubility limits may be as low as 0.3 mg/L at pH 4 and 0.001 mg/L at pH 8 [18]. Therefore, in situations in which concentrations of lead exceed these values, the estimated distribution coefficients (K_d) may reflect precipitation reactions rather than adsorption reactions. Anionic constituents such as phosphate, chloride, and carbonate are known to influence lead reactions in soils either by precipitation of minerals of limited solubility or by reducing adsorption through complex formation [18]. Depending upon the nature of the tailings or other residues, the deposits may or may not have engineered containment.

It is also reported in Ref. [7] that the analytical prediction of the geochemical behaviour of secondary minerals at the Ronneburg site is very complex and quantitative predictions of the water quality in the seepage are notoriously unreliable.

Typical leachate data for selected tailings and waste rock sites are given in Table 7. The analytical data for mine and seepage water from the Wismut mining district located in Ronneburg, Germany are shown in Table 8.

TABLE 7. TYPICAL LEACHATE ACTIVITY CONCENTRATIONS ASSOCIATED WITH NORM RESIDUES

	Activity concentration (Bq/L)				
	Waste rock, copper mining	Phospho-gypsum	Red mud	Coal ash	Waste rock niobium mining
U-238	2.4	0.043[a]	0.1–0.6	–	<0.2
Ra-226	0.26	<0.74[a] 0.07–0.53[b]	0.02–0.03	0.8	1

[a] From Ref. [15].

[b] From Ref. [19].

TABLE 8. COMPOSITION OF TYPICAL MINE AND SEEPAGE WATERS OF THE RONNEBURG SITE, GERMANY [7]

	pH	Uranium concentration (mg/L)	Activity concentration (Bq/L)	
			U-238	Ra-226
Mine water, central section, measuring point e-567	3.8	0.1	1.2	0.150
Mine water, central section, influenced by open pit mine and dumps, measuring point e-480	2.9	4.4	55	0.384
Mine water, SE section of deposit, measuring point MW 435/2	7	1.7	21	0.192
Seepage, Absetzerhalde dump, measuring point e-440	2.8	7.2	89	<0.01
Seepage, Beerwalde dump site, measuring point s-611	7.6	5.2	64	0.136

The concentrations of radionuclides in the Lerchenbach creek north of the former tailings management area of a Wismut site in East Trunzig, Germany are shown in Table 9 [20]. The tailings were generated from ore containing ^{238}U decay series radionuclides at an activity concentration of the order of 10 Bq/g. It is concluded that, given the characteristics of waters encountered in the vicinity of abandoned Wismut sites (with mostly neutral or acidic pH), the radionuclide transport was typically determined by the uranium isotopes ^{238}U, ^{234}U and ^{235}U. These radionuclides accounted for the major part of the effective dose.

TABLE 9. ACTIVITY CONCENTRATIONS IN SURFACE WATER NEAR THE TAILINGS MANAGEMENT AREA, EAST TRUNZIG, GERMANY [20]

	Activity concentration in water (Bq/L)
U-238	5.2
U-234	6.1
Th-230	0.17
Ra-226	0.02
Pb-210	0.025
Po-210	0.025
U-235	0.24
Pa-231	0.015
Ac-227	0.015

It is reported in Ref. [8] that the uranium concentration in surface water in the close vicinity of Barzava uranium mine in Romania was 0.014 mg/L (^{238}U: 0.17 Bq/L), while the activity concentration of radium was 0.043 Bq/L. The authors concluded that the argillaceous soil at the bottom of the waste rock pile worked as a barrier that retarded the movement of radioactive contaminants toward the ground and consequently surface water.

The analytical results for well water in the vicinity of the Kalna abandoned uranium mines in Serbia are reported in Table 10 [9]. It was concluded that the geochemical barriers of clays and organic materials (alluvium) and faster filtration of groundwater control the dissolution of uranium and trace elements in groundwater downstream of the mines. Uranium concentrations were all less than 0.009 mg/L (0.1 Bq/L). A radiometric analysis of samples from the mine residue was not provided in this study.

TABLE 10. RANGES OF MEASURED ELEMENTS IN WELL WATER IN VARIOUS LOCATIONS AROUND THE KALNA ABANDONED URANIUM MINE

	Uranium concentration (mg/L)	Activity concentration (Bq/L)		pH
		^{238}U	^{226}Ra	
Balta Berilovac	0.005–0.009	0.062–0.111	0.05–0.07	7.3–7.4
Vrtovci	0.001–0.002	0.012–0.025	0.09–0.20	7.0–7.1
Inovo	0.002–0.008	0.025–0.100	0.05–0.06	7.4–7.7
Gornja Kamenica	0.001–0.002	0.012–0.025	0.20–0.60	7.3–7.5
Donja Kamenica	0.001–0.003	0.012–0.037	0.10–0.19	7.5–7.8
Strbac	0.001–0.003	0.012–0.037	0.05–0.07	7.1–7.3
Baranica	0.001–0.003	0.012–0.037	0.11–0.20	7.4–7.6

During the uranium milling periods at Moab uranium mine site in Utah, USA, Atlas Minerals monitored ^{226}Ra, ^{230}Th, ^{222}Rn and uranium at the tailings pond [21]. The results indicated that the ^{226}Ra and ^{230}Th activity concentrations were 3.7 and 1.9 Bq/L, respectively. The uranium activity concentration was 22 Bq/L.

Data published in Ref. [22] for typical seepage/groundwater levels measured in the plume from the Nordic tailings management area (Elliot Lake, Ontario, Canada) are shown in Table 11. The average radionuclide activity concentration of the Nordic tailings is about 10 Bq/g [23].

TABLE 11. ACTIVITY CONCENTRATIONS IN SEEPAGE/GROUNDWATER AT VARIOUS DISTANCES FROM THE NORDIC TAILINGS DAM, ELLIOT LAKE, CANADA

Distance from dam (m)	pH	Activity concentration (Bq/L)				
		^{226}Ra	^{238}U	^{210}Pb	^{232}Th	^{230}Th
0	4.5	5.5	6.5	4	Not detectable	0.14
10	4.5	2.0	1.2	Not detectable	Not detectable	0.14
20	5.0	0.4	<0.1	Not detectable	Not detectable	Not detectable
70	6.3	<0.1	<0.1	Not detectable	Not detectable	Not detectable

A mathematical modelling of leaching of radionuclides from the Urgeirica uranium tailings located in central Portugal is reported in Ref. [24]. The modelling results indicated that the activities of ^{226}Ra and total U in well water 500 m away could reach 1 and 6 Bq/L, respectively, after 100 years of leaching and movement of these radionuclides through the groundwater. Based on the modelling results, it was concluded that due to the slow rates of contamination migration, only radionuclides with relatively long half-lives are important in the transport process. Therefore, with regard to polonium, a nil concentration was estimated in the well water. This can be explained by the fact that the half-life of ^{210}Po is about 137 d, which is very short compared with its travel time. The radionuclides ^{230}Th and ^{210}Pb are generally not transported over significant distances due to the particle-reactive nature of thorium and the great tendency of lead to be adsorbed by the aquifer sediments [24].

In Germany, at the former Gessenhalde uranium heap leaching facility, the average leachate concentrations of uranium and lead were 59 and 0.9 µg/L, (0.73 and 0.11 Bq/L), respectively [11]. It was concluded that, based on the current level of contamination and the available historical information, leachate infiltrated through the barrier soil into the loamy soil and was trapped on top of the glacial clay, thereby being prevented from infiltrating further downwards. The water concentrations were found to be pH dependent for most elements, with low pH areas exhibiting higher concentrations. Most elements indicated an almost logarithmic increase with decreasing pH [11].

Refs [25], [26] describe a geochemical and radiological assessment of a uranium tailings dam in Poços de Caldas, Brazil. The activity concentrations of ^{238}U series radionuclides in the solid material were in the range 4.1–6.6 Bq/g. Radionuclide concentrations in near-surface seepage and in the tailings dam pond water are summarized in Table 12. Although the data are limited, they show that the pond water values are very much lower than the seepage values. For example, the ^{226}Ra activity concentrations differ by a factor of about 40, while ^{210}Pb concentrations differ by a factor of 200. It was also noted that the activity concentrations of

^{238}U and ^{226}Ra in the groundwater were essentially the same upstream and downstream of the point at which the effluents from the tailings dam were released because the effluents were treated with lime and $BaCl_2$.

Ref. [27] gives a compilation of the results of studies conducted over several years to determine radionuclide concentrations in waters impacted by gold mining operations in the sedimentary gold deposits of the Witwatersrand Basin in South Africa. The area is characterized by low grade uranium mineralization with an average uranium oxide grade from a particular conglomerate formation in a mine varying from 0.001 to 0.078 %. The mean activity concentrations of ^{238}U and ^{226}Ra in the mine tailings were each about 1 Bq/g. The mean activity concentrations of ^{238}U and ^{226}Ra in surface water and groundwater samples are given in Table 13. The activity concentrations of measured radionuclides other than uranium isotopes and ^{226}Ra were close to natural background concentrations.

Heavy metals and other toxic elements and radionuclides are not readily leachable from phosphogypsum solids [16]. It was noted, however, that process fluids contained in the pore spaces of the phosphogypsum stacks can be a source of groundwater contamination.

TABLE 12. ACTIVITY CONCENTRATIONS IN SEEPAGE WATER AND POND WATER AT A URANIUM MINE TAILINGS DAM IN BRAZIL

	Activity concentration (Bq/L)	
	Seepage water (n=20)	Pond water (n=1)
Ra-226	1.27 (0.33–5.0)	0.03
Ra-228	<1.30	0.38
Th-232	0.10 (0.01–0.7)	<0.015
U-238	2.89 (0.15–11)	0.22
Pb-210	5.3 (0.07–38)	<0.02

TABLE 13. ACTIVITY CONCENTRATIONS IN SURFACE WATER AND GROUNDWATER CONTAMINATED BY GOLD TAILINGS IN SOUTH AFRICA [27]

	Mean activity concentration (Bq/L)				
	Groundwater		Surface water		
	At edges of tailings dams	Surrounding aquifers	Mine discharge	Tailings return	Surrounding streams, lakes and dams
U-238	1.22	0.10	1.19	3.44	0.37
Ra-226	0.04	0.02	0.29	0.72	0.04

It is reported in Ref. [15] that groundwater samples taken from the vicinity of phosphogypsum stacks in Florida, USA exhibited activity concentrations of 0.002–0.043 Bq/L for ^{238}U, 0.002–0.021 Bq/L for ^{230}Th and 0.004–0.74 Bq/L for ^{226}Ra. It was noted that the phosphogypsum slurry is discharged into the stacks in an acidic state and that the characteristics of the pore water in the phosphogypsum stacks are determined by the phosphoric acid production process rather than the derived leachate from solids.

