Conditional Exemption Limits for NORM Wastes

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ABSTRACT

As part of the UK Government's better regulation agenda, the Department of Energy and Climate Change (DECC), in conjunction with the Devolved Administrations, is reviewing the Radioactive Substances Act 1993 (RSA93) Exemption Order regime and proposes to replace all the existing exemption orders with a single, conditional exemption order. Under the present regime, disposal of some naturally occurring radioactive waste (NORM) is exempt from the RSA93 registration and authorisation requirements. DECC has asked HPA to investigate the amount of NORM waste, with a head of chain activity concentration of up to 5 Bq g^{-1}, that can be disposed of to landfill without exceeding specified dose criteria. (A head of chain activity of up to 5 Bq g^{-1} implies that each member of the chain has a maximum activity concentration of 5 Bq g^{-1}). Using the methodology previously developed to assess the radiological impact of disposal of high volume, very low level waste (HV-VLLW) to landfill, maximum activity capacities for a landfill site were calculated for each chain segment in the naturally occurring decay chains (^{232}Th, ^{238}U and ^{235}U) and for the entire chains. The capacity ranged from 3 10^{12} Bq for the entire ^{232}Th chain to 2 10^{15} Bq for ^{210}Po. From these landfill capacity values, annual consignor activity limits were derived, and these are presented as annual mass limits at several activity concentration levels.

HPA recommends that it would be appropriate to specify a generic upper limit on the annual mass of NORM waste (containing radionuclide activity concentrations that are all below 5 Bq g^{-1}) that can be disposed of to a landfill site, per consignor, of 10^4 t. This generic upper limit could be used in a UK exemption order for NORM wastes and would ensure that the specified dose criteria were not exceeded, as long as the annual mass capacity of the landfill site was 5 10^4 t or greater. If the waste contains radionuclides from more than one natural decay chain then the summation rule (sum of fractions of the radionuclide capacity for each radionuclide present) would be used to determine the quantity of that waste that can be disposed of, per consignor, per year.

The report also discusses other disposal options for this NORM waste, for example re-use as hardcore for a road or car park, and gives recommendations for the appropriate thickness of the covering layer.

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1 INTRODUCTION

1.1 Background

As part of the UK Government's better regulation agenda, the Department of Energy and Climate Change (DECC), in conjunction with the Devolved Administrations, is reviewing the Radioactive Substances Act 1993 (RSA93) (Great Britain, 1993) Exemption Order regime and proposes to replace all the existing UK exemption orders with a single, conditional UK exemption order.

Under the present regime, disposal of some naturally occurring radioactive waste (NORM) is exempt from the RSA93 registration and authorisation requirements. European guidance on clearance levels for waste containing naturally occurring radionuclides (European Commission, 2002) specifies that NORM wastes with a head of chain activity below 0.5 Bq g^{-1} can be disposed of unconditionally. DECC proposes to include this unconditional clearance provision in the revision of Schedule 1 to the RSA93, which defines material that is excluded from the Act[*]. DECC has asked HPA to investigate the amount of NORM waste, with a head of chain activity concentration of up to a specified constraint of 5 Bq g^{-1}, that can be disposed of to landfill without exceeding specified dose criteria. (A head of chain activity of up to 5 Bq g^{-1} implies that each member of the chain has a maximum activity concentration of 5 Bq g^{-1}).

This report describes the methodology used and presents the results of that investigation. this work considers wastes with a head of chain activity concentration between 0.5 and 5 Bq g^{-1}.

1.2 Assumptions and parameters used in this study

This study draws on the methodology developed for radiological assessments of disposal of high-volume, very low level waste (HV-VLLW) to landfill sites and takes many of the assumptions and parameters from that study (Chen et al, 2007). Where assumptions or parameters differ they are discussed below and key assumptions and parameters are reiterated for clarity. The aim is to use realistic but conservative assumptions for the assessment.

1.2.1 Radionuclides

The radionuclides considered in this study are those in the naturally occurring decay chains of ^{232}Th, ^{238}U and ^{235}U. The chains are divided into a series of segments, each

[*] UK exemption orders should not be confused with the EC concept of exemption and clearance. The UK exemption orders generally contain conditions. The exception is the Substances