A ^{226}Ra activity concentration of 0.07–0.53 Bq/L was measured in leachates from phosphogypsum stockpiles and it was found that the leaching of radium may be slow in field conditions near the stockpiles [19]. However, considering the large quantity of phosphogypsum disposed of at individual sites, contamination of surface water and groundwater was considered to be of concern at that locality. In another study, the leachates from Togo phosphate rock were found to have ^{226}Ra concentrations of 0.23–0.55 Bq/L [28].

Few data are available on the concentrations of ^{210}Pb and ^{210}Po in leachate from NORM residue deposits. However, data from the monitoring of seepage at uranium tailings basins may be used as an indication of the potential levels of these radionuclides compared with radium. For neutral seepages, concentrations of ^{210}Pb and ^{210}Po tend to be <20% and <5% of the ^{226}Ra concentrations, respectively. In acidic seepages, concentrations of ^{210}Pb are similar to those of ^{226}Ra, while ^{210}Po concentrations are about 10%.[1] This will vary greatly by the type of NORM residue but nonetheless the uranium mining data provide an indication of what levels might be expected in NORM residues.

A study of contamination in a subsurface disposal area (SDA) at the Idaho National Engineering Laboratory (INEL) in south-eastern Idaho is reported in Ref. [29]. The SDA had been used since 1952 to dispose of radioactive wastes, some of which contained ^{238}U, ^{226}Ra and ^{210}Po. In this study, multi-media sampling and analysis indicated that the ^{210}Po levels detected in all media collected within and adjacent to the SDA were not statistically different from those concentrations detected in control area samples. The low concentrations of ^{210}Po detected in SDA media indicate that the disposal of radioactive wastes has not resulted in elevated ^{210}Po levels in water, surface soil, vegetation or small mammal tissues in the surrounding environment. The highest ^{210}Po concentration in soils (0.23 ± 0.06 Bq/g) was statistically greater than all other sampling locations (range of means was 0.03–0.1 Bq/g).

The data reviewed above suggest that although there is considerable variability from one residue deposit to another, the leachates are of roughly similar quality.

For uranium mining residues, this review suggests that the uranium activity concentrations of most leachates are within the range 0.5–170 Bq/L while the thorium activity concentrations are typically very much lower, of the order of 1.9 Bq/L or less. The activity concentrations in groundwater are generally within the range 0.005–55 Bq/L. Comparing the range of uranium concentrations in uranium mining residues summarized in Section 2.3.2 (0.1–17 Bq/g) with the corresponding range of leachate activity concentrations discussed above (0.5–170 Bq/L), it can be concluded that the range of leachate concentrations (in Bq/L) is roughly an order of magnitude higher than the range of uranium activity concentrations (in Bq/g) in the residues concerned. The higher leachate activity concentration of 170 Bq/L is for acidic waste rock leachate in Rabbit Lake uranium mine.

For non-uranium mining residues, the uranium activity concentrations of the leachate are generally within the range 0.1–2.4 Bq/L. Comparing this range of values with the range of 0.1–11 Bq/g in residues, it can be concluded that for every becquerel per gram of uranium activity concentration in the residue, the leachate concentration is less than 1 Bq/L.

For thorium, for every becquerel per gram of activity concentration in the residue, the leachate activity concentration in the leachate is generally less than 1 Bq/L.

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[1] Data obtained from groundwater monitoring data at Denison and Stanrock uranium mines, Elliot Lake, Canada.

2.3.4. Radionuclide transport in groundwater

The transport of radionuclides in groundwater has been studied extensively. Almost without exception, rocks and soils attenuate radionuclides when the concentrations of such radionuclides in the source exceed background levels.

The transport process of soluble materials in water generally has two components: advection and dispersion. Advection is transport along with the average pore water velocity. Dispersion is transport within the water, due to both molecular diffusion and small scale differences in flow speeds. When the pore water velocity is high, the dispersion component is not a significant contributor to the movement of the soluble chemicals. This is particularly true when the groundwater is moving in coarse sandy soil, where the pore water movement is relatively fast, and the retardation factor is relatively small. A range of hydraulic conductivities and permeabilities for various rocks and unconsolidated deposits can be found in Ref. [30]. The hydraulic conductivity K ranges from 0.1 m/s for coarse gravel to 10^{-13} m/s for metamorphic and igneous rock. It is expected that the hydraulic conductivity for residue materials with large particle sizes would have a wide range depending on profiles and surface conditions of the residue deposits; the nature, size range and size segregation of the materials; the pore volumes; compaction; and climatic conditions such as freeze–thawing, wetting and drying.

The retardation factor (a function of the soil–water distribution coefficient, K_d) for natural materials can vary over orders of magnitude and is dependent on numerous factors including pH, grain size, cation exchange capacity (CEC), redox conditions, salinity and other groundwater chemical characteristics. Without site-specific testing, applying a distribution coefficient in a contaminant transport model is subject to uncertainty.

The strong effect of the distribution coefficient, which reflects the partitioning of the radionuclide concentration between the solid and liquid phases, was demonstrated by a study at the site of the former White King uranium mine in the USA [31]. Although no actual distribution coefficients were calculated at the White King site, no down-gradient uranium was detected, even though pore water uranium concentrations in the stockpile were about 1000 Bq/L. The overburden stockpile pore water concentrations were less than 0.67 Bq/L, with a concentration of only 2.87 Bq/L immediately under the stockpile. Thus, with an inferred high distribution coefficient at this site, the uranium appears to be quite immobile. No radium was detected in the leachate, apparently having been contained within metal sulphates.

One example of groundwater migration from a uranium tailings deposit is the Nordic tailings deposit in Elliot Lake, Canada, where it was found that the migration of uranium and radium from acid leachates was greatly retarded in the groundwater aquifer below the deposit [22]. In the inner acid plume, radium and uranium distribution coefficients were calculated to be 70 and 2000 mL/g, respectively. It is likely that the migrating acid plume is being neutralized by alkalinity in soils. In the neutral outer zone of the plume, distribution coefficients for uranium were reported to be about 3600 mL/g.

Leaching rates for residue deposits imply that typically only a very small fraction of the inventory of NORM constituents is mobilized. As such, leaching is expected to continue for long time periods and retardation is likely to only provide a delay in the time at which the peak contaminant levels will reach the receptor. Therefore, for modelling purposes, the peak long term source concentrations in the absence of radioactive decay are likely to be similar to the concentration in the diluted source. This would not be the case if the radionuclides were permanently removed (that is, if the reactions were not reversible). This can occur with elements such as uranium which under reducing conditions may change its oxidation state and hence reduce the solubility (availability) of the mineral.

2.3.5. Distribution coefficients

In an experimental study to evaluate distribution coefficients for a soil–groundwater system, values were obtained for five types of soils that resemble NORM residue deposit materials [32]. Table 14 gives these values for U and Th.

At concentration ranges that occur in the leachate from residue deposits, the extent of thorium adsorption can be estimated from the soil pH. Studies have shown that the lowest thorium distribution coefficient was 20 mL/g for a measurement made on a pH 10 soil, while the largest thorium distribution coefficient was 170 000 mL/g for a measurement made on a silt–quartz soil of schist origin [18]. Based on the studies reviewed in Ref. [18], thorium distribution coefficients were estimated for various pH ranges and are given in Table 15.

TABLE 14. DISTRIBUTION COEFFICIENTS FOR U AND Th IN VARIOUS SOIL TYPES [32]

	Distribution coefficient (mL/g)	
	U	Th
Glacial till: Sandy silty clay with included fine to coarse gravel with very occasional cobbles.	46	24 000
Sand: Fine to medium sand	560	280
C.1.2: Slightly silty fine to coarse sand and mainly fine with occasional cobbles	46	5800
C.3: Medium to coarse sand with a lot of fines with occasional cobbles	900	280
C.6: Largely coarse grained sand with a lot of fines with numerous cobbles and small boulders	2200	5800

TABLE 15. RANGES OF DISTRIBUTION COEFFICIENTS FOR THORIUM [18]

	Distribution coefficient (mL/g)		
	pH 3–5	pH 5–8	pH 8–10
Minimum	62	1700	20
Maximum	6200	170 000	2000

A compilation of many studies on uranium partitioning indicates that pH and dissolved carbonate concentrations are the two most important factors influencing the adsorption behaviour of U(VI), the most mobile species of uranium [18]. Table 16 shows the range of estimated minimum and maximum distribution coefficients for uranium for partitioning between water and soil and crushed rock.

TABLE 16. RANGES OF DISTRIBUTION COEFFICIENTS FOR URANIUM [18]

	Distribution coefficient (mL/g)							
	pH 3	pH 4	pH 5	pH 6	pH 7	pH 8	pH	pH 10
Minimum	<1	0.4	0.25	100	63	0.4	<1	<1
Maximum	32	5000	160 000	1 000 000	630 000	250 000	7900	5

The indications in Ref. [18] are that the lower bound of the range of uranium distribution coefficients is based on values estimated for quartz. It is unlikely that actual distribution coefficients for U(VI) can be much lower than those represented by this lower bound.

Ref. [18] also cites field-derived uranium distribution coefficients for ^{238}U and ^{235}U from Ref. [33] and plots the derived values versus pH. As shown in Fig. 1, the uranium distribution coefficients vary from 1.2 to 34 000 mL/g over a pH range of approximately 3–6.7.

A summary of distribution coefficients for lead adsorption on soils from various studies is provided in Ref. [18] and is shown in Table 17. This information confirms that within the pH range of soils (4–11), lead adsorption increases (as does precipitation) with increasing pH.

In support of the US Department of Energy RESRAD modelling system for estimating doses, a great deal of information has been compiled, including information on distribution coefficients (see, for instance, Ref. [34]). As discussed in Section 3.1.2, distribution coefficients from the RESRAD data collection are compared with values based on the leachate and residue concentrations cited earlier in this Section to provide a basis for distribution coefficients that are considered to be reasonably representative of a typical NORM residue deposit.

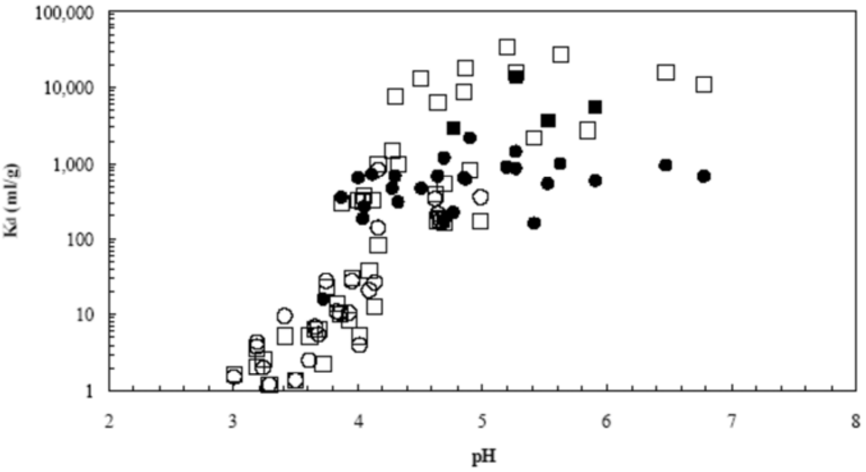

FIG. 1. Field-derived distribution coefficients plotted as a function of pore water pH for contaminated soil and pore water samples. Square and circle symbols represent field-derived distribution coefficients for ^{238}U and ^{235}U, respectively. Solid square and circle symbols represent minimum distribution coefficients for ^{238}U and ^{235}U, respectively, that were based on minimum detection limit values for the concentrations for the respective uranium isotopes in pore waters associated with the soil sample. (From Ref. [18])

TABLE 17. SUMMARY OF DISTRIBUTION COEFFICIENTS FOR LEAD ADSORPTION ON VARIOUS TYPES OF SOIL [18]

	pH	Distribution coefficient (mL/g)
Sand	4.5	280
Sand	5	1295
Sand	5.27	13 000–79 000
Medium sand	5.8	19
Sandy loam	7.5	3000
Sandy loam	8	4000
Fine sandy loam	8.7	59 000
Loam	7.3	21 000
Organic soil	5.5	30 000

2.4. Leachate characteristics and distribution coefficients for a representative NORM residue deposit

Taking account of the information presented in Section 2.3.3, the data in Table 18 are suggested as providing a reasonable radiological characterization of the leachate from a representative NORM residue deposit. Data are provided for both non-acid-generating and acid generating residues, although it is highly unlikely that a person would routinely depend on an acidic water supply for drinking purposes. Constituents such as pH, salinity, iron levels and metals would result in aesthetic characteristics that are likely to preclude the use of such water. Although some of the assumed leachate characteristics differ by up to a factor of about 10, this distinction is typically expected to reduce quite rapidly in most situations as alkalinity in the receiving environment neutralizes the leachate as it moves through soils and rocks. Although data are limited, it is thought that the characteristics presented in Table 18 are in the range of the upper 10th percentile.

TABLE 18. SOURCE CONCENTRATIONS FOR A REPRESENTATIVE URANIUM CONTAINING RESIDUE DEPOSIT

	Non-acidic	Acidic
U and Th content		
U_3O_8	0.01% (^{238}U: 1 Bq/g)	0.01% (^{238}U: 1 Bq/g)
ThO_2	0.01% (0.36 Bq/g)	0.01% (0.36 Bq/g)
Leachate Quality		
U	1 mg/L (^{238}U: 12 Bq/L)	10 mg/L (^{238}U: 123 Bq/L)
Ra-226	1 Bq/L	1 Bq/L
Th	<0.01 mg/L (^{232}Th: <0.041 Bq/L)	0.1 mg/L (^{232}Th: 0.41 Bq/L)
Pb-210[a]	0.2 Bq/L	1 Bq/L
Po-210[a]	0.05 Bq/L	0.1 Bq/L

[a] Based on seepage/groundwater monitoring data for a uranium mine.

Distribution coefficients for sand and clay can be established from RESRAD data [34] which in turn have been obtained from the literature or predicted by using concentration ratios. The values from RESRAD data for sands provide the lower bound for all the data cited and thus would give an unrealistically high estimate of groundwater concentration. The data for clay are not considered to represent the actual characteristics of NORM residue deposits. Table 19 shows the RESRAD values for sand and clay, together with values that have been estimated based on the leachate and residue concentrations cited earlier in this section and are considered to reasonably describe the distribution coefficients of a representative NORM residue deposit.

TABLE 19. DISTRIBUTION COEFFICIENTS FOR SAND, CLAY AND A REPRESENTATIVE NORM RESIDUE DEPOSIT

	Distribution coefficient (mL/g)		
	RESRAD geometric mean value		Conservatively derived value for a representative NORM residue deposit
	Sand	Clay	
Uranium	35	1600	50
Radium	500	9100	1250
Polonium	150	3000	2000
Lead	270	550	5000
Thorium	3200	5800	1000

2.5. Reported doses arising from NORM residue deposits

Public exposure via water pathways arising after the termination of uranium mining and processing in the former East Trunzig tailings management area in Germany is discussed in Ref. [20]. The exposure was dominated by the discharge of radioactivity carried by water both into receiving streams and underground. The assessment of the dose to people living nearby was based on the concentration of radionuclides in the Lerchenbach creek north of the tailings basin (see Table 9). The exposure pathway analyses performed on standard (hypothetical) scenarios in Ref. [20] showed that the highest individual dose was that received by a child up to one year of age, based on the consumption of breast milk and baby food prepared with the water. While this dose could exceed 1 mSv/a, it is stated that “the effective doses due to aquatic pathways are as a rule less than 1 mSv/a”. Moreover, Ref. [20] goes on to indicate that when site specific, rather than standard, scenarios are considered, “exposure pathways analysis almost always leads to effective doses due to the aquatic pathway of significantly less than 1 mSv/a”. The calculated dose was based on the exposure to radioactivity originating from tailings with an activity concentration of about 10 Bq/g. Simple scaling suggests that the effective doses due to aquatic pathways are less than 0.1 mSv/a for 1 Bq/g of activity concentration in the tailings.

Public exposure attributable to the contamination of water by gold mine tailings in South Africa was estimated from the measurements of radionuclide activity concentrations in surface water and groundwater reported in Section 2.3.3 [27]. The average radionuclide activity concentration in the tailings dams was about 1 Bq/g. The majority of the radiation dose came from uranium and ^{226}Ra. The concentrations of the other radionuclides were similar to the measured background levels and their contributions to the total dose were insignificant. It was found that 98% of the dose came from the direct ingestion of surface water and groundwater and from the ingestion of fish. The ingestion of milk, meat, grains and cereals, leafy vegetables, root vegetables and fruit accounted for the remaining 2%. Table 20 shows the estimated radiation doses received by an individual of age 17 or more via the various water-related pathways.

TABLE 20. INGESTION DOSES ARISING FROM THE CONTAMINATION OF WATER BY GOLD TAILINGS IN SOUTH AFRICA [27]

	Committed effective dose (mSv/a)				
	Minimum	10th percentile	Mean	90th percentile	Maximum
Direct ingestion of water					
Surface water	0.0002	0.0005	0.0027	0.0066	0.0131
Groundwater	0.0005	0.0016	0.0086	0.0254	0.0393
Ingestion of foodstuffs					
Fish	0.0008		0.019	0.034	0.086
Milk, meat, grains and cereals, leafy vegetables, root vegetables and fruit	0.000 030				0.002 17
Total (rounded)	0.0015		0.030	0.066	0.14

A risk assessment was conducted on the basis of the uranium concentration of surface water in the close vicinity of Barzava uranium mine in Romania [8]. The result indicated that the dose received by a person living in the vicinity of the residue deposit via the water, vegetable, meat, milk, and fish ingestion pathways was 0.051 mSv/a. The activity concentration of the residues varied between 0.5 and 1.5 Bq/g, depending on the location. Therefore, the calculated radiation dose corresponds to approximately 1 Bq/g of activity concentration in the residue.

The radiological impact of a uranium tailings dam at Poços de Caldas, Brazil is described in Ref. [25]. A worst case scenario is described, which includes the cessation of water treatment, the discharge of untreated water and the resulting exposure of people living on the Soberbo River. Conservative and non-conservative radiological assessments are presented, based on a dilution of the average and maximum seepage concentrations and the average and maximum ingestion rates. The ingestion of contaminated water was not identified as a realistic exposure pathway at that site and the inclusion of this pathway in the assessment therefore represented a conservative approach. It was noted that the conservative approach “may substantially overestimate the majority of the actual exposure to individuals” and that the non-conservative approach provided a more realistic scenario that “should not substantially overestimate the maximum exposure to individuals”. For the non-conservative (realistic) approach, annual committed effective doses of 0.62 and 0.48 mSv were estimated for children and adult members of the critical group, respectively. Higher annual doses, exceeding 1 mSv, were estimated for the conservative approach and were attributed primarily to ^{210}Pb and ^{210}Po. However, it was not evident that the simulations considered the removal of ^{210}Pb and ^{210}Po, which occurs through the immobilization resulting from the formation of insoluble compounds with sulphate ions, these being present in abundance from the ongoing acidification process. In a follow-up paper [26], the ingestion of contaminated water was again not considered to be a realistic scenario because “it cannot be considered a real exposure pathway for the studied scenario”. Using the realistic approach, it was estimated that the annual committed effective dose received by an adult in the critical group would be 0.35 mSv, while for the conservative approach the corresponding dose would be 1.1 mSv. Since the activity concentrations of radionuclides in the ^{238}U series were in the range 4.1–6.6 Bq/g in the residue deposit, the doses per unit activity concentration would be about 0.07 mSv for the realistic approach and about 0.2 mSv for the conservative approach.

3. DOSE CALCULATION

This section deals with the assessment of doses arising from exposure to a representative NORM residue deposit. For the purposes of this assessment, and for consistency with previous work, the modelling approach described in Ref. [5] was used as the basis of the dose estimates where appropriate and the realistic case parameters were used unless otherwise indicated. As discussed previously, the dose from radon is excluded from these calculations; however, Appendix II provides a separate assessment of the expected radon concentrations near the representative NORM residue deposit.

3.1. Characteristics of the representative NORM residue deposit

3.1.1. Physical characteristics

The modelling of bulk amounts of material such as NORM residue deposits requires many assumptions to be made about the quantity of material stored or disposed, the location, where it is placed relative to the public, and the characteristics of the environment which affect the pathway calculations, especially the water pathways. All these quantities are highly variable and site-specific. Thus, assumptions are needed to characterize a representative NORM residue deposit. For purposes of this assessment, the NORM residue deposit is assumed to have the following characteristics:

• Area of residue deposit:	100 000 m^2;
• Volume of deposit:	2 million m^3;
• Distance to nearest residence:	20 m;
• Thickness of contaminated zone:	20 m;
• Density of residue:	1.8 t/m^3;
• Mass of residue (calculated from above):	3.6 million t.

The dose assessment took into account exposure to ^{238}U, ^{234}U, ^{235}U, ^{230}Th, ^{226}Ra, ^{210}Pb, ^{210}Po, ^{232}Th, ^{228}Th and Ra^{228}. The activity concentration of each radionuclide (other than ^{235}U) was taken to be 1 Bq/g.[2]

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[2] The uranium isotopes were considered as being those constituting natural uranium (U$_{nat}$), defined as ^{238}U and ^{234}U in equilibrium (at 1 Bq/g each) and ^{235}U at its natural abundance ratio of 0.046.

3.1.2. Distribution coefficients

The distribution coefficients given in Table 19 for a representative mine residue deposit are considered be conservative (but nevertheless realistic) values and were therefore used in the assessment. The values for sandy soils given in Table 19 are not considered to be realistic and could be regarded as ‘worst case’ values that are not representative of the vast majority of situations.

3.1.3. Leach rates

For the water pathways, the model used in Ref. [5] assumes that all the radionuclides in the source are available for migration into an aquifer. The same assumption is used here, even though it is highly conservative and will result in an overestimation of the dose. The rate at which the radionuclides migrate from the source is determined by their respective distribution coefficients. The applicable equation, as given in Ref. [5], is as follows:

$$L_i = \frac{I}{\theta^{cz}\ z^{cz}\ R_i^{cz}}$$

where

L_i is the leach rate (a^{-1});
I is the infiltration rate (m/a);
θ^{cz} is the volumetric water content of the contaminated zone;
z^{cz} is the thickness of contaminated zone (m);
R_i^{cz} is the retardation factor for radionuclide i.

The retardation factor is given by:

$$R_i^{cz} = 1 + \frac{\rho^{cz}\ K_{d\,i}}{\theta^{cz}}$$

where

ρ is the density of the contaminated zone (g/cm^3);
$K_{d\,i}$ is the distribution coefficient for radionuclide i (cm^3/g).

The calculated leach rates are given in Appendix I.

3.1.4. Air emissions

Dust emissions for the NORM residue deposit are assumed to arise as a result of wind erosion. For this assessment, the basis for the dust emissions was a wind erosion equation using meteorological data for two exemplary Canadian sites, as described in Appendix II.

3.2. Exposure pathways

The pathways considered in this assessment were:

- Inhalation of airborne dust;
- External exposure from dust deposited on the ground at the residence;
- Ingestion of dust from dust deposited on the ground at the residence;

- Ingestion of garden and agricultural products from irrigation with contaminated groundwater;
- Ingestion of contaminated groundwater;
- Ingestion of fish obtained from contaminated surface water.

For consistency with the approach taken in Refs [3, 5], inhalation of radon was not formally included in the scope of the investigation. However, the results of separate calculations for airborne radon concentrations are given in Appendix II.

The evaluation of the pathways for surface water, groundwater and ingestion of garden and agricultural products are based on Scenario RW from Ref. [5] except that the presence of an unsaturated zone beneath the residue deposit was disregarded — this was done in order to be more conservative, since the presence of an unsaturated zone greatly reduces the final dose.

The groundwater pathway takes into account drinking water obtained by a nearby resident from a well contaminated by the residue deposit. The ingestion of garden and agricultural products pathway considers private garden food sources which are irrigated by the well water. The surface water pathway evaluates the dose originating from contaminated river or lake water, which is administered to the resident through the consumption of freshwater fish taken from the contaminated surface water. The eventual discharge of the groundwater into a surface water body such as a lake or pond dilutes the concentration of the radionuclides so that it is not necessary to consider the use of surface water explicitly. Consideration of only the ingestion of contaminated fish is taken to be sufficiently conservative.

It is assumed that radionuclides within the residue deposit are available for migration into the aquifer. The rate at which the radionuclides move is determined by a distribution coefficient approach, which takes into account leach rates and uses a retardation factor to determine the seepage and water concentrations of radionuclides.

The ingestion and inhalation of dust pathways are evaluated based on the scenarios RL-C and RL-A (C referring to children and A referring to adults) from Ref. [5]. The dose from the external exposure pathway was calculated by using the soil concentration (including airborne deposition to the soil and leaching factors) along with a dose conversion factor for exposure to contaminated soil. The scenarios used for each pathway are summarized in Table 21.

TABLE 21. EXPOSURE SCENARIOS

	Scenario from Ref. [5]		Scenario developed separately
	RW	RL-A, RL-C	
Inhalation of dust		×	
Ingestion of dust		×	
External exposure			×
Ingestion of garden and agricultural products	×		
Ingestion of groundwater	×		
Ingestion of fish	×		

3.3. Exposure and uptake

The model used for this dose assessment considers a child aged 1–2 years and an adult aged from 17 upwards. The dietary parameters consistent with these age groups are those used in Ref. [5]. The model required input parameters for the ingestion of drinking water, leafy vegetables, non-leafy vegetables, fruit, fish, and dust. In Ref. [5], it is assumed that, for the realistic scenario, only 25% of the total amount of water and food consumed is affected by radionuclides. The resulting parameters are outlined in Table 22.

TABLE 22. INGESTION PARAMETERS

	Consumption (kg/a)	
	Child, 1–2 years	Adult
Drinking water	100	350
Leafy vegetables	6	13
Non-leafy vegetables	17	40
Fruit	17	40
Fish	0.6	1.5
Dust	0.025	0.025

For the inhalation and ingestion of dust, additional factors need to be considered. Assuming moderate physical activity, the breathing rate for adults was taken to be 1.2 m^3/h, while the breathing rate for children was given as 0.22 m^3/h. A realistic period for which residents are exposed to dust from a facility was taken to be 1000 h/a. A value of 0.01 was chosen as the dilution factor of the dust to take into account the fact that only a part of the soil or dirt will consist of radioactive material.

Overall, the key parameters related to the physical characteristics of the residue and residential area were chosen according to the so-called ‘realistic’ scenario of Ref. [5] where possible. The values used were chosen to reflect hypothetical reference situations, thus ensuring the (reasonable) applicability of the calculated doses to worldwide situations.

3.4. Dose

Since only radionuclides of artificial origin are considered in Ref. [5], parameter values related to the radionuclides of natural origin considered in this report were obtained elsewhere. The distribution coefficients used in this assessment are the conservatively derived values given in Table 19. Inhalation and ingestion dose coefficients were those recommended by the International Commission on Radiological Protection (ICRP) [35]. Root transfer factors (transfer from soil to plant) used in the calculation of irrigation transfer factors for plants were taken from Refs [36, 37]. Freshwater-to-fish transfer factors came from Ref. [37] and other scientific literature, as indicated in Appendix I.

3.4.1. *Groundwater, ingestion of garden and agricultural products, and surface water pathways*

The dose calculations for these three pathways were based on Scenario RW in Ref. [5]. Once the leach rates had been determined (see Section 3.1.3), the radionuclide concentration in the seepage could be determined using the following equation from Ref. [5]:

$$C_i^s = \frac{M\ c_i\ L_i}{U^s}$$

where

C_i^s is the radionuclide concentration in the seepage (Bq/m^3);
M is the total mass of contaminated material (g);
c_i is the activity concentration of radionuclide i in the contaminated material (Bq/g);
U^s is the volume of seepage through the contaminated zone (m^3/a);
L_i is the leach rate for radionuclide i (a^{-1}).

The value of U^s is given by:

$$U^s = I\ A^{cz}$$

where

I is the infiltration rate (m/a);
A^{cz} is the surface area of the contaminated zone (m^2).

The resulting radionuclide concentrations in the seepage, along with the (nominal) concentrations inferred from measurements of NORM residue deposits (see Table 18) are provided in Table 23. A comparison of the calculated leachate concentrations with the nominal values of both the non-acidic and acidic residue deposits suggests that, other than for uranium in an acidic residue deposit, the estimated leachate concentrations are conservative. The uranium concentration of 124 Bq/L for an acidic residue deposit is exceptional and, as shown in Table 11, will decrease with distance from the residue deposit as the soil neutralizes the acid.

TABLE 23. RADIONUCLIDE CONCENTRATIONS IN SEEPAGE

	Concentration (Bq/L)		
	Based on measurements (see Table 18)		Calculated (using realistic distribution coefficients)
	Non-acidic	Acidic	
U-238	12	123	20
Th-230	–	–	1.0
Ra-226	1	1	0.80
Pb-210	0.2	1	0.20
Po-210	0.05	0.1	0.50
Th-228	–	–	1.0
Th-232	<0.041	0.41	1.0
Ra-228	–	–	0.80

Assuming that there is no unsaturated zone between the contaminated zone and the aquifer, the radionuclide concentration in the well water is determined by considering the dilution with the groundwater volume flowing underneath the contaminated zone. The groundwater volume is calculated using the following equation from Ref. [5]:

$$U^{gw} = z^{gw}\ w^{gw}\ v^{gw}\ p^{gw}$$

where

U^{gw} is the volume of groundwater flowing underneath the contaminated zone (m^3/a);
z^{gw} is the thickness of aquifer (m);
w^{gw} is the width of the contaminated zone perpendicular to the flow of the aquifer (m);
v^{gw} is the pore water velocity of the groundwater (m/a);
p^{gw} is the effective porosity of the aquifer.

The concentration of radionuclide i in the well water is then given by the following equation (adapted from Ref. [5]):

$$C_i^w = C_i^s \frac{U^s}{U^{gw} + U^s}$$

where

C_i^w is the radionuclide concentration in the well water (Bq/m^3);
C_i^s is the radionuclide concentration in the seepage (Bq/m^3);
U^s is the volume of seepage (m^3/a);
U^{gw} is the volume of groundwater (m^3/a).

The ingestion dose from the consumption of well water can then be calculated by multiplying C_i^w by the ingestion parameters and ingestion dose coefficients previously discussed.

The process used to calculate the intake of fruit and vegetables irrigated by contaminated well water was very similar, following the same steps as the method for drinking water, but multiplying in an extra factor known as the ‘transfer factor’ at the last step. The transfer factors were calculated using the root transfer factors from soil to plant as shown in the following equation (from Ref. [5]).

$$f_t = \frac{I_{rr}\ f_r\ T_f\ (1-e^{-\lambda_w t_e})}{Y_w\ \lambda_w} + \frac{I_{rr}\ (1-f_r)\ f_{t\,i}\ (1-e^{-L_i t_e})}{\rho^e\ L_i}$$

where

f_t is the transfer factor from water to plants (m^3/kg);
I_{rr} is the irrigation rate (m/a);
f_r is the fraction of deposited radionuclides retained on the vegetation (0.25);
T_f is the foliage to food transfer coefficient (0.1 for fruit and non-leafy vegetables and 1 for leafy vegetables)
λ_w is the weathering removal constant ($20\ a^{-1}$);
t_e is the period of exposure during the growing season (0.17 a for fruit and non-leafy vegetables and 0.25 a for leafy vegetables);
Y_w is the wet weight crop yield ($0.7\ kg/m^2$ for fruit and non-leafy vegetables and $1.5\ kg/m^2$ for leafy vegetables);
$f_{t,i}$ is the root transfer factor for radionuclide i;

L_i is the leach rate for radionuclide i (a^{-1});
ρ^e is the effective surface density of soil (225 kg/m^2).

The surface water pathway calculation follows the same steps as above, except that U^{gw} is replaced by the flow rate of the river, which is taken to be 1.58×10^8 m^3/a [5]. The resulting concentration value is then multiplied by the ingestion dose coefficients, the appropriate ingestion parameters and the freshwater-to-fish transfer factors to yield the dose.

3.4.2. Ingestion and inhalation of dust

Calculations for the ingestion and inhalation of dust also made use of equations given in Ref. [5]. Because all the radionuclides of interest remain in equilibrium, it was not necessary to include the time factors within the equations. As previously discussed, these calculations were based on scenarios RL-C and RL-A, and made use of the values given in Ref. [5]. The annual committed effective dose for ingestion of dust was calculated using the following equation (adapted from Ref. [5]):

$$E_{ing,C} = e_{ing}\ q\ f_d\ \ f_c\ A$$

where

$E_{ing,C}$ is the annual committed effective dose for ingestion of dust (mSv/a);
e_{ing} is the effective dose coefficient for ingestion (mSv/Bq);
q is the quantity ingested per year (g/a);
f_d is the dilution factor;
f_c is the concentration factor of specific activity in the fine fraction;
A is the activity concentration (Bq/g).

The annual committed effective dose for inhalation of dust was calculated using the following equation (adapted from Ref. [5]):

$$E_{inh,C} = e_{inh}\ t_e\ f_d\ \ f_c\ C_{dust}\ V\ A$$

where

$E_{inh,C}$ is the annual committed effective dose for inhalation of dust (mSv/a);
e_{inh} is the effective dose coefficient for inhalation (mSv/Bq);
t_e is the annual exposure period (h/a);
C_{dust} is the effective dust concentration in the air (g/m^3);
V is the breathing rate (m^3/h).

3.4.3. External exposure

External exposure is assumed to consist of the gamma radiation arising from dust transported from the NORM residue deposit and deposited on the ground at the nearby residence (assuming that the soil is contaminated to a depth of 1 cm). The first step in determining the dose from external exposure is to calculate the deposition rate, according to the following equation:

$$R_{dep} = C_{air}\ V_{set}\ \ CF_t$$

where

R_{dep} is the deposition rate (µg·m^{-2}·a^{-1});
C_{air} is the air concentration (µg/m^3);
V_{set} is the settling velocity (m/s);
CF_{t} is the conversion factor for time (31 557 600 s/a).

The second step is to calculate the leaching coefficient for each radionuclide at 1 cm depth using the following equation:

$$LC_i = \frac{V_w\ CF_d}{d\left(1+\dfrac{\rho\ K_{d\ i}}{\theta}\right)}$$

where

LC_i is the leaching coefficient for radionuclide i (a^{-1});
V_w is the velocity of water percolation downward through soil (cm/d);
CF_{d} is the conversion factor for time (365.25 d/a);
d is the depth of the soil zone of interest (assume 1 cm);
ρ is the density of the soil (g/cm^3);
$K_{d\ i}$ is the distribution coefficient for radionuclide i (mL/g);
θ is the soil water content (mL/cm^3).

The deposition rate and leaching coefficient are used to calculate the soil concentration for each radionuclide, according to the following equation:

$$C_{soil\ i} = \frac{R_{dep}\ CF\left(1-e^{-LC_i\ t}\right)}{d\ \rho\ LC_i}$$

where

$C_{\mathrm{soil}\ i}$ is the concentration of radionuclide i in soil (µg/g);
R_{dep} is the deposition rate (µg·m^{-2}·a^{-1});
CF is the conversion factor from m^2 to cm^2 (0.0001 m^2/cm^2);
LC_i is the leaching coefficient for radionuclide i (a^{-1});
t is the time period of interest (assume 100 years);
d is the depth of the soil zone of interest (assume 1 cm);
ρ is the density of the soil (g/cm^3).

Finally, the dose from external exposure for each radionuclide is calculated using the following equation:

$$D_{ext\ i} = C_{soil\ i}\ A_i\ CF_{mass}\ DF_{ext\ i}\ CF_{dose}$$

where

$D_{ext\ i}$ is the annual effective dose from external exposure to radionuclide i (mSv/a);
$C_{soil\ \mathrm{i}}$ is the concentration of radionuclide i in soil (µg/g);

A_i is the activity concentration of radionuclide i (Bq/g);
CF_{mass} is the conversion factor for mass (10^{-6} g/μg);
$DF_{ext\ i}$ is the dose conversion factor for radionuclide i for contaminated soil at 1 cm depth (Sv/a per Bq/g);
CF_{dose} is the conversion factor for dose (10^3 mSv/Sv).

3.4.4. *Results*

The models described in Ref. [5] simulate the chemistry of a NORM residue deposit through a 'lumped' surrogate parameter (K_d). Therefore, the model used to estimate the doses is not capable of dealing specifically with detailed geochemical issues such as acid generation. However, as illustrated in Table 23, the estimated seepage concentrations compare well with the nominal concentrations inferred from measurements of various NORM residue deposits.

The doses from each pathway included in this assessment are given in Table 24. The corresponding doses for only the ^{238}U series radionuclides are given in Table 25.

TABLE 24. RESULTS OF DOSE CALCULATIONS FOR RADIONUCLIDES IN THE ^{238}U AND ^{232}Th DECAY SERIES, EACH AT AN ACTIVITY CONCENTRATION OF 1 Bq/g

	Annual committed effective dose (mSv)	
	Child, 1–2 a	Adult
Inhalation of dust	0.000 34	0.000 65
Ingestion of dust	0.011	0.0018
External exposure	0.0056	0.0043
Ingestion of garden and agricultural products	0.000 052	0.000 027
Ingestion of groundwater	0.24	0.19
Consumption of fish	0.000 076	0.000 039
Total	0.26	0.20

TABLE 25. RESULTS OF DOSE CALCULATIONS FOR ONLY RADIONUCLIDES IN THE ^{238}U DECAY SERIES, EACH AT AN ACTIVITY CONCENTRATION OF 1 Bq/g

	Annual committed effective dose (mSv)	
	Child, 1–2 a	Adult
Inhalation of dust	0.000 13	0.000 25
Ingestion of dust	0.0070	0.0012
External exposure	0.0025	0.0019
Ingestion of garden and agricultural products	0.000 034	0.000 021
Ingestion of groundwater	0.16	0.15
Consumption of fish	0.000 047	0.000 027
Total	0.17	0.15

4. DISCUSSION AND CONCLUSIONS

4.1. Conservatism in the dose calculations

It is clear from the description of the dose calculations in Section 3 that the approach taken has been conservative, although care has been taken to avoid highly unrealistic assumptions. As shown in Table 23, this conservatism is evident when comparing the estimated radionuclide concentrations in seepage with the nominal radionuclide concentrations inferred from measurements of various non-acid-generating NORM residue deposits. Furthermore, the calculated activity concentrations in groundwater appear to be conservative when compared against those measured at sites in Serbia and South Africa, as shown in Table 26.

TABLE 26. ACTIVITY CONCENTRATIONS IN GROUNDWATER CONTAMINATED BY NORM RESIDUE DEPOSITS

	Activity concentration (Bq/L)		
	Serbia (Table 10)	South Africa (Table 12)	Calculated in this report
U-238	0.005–0.1	1.22	2.7
Ra-226	0.05–0.6	0.04	0.1

The effects of groundwater dilution, natural attenuation and dispersion have not been taken into account in the dose calculations. In developing legislation for waste disposal, exposure scenarios have typically been based on the assumption that dilution by groundwater between the waste source and receptor will occur. Inert wastes or non-toxic wastes in many jurisdictions are defined as wastes whose leachate has concentrations of elements <10 to 100 times drinking water criteria[3]. Thus, a reasonable exposure scenario would assume dilution and, based on experience, a minimum dilution of 10–100 would be defensible. In addition to dilution, natural attenuation and dispersion would reduce migration rates and concentrations.

The data summarized in Section 2.5 suggest that for actual situations, as opposed to hypothetical scenarios, the doses per unit activity concentration received by individuals living near NORM residue deposits are likely to be lower than those calculated for the representative residue deposit.

4.2. Comparison with doses determined from measured radionuclide concentrations in water

Since more than 90% of the doses calculated for the representative NORM residue deposit are received via water pathways, it is useful to compare the values calculated for water pathways with those obtained from on-site measurements of radionuclides in water. The results of such a comparison are shown in Table 27. The values obtained from on-site measurements fall within a relatively narrow range that lies well below the calculated values. This provides further evidence of the conservative nature of the dose calculations.

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[3] For instance, the US Environmental Protection Agency originally proposed a dilution factor of 10 and later increased it to 100 [48].

TABLE 27. COMPARISON OF CALCULATED DOSES WITH THOSE OBTAINED FROM DIRECT MEASUREMENTS IN THE FIELD, WATER PATHWAYS

	Annual committed effective dose per unit activity concentration (mSv/a per Bq/g)
East Trunzig former uranium mine, Germany	<0.1
Barzava uranium mine, Romania	~0.05
Gold mines, South Africa	0.0015–0.14[a]
Uranium mine, Brazil	0.07
Calculated (representative NORM residue deposit)	
Adult	0.19
Child, 1–2 a	0.24

[a] The mean and 90th percentile values were 0.030 and 0.066 mSv per Bq/g, respectively.

4.3. Acid generating NORM residue deposits

As indicated in this report, there is a potential for some NORM residue deposits to be acid generating. The Nordic tailings management area in Elliot Lake, Canada is one such example. However, as illustrated by the data in Table 11, the pH increases and the concentrations of radionuclides in groundwater decrease rapidly with increasing distance as the soil neutralizes the acid. Moreover, it is unreasonable and extremely conservative to assume that people would routinely drink an acidic water supply, as it would not meet minimal drinking water quality standards for conventional parameters such pH, total dissolved solids, iron, heavy metals and aesthetic qualities (taste and colour for example) and would be unpleasant to drink.

4.4. Disequilibrium in NORM residue deposits

Since some of the NORM residue deposits will not be in equilibrium as a result of chemical processing, it is useful to consider the dose contributions from individual radionuclides. This is illustrated in Table 28, which shows the contributions of individual radionuclides to the dose arising from the groundwater pathway. For a deposit such as phosphogypsum, which contains essentially only ^{226}Ra and its decay products, the dose per unit activity concentration from the groundwater pathway is expected to be less than 0.1 mSv/a.

4.5. Sensitivity of the results to the values of the distribution coefficients

Given that the values of the distribution coefficients reported in the literature vary over wide ranges, the implications of using lower (more conservative) values have been considered. The distribution coefficients provided in the RESRAD data collection for sandy material [34], the values of which are given in Table 19, provide the lower bound for all the data cited and can therefore be considered as being representative of a ‘worst case’ (and unrealistic) situation. The intermediate and final results of the calculations (for the case where all radionuclides in the uranium and thorium decay series are present in the residue at an activity concentration of 1 Bq/g) are given in Appendix III. The implications of using these ‘worst case’ values are that the annual committed effective dose rises from 0.26 to 1.3 mSv in the case of a child and from 0.20 to 0.75 mSv in the case of an adult. So, even in this unrealistic situation, the dose is still of the order of 1 mSv or less.

TABLE 28. DOSES FROM THE GROUNDWATER PATHWAY, BY RADIONUCLIDE

	Annual committed effective dose (mSv)	
	Child, 1–2 a	Adult
U_{nat}	0.0705	0.0927
Th-230	0.005 65	0.0101
Ra-226	0.0106	0.0108
Pb-210	0.009 93	0.006 66
Po-210	0.0607	0.0290
Th-228	0.0142	0.006 61
Th-232	0.006 21	0.0111
Ra-228	0.0629	0.0266
Total, all radionuclides	0.24	0.19
Total, only Ra-226, Pb-210, Po-210	0.081	0.046

4.6. Overall conclusions

It is evident from Tables 24 and 25 that the total dose per unit activity concentration, as determined from the calculations, is well below 1 mSv/a and that the majority of this dose (>90%) comes from the groundwater ingestion pathway. Even if the results of the dose calculations were in error (too low) by a factor of 300%, the dose would still not exceed 1 mSv.

The doses in real situations will depend on the prevailing site conditions. The dose assessment described in this report does not cover all potentially occurring individual site parameters. Nevertheless, the assessment is considered to be sufficiently conservative to cover the vast majority of real cases and, indeed, the doses determined from actual measurements in the field have all been found to be significantly lower than the calculated doses. On this basis, the calculated doses are considered appropriate also for sites where some of the relevant site factors are more unfavourable then assumed here.

Although the dose received from inhalation of radon has not been included in the assessment, separate calculations show that the activity concentration in the outdoor air is expected to be about 10–20 Bq/m^3 (see Appendix II). Comparing this result with the normal variations in indoor and outdoor radon concentrations, it can be concluded that the inhalation of radon emitted from the representative NORM residue deposit is not a significant exposure pathway.

In view of the results of the dose calculations for a representative NORM residue deposit and the supporting information based on actual measurements in the field, as summarized in Table 27, it is concluded that the dose received in a year by an individual living next to a bulk mine residue deposit will, in all reasonable situations, be significantly less than 1 mSv/a per unit activity concentration (in becquerels per gram) in the residue.

APPENDIX I. DOSE CALCULATION

This Appendix provides the parameters and intermediate results of the dose calculations for each pathway considered in this assessment.

I.1. Input parameters

The key input parameters for each of the pathways are provided in Table 29.

TABLE 29. KEY INPUT PARAMETERS

	Value	Reference
Activity concentration	1 Bq/g	Assumed
Deposit area	100 000 m^2	Assumed
Deposit volume	2 million m^3	Assumed
Distance to residential area	20 m	Assumed
Aquifer flow rate	1 m/d	Assumed
Volumetric water content of NORM residue deposit	0.16	[5]
Density of NORM residue deposit	1.8 g/cm^3	[5]
Thickness of NORM residue deposit	20 m	Assumed
Infiltration rate	0.2 m/a	[5]
Total mass of NORM residue deposit	3.6 million t	Calculated
Thickness of aquifer	5 m	[5]
Width of NORM residue deposit perpendicular to aquifer flow	100 m	[5]
Pore water velocity of groundwater	1000 m/a	[5]
Effective porosity of aquifer	0.25	[5]
Flow rate of river	1.58×10^8 m^3/a	[5]
Fraction of deposited radionuclides retained on vegetation	0.25	[5]
Foliage to food transfer coefficient		
Fruit and non-leafy vegetables	0.1	[5]
Leafy vegetables	1	[5]
Weathering removal constant for vegetation	20	[5]
Time of exposure during growing season		
Fruit and non-leafy vegetables	0.17 a	[5]
Leafy vegetables	0.25 a	[5]
Wet weight crop yield		
Fruit and non-leafy vegetables	0.7 kg/m^3	[5]
Leafy vegetables	1.5 kg/m^3	[5]
Effective surface density of soil	225 kg/m^2	[5]
Irrigation rate	0.2 m/a	[5]
Seepage rate through contaminated zone	20 000 m^3/a	Calculated
Groundwater flow rate underneath area of contaminated zone	125 000 m^3/a	Calculated

TABLE 29. KEY INPUT PARAMETERS (contd)

	Value	Reference
Ingestion parameters, child		
Drinking water	100 kg/a	[5]
Leafy vegetables	6 kg/a	[5]
Non-leafy vegetables	17 kg/a	[5]
Fruit	17 kg/a	[5]
Fish	0.6 kg/a	[5]
Ingestion parameters, adult		
Drinking water	350 kg/a	[5]
Leafy vegetables	13 kg/a	[5]
Non-leafy vegetables	40 kg/a	[5]
Fruit	40 kg/a	[5]
Fish	1.5 kg/a	[5]
Ingested quantity of dust		
Child	25 g/a	[5]
Adult	25 g/a	[5]
Dilution factor	0.01	[5]
Concentration factor for ingestion of dust	2	[5]
Exposure period spent outside	1000 h/a	[5]
Concentration factor of specific activity in fine fraction	4	[5]
Effective dust concentration in air	0.0001 g/m^3	[5]
Breathing rate		
Child	0.22 m^3/h	[5]
Adult	1.2 m^3/h	[5]
Settling velocity	0.0014 m/s	[37]
Air concentration	15 µg/m^3	Mean of range in Appendix II
Velocity of water percolation downward through soil	0.1 cm/d	Assumed
Bulk soil density	1.6 g/cm^3	[38]
Soil water content	0.2 ml/cm^3	[39]
Depth of soil zone of interest (external pathway)	1 cm	Assumed
Time period of interest (external pathway)	100 a	Assumed

I.2. Intermediate and final dose results

The intermediate results for each pathway are given in Table 30.

TABLE 30. INTERMEDIATE RESULTS OF DOSE CALCULATIONS

	U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-228	Th-232	Ra-228	Sum
Coefficients									
Distribution coefficient (Table 19) (mL/g)	50	1000	1250	5000	2000	1000	1000	1250	
Ingestion dose coefficient (μSv/Bq) [35]									
Child	0.26	0.41	0.96	3.6	8.8	1.03	0.45	5.7	
Adult	0.10	0.21	0.28	0.69	1.2	0.137	0.23	0.69	
Dust inhalation coefficient (μSv/Bq) [35]									
Child	55.20	35	29	18	14	139.2	50	48	
Adult	17.79	14	9.5	5.6	4.3	43.4	25	16	
External dose coefficient (Sv/a per Bq/g) [38]									
Child	5.95e-6	1.53e-7	7.06e-4	5.42e-7	3.49e-9	5.23e-4	7.61e-8	3.92e-4	
Adult	4.58e-6	1.18e-7	5.43e-4	4.17e-7	2.68e-9	4.03e-4	5.85e-8	3.01e-4	
Ingestion of groundwater									
Retardation factor	564	11 300	14 100	56 300	22 500	11 300	11 300	14 100	
Leach rate (a^{-1})	1.11e-4	5.56e-6	4.44e-6	1.11e-6	2.78e-6	5.56e-6	5.56e-6	4.44e-6	
Activity concentration in seepage (Bq/m^3)	20 000	1000	800	200	500	1000	1000	800	
Activity concentration in well water (Bq/m^3)	2750	138	110	27.6	69.0	138	138	110	
Committed effective dose (mSv/a)									
Child	0.070	0.006	0.011	0.010	0.061	0.014	0.006	0.063	0.241
Adult	0.093	0.010	0.011	0.007	0.029	0.007	0.011	0.027	0.194

TABLE 30. INTERMEDIATE RESULTS OF DOSE CALCULATIONS (contd)

	U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-228	Th-232	Ra-228	Sum
Ingestion of garden and agricultural products									
Root transfer factor [36], [37]	1.00e-3	3.30e-4	1.10e-2	1.00e-3	1.20e-4	3.30e-4	3.30e-4	1.10e-2	
Overhead irrigation transfer factor									
Leafy vegetables	1.66e-3	1.66e-3	1.66e-3	1.66e-3	1.66e-3	1.66e-3	1.66e-3	1.66e-3	
Non-leafy vegetables	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	
Fruit	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	3.45e-4	
Committed effective dose (mSv/a)									
Leafy vegetables									
Child	7.00e-6	5.62e-7	1.05e-6	9.86e-7	6.03e-6	1.41e-6	6.16e-7	6.25e-6	2.39e-5
Adult	5.70e-6	6.23e-7	6.66e-7	4.10e-7	1.78e-6	4.07e-7	6.83e-7	1.64e-6	1.19e-5
Non-leafy vegetables									
Child	4.14e-6	3.32e-7	6.24e-7	5.83e-7	3.56e-6	8.34e-7	3.64e-7	3.70e-6	1.41e-5
Adult	3.66e-6	4.00e-7	4.28e-7	2.63e-7	1.14e-6	2.61e-7	4.38e-7	1.06e-6	7.65e-6
Fruit									
Child	4.14e-6	3.32e-7	6.24e-7	5.83e-7	3.56e-6	8.34e-7	3.64e-7	3.70e-6	1.41e-5
Adult	3.66e-6	4.00e-7	4.28e-7	2.63e-7	1.14e-6	2.61e-7	4.38e-7	1.06e-6	7.65e-6
Fish consumption									
Activity concentration in surface water (Bq/m^3)	2.53	0.127	0.101	0.0254	0.0634	0.127	0.127	0.101	
Water-to-fish transfer factor [36][40][41]	2.00e-5	1.00e-4	5.00e-5	3.00e-4	5.00e-5	1.00e-4	1.00e-4	5.00e-5	

TABLE 30. INTERMEDIATE RESULTS OF DOSE CALCULATIONS (contd)

	U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-228	Th-232	Ra-228	Sum
Fish consumption (contd)									
Committed effective dose (mSv/a)									
Child	7.78e-6	3.12e-6	2.92e-6	1.64e-5	1.67e-5	7.84e-6	3.42e-6	1.73e-5	7.56e-5
Adult	7.30e-6	3.99e-6	2.13e-6	7.88e-6	5.71e-6	2.61e-6	4.38e-6	5.25e-6	3.92e-5
Ingestion of dust									
Committed effective dose (mSv/a)									
Child	1.28e-4	2.05e-4	4.80e-4	1.80e-3	4.40e-3	5.15e-4	2.25e-4	2.85e-3	1.06e-2
Adult	4.81e-5	1.05e-4	1.40e-4	3.45e-4	6.00e-4	6.85e-5	1.15e-4	3.45e-4	1.77e-3
Inhalation of dust									
Committed effective dose (mSv/a)									
Child	4.86e-5	3.08e-5	2.55e-5	1.58e-5	1.23e-5	1.22e-4	4.40e-5	4.22e-5	3.42e-4
Adult	8.54e-5	6.72e-5	4.56e-5	2.69e-5	2.06e-5	2.08e-4	1.20e-4	7.68e-5	6.51e-4
External exposure									
Committed effective dose (mSv/a)									
Child	2.71e-6	5.08e-7	2.45e-3	2.15e-6	1.29e-8	1.74e-3	2.53e-7	1.36e-3	5.55e-3
Adult	2.08e-6	3.91e-7	1.88e-3	1.65e-6	9.94e-9	1.34e-3	1.95e-7	1.05e-3	4.27e-3
Total dose, child									0.26
Total dose, adult									0.20

APPENDIX II. RADIONUCLIDE CONCENTRATIONS IN AIR

Radioactivity releases to the atmosphere are subject to dilution in air and are dispersed by the prevailing winds. For the present assessment, a standard air dispersion model was used to provide the linkage between a source of airborne radioactivity and the concentrations expected at a receptor. In general terms, the relation between a release to the atmosphere and the annual average concentration of radioactivity in the air at a certain distance from the release point can be assessed using the following equation:

$$C_x = K_h\ R$$

where:

C_x is the annual average radionuclide concentration at a distance x from release point (Bq/m^3);
K_h is the dilution factor for a release point at a height h above the ground and at a distance x from the receptor (s/m^3);
R is the annual average release rate from the source (Bq/s).

For the present analysis, the dilution factor K_h has been estimated using the ISCST3 model of the US Environmental Protection Agency [42]. The model is a steady-state Gaussian plume model recommended for use in the above situation [43] and is widely used in mining applications to check for compliance with air quality standards.

Meteorology can have an effect on both the production of air emissions through wind erosion and the manner in which the emissions are dispersed in the atmosphere. Some of the meteorological factors that influence air quality include:

- Atmospheric stability;
- Wind speed;
- Wind direction;
- Mixing height;
- Temperature.

The ISC model accepts hourly meteorological data records to define the conditions for plume rise, transport and dispersion. The model estimates the concentration (or deposition) value for each source–receptor combination, for each hour of input meteorology, and calculates short-term averages, such as one hour, eight hour and 24 hour averages. The hourly averages can also be combined into longer averages (monthly, seasonal, annual or period). For present purposes, the annual values were used.

II.1. Dust emissions

For present purposes, wind erosion from the surface of the waste rock pile has been estimated using a wind erosion factor based on the following equation [44]:

$$E = 1.9\left(\frac{s}{1.5}\right)\left(\frac{365-p}{235}\right)\left(\frac{f}{15}\right)$$

where

E is the wind erosion factor ($kg \cdot d^{-1} \cdot ha^{-1}$);
s is the silt content of the aggregate (%);
p is the number of days with >0.25 mm (>0.01 inch) of precipitation per year;
f is the percentage of time that the unobstructed wind speed exceeds 5.4 m/s (12 miles/h) at the mean pile height.

The parameters used to estimate the wind release rates were $s = 5$, $p = 144$ and $f = 21\%$, which leads to an emission rate of approximately 1 $\mu g \cdot m^{-2} \cdot s^{-1}$ or, for the entire residue deposit, 8.338 $kg \cdot d^{-1} \cdot ha^{-1}$.

Meteorological conditions can vary widely from place to place. To take account of this inherent variability, two data sets from previous analyses for sites in Canada were used, namely a regional meteorological data set for Sudbury, Ontario (1992–1996) and meteorological data for McClean Lake, Saskatchewan (2001). For illustrative purposes, the 10 ha representative NORM residue deposit was modelled as an area source with dimensions of 316 m × 316 m assuming an area emission rate of 1 $\mu g \cdot m^{-2} \cdot s^{-1}$. Figures 2 and 3 show the corresponding concentration lines for the meteorological conditions at Sudbury and McLean Lake, respectively. Figure 4 is a plot of the unit dilution factors for both locations along the line of maximum concentration (poorest dispersion). The annual average wind speeds are 3.1 m/s for the McClean site and 3.6 m/s for the Sudbury site. The ISC model at the two ‘reference’ sites gave concentrations in the range 10–20 $\mu g/m^3$ and the mid-point of this range, 15 $\mu g/m^3$, was used in the calculations.

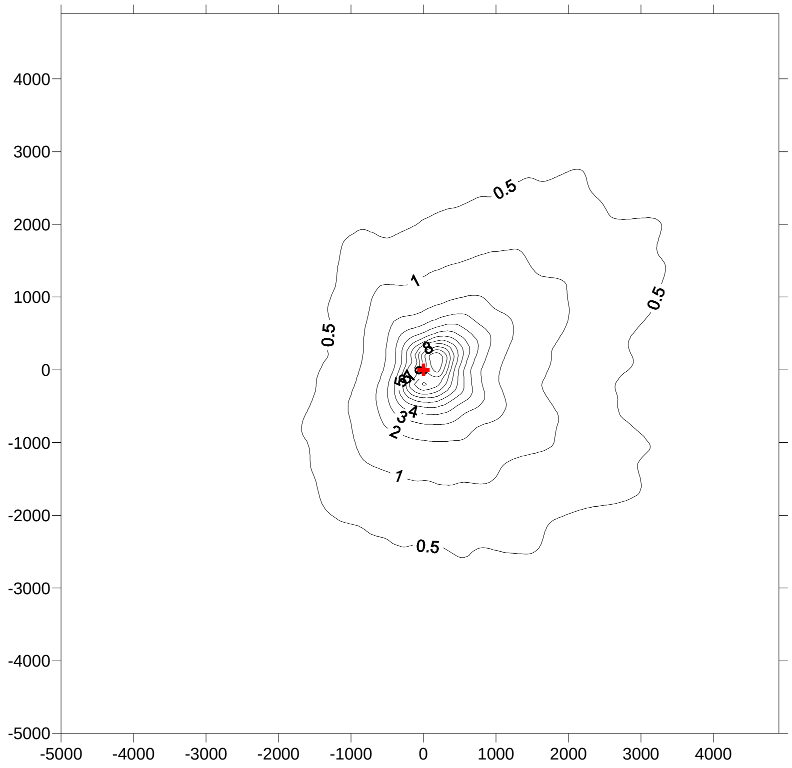

FIG. 2. Isoconcentration lines for meteorological conditions at the Sudbury site (in units of $\mu g/m^3$).

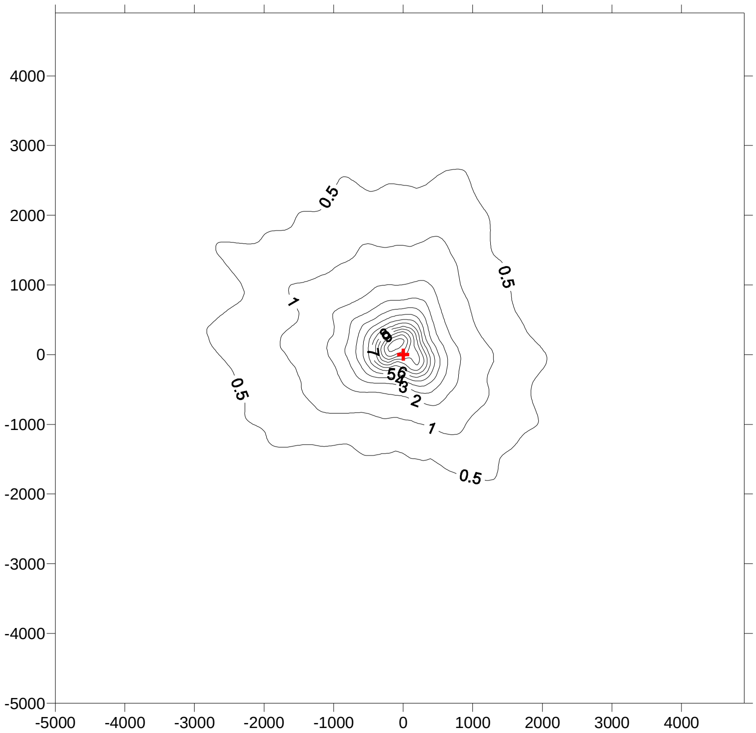

FIG. 3. Isoconcentration lines for meteorological conditions at the McLean Lake site (in units of µg/m³).

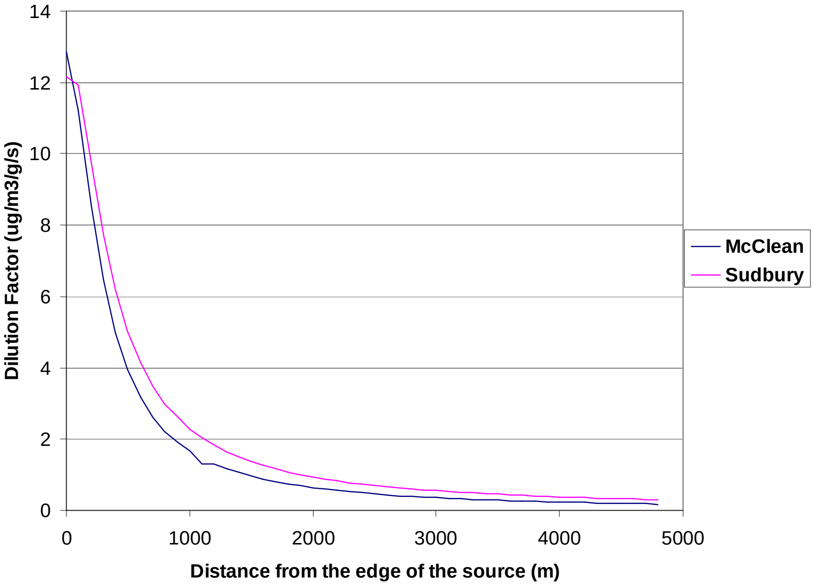

FIG. 4. Unit dilution factors for meteorological conditions at the Sudbury and McClean Lake sites along the line of maximum concentration.

II.2. Radon emissions

Although not formally within the scope of this study, some calculations were performed to estimate the radon concentration in air around the NORM residue deposit. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) provides information on sources of radon and the processes that affect the release of radon from soils [1]. A key parameter that controls radon transport in soils is the radon diffusion coefficient. A number of models for estimating the radon flux from the surface of porous media such as soil or waste rock are reported in the literature (see, for instance, Refs [1, 45, 46]). For dry soils, using the methods and values reported in Ref. [45], the estimated unit area radon flux per unit ^{226}Ra activity concentration (in becquerel per gram) is about 1 $\mathrm{Bq \cdot m^{-2} \cdot s^{-1}}$. The use if this value as the basis for the calculations was conservative — diffusion occurs through the unsaturated pore space of the soil and therefore the diffusion of radon in soil, where the soil is compacted or the pore space is filled with water (saturated), will be very much slower than in non-compacted or unsaturated soils.

Radon emissions from the representative residue deposit were calculated using a simple air dispersion model. The outdoor radon concentration was found to be about 10–20 Bq/m^3 in the immediate vicinity of the deposit. This range of radon concentrations is comparable with the range of natural variability of outdoor radon concentrations [1]. Indoor radon levels vary over a very large range up to well over 100 Bq/m^3, with a worldwide average of the order of 40 Bq/m^3 [47]. Given this intrinsic variability in natural radon levels, it would be very difficult to identify any clear increase in radon levels in the vicinity of a NORM residue deposit.

APPENDIX III. SENSITIVITY OF THE DOSE CALCULATIONS TO VARIATIONS IN DISTRIBUTION COEFFICIENTS

The dose calculations described in Section 3 were repeated using 'worst case' distribution coefficients for sandy soil (see Table 19). The results are shown in Tables 31 and 32. The intermediate results, where different from those of the main dose calculations given in Table 30 (that is, those using realistic distribution coefficients), are given in Table 33.

TABLE 31. RESULTS OF DOSE CALCULATIONS FOR RADIONUCLIDES IN THE ^{238}U AND ^{232}Th DECAY SERIES, EACH AT AN ACTIVITY CONCENTRATION OF 1 Bq/g, USING SAND DISTRIBUTION COEFFICIENTS

	Annual committed effective dose (mSv)	
	Child, 1–2 a	Adult
Inhalation of dust	0.000 34	0.000 65
Ingestion of dust	0.011	0.0018
External exposure	0.0050	0.0039
Ingestion of garden and agricultural products	0.000 28	0.000 10
Ingestion of groundwater	1.3	0.74
Consumption of fish	0.000 59	0.000 25
Total	1.3	0.75

TABLE 32. RESULTS OF DOSE CALCULATIONS FOR ONLY RADIONUCLIDES IN THE ^{238}U DECAY SERIES, EACH AT AN ACTIVITY CONCENTRATION OF 1 Bq/g, USING SAND DISTRIBUTION COEFFICIENTS

	Annual committed effective dose (mSv)	
	Child, 1–2 a	Adult
Inhalation of dust	0.000 13	0.000 25
Ingestion of dust	0.0070	0.0012
External exposure	0.0019	0.0015
Ingestion of garden and agricultural products	0.000 24	0.000 094
Ingestion of groundwater	1.1	0.67
Consumption of fish	0.000 55	0.000 24
Total	1.1	0.67

TABLE 33. INTERMEDIATE RESULTS OF DOSE CALCULATIONS, SAND DISTRIBUTION COEFFICIENTS

	U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-228	Th-232	Ra-228	Sum
Distribution coefficient (Table 19) (mL/g)	35	3200	500	270	150	3200	3200	500	
Ingestion of groundwater									
Retardation factor	395	36 000	5630	3040	1690	36 400	36 000	5630	
Leach rate (a^{-1})	1.58e-4	1.74e-6	1.11e-5	2.06e-5	3.70e-5	1.74e-6	1.74e-6	1.11e-5	
Activity concentration in seepage (Bq/m^3)	28 500	312	2000	3700	6660	312	312	2000	
Activity concentration in well water (Bq/m^3)	3930	43.1	276	511	919	43.1	43.1	276	
Committed effective dose (mSv/a)									
Child	0.10	0.002	0.026	0.018	0.81	0.0044	0.0019	0.16	1.29
Adult	0.13	0.003	0.027	0.012	0.39	0.0021	0.0035	0.067	0.74
Ingestion of garden and agricultural products									
Committed effective dose (mSv/a)									
Leafy vegetables									
Child	1.00e-5	1.76e-7	2.63e-6	1.83e-5	8.03e-5	4.41e-7	1.93e-7	1.56e-5	1.28e-4
Adult	8.14e-6	1.95e-7	1.66e-6	7.58e-6	2.37e-5	1.27e-7	2.13e-7	4.10e-6	4.58e-5
Non-leafy vegetables									
Child	5.91e-6	1.04e-7	1.56e-6	1.08e-5	4.75e-5	2.61e-7	1.14e-7	9.23e-6	7.55e-5
Adult	5.22e-6	1.25e-7	1.07e-6	4.87e-6	1.52e-5	8.16e-8	1.37e-7	2.64e-6	2.94e-5
Fruit									
Child	5.91e-6	1.04e-7	1.56e-6	1.08e-5	4.75e-5	2.61e-7	1.14e-7	9.26e-6	7.55e-5
Adult	5.22e-6	1.25e-7	1.07e-6	4.87e-6	1.52e-5	8.16e-8	1.37e-7	2.64e-6	2.94e-5

TABLE 33. INTERMEDIATE RESULTS OF DOSE CALCULATIONS, SAND DISTRIBUTION COEFFICIENTS (contd)

	U-nat	Th-230	Ra-226	Pb-210	Po-210	Th-228	Th-232	Ra-228	Sum
Fish consumption									
Activity concentration in surface water (Bq/m^3)	3.61	0.0396	0.254	0.470	0.845	0.0396	0.0396	0.245	
Committed effective dose (mSv/a)									
Child	1.11e-5	9.75e-7	7.30e-6	3.04e-4	2.23e-4	2.45e-6	1.07e-6	4.34e-5	5.94e-4
Adult	1.04e-5	1.25e-6	5.33e-6	1.46e-4	7.60e-5	8.14e-7	1.37e-6	1.31e-5	2.54e-4
External exposure									
Committed effective dose (mSv/a)									
Child	1.90e-6	5.90e-7	1.92e-3	1.08e-6	4.53e-8	2.02e-3	2.94e-7	1.06e-3	5.01e-3
Adult	1.46e-6	4.54e-7	1.47e-3	8.34e-7	3.48e-9	1.55e-3	2.26e-7	8.18e-4	3.58e-3

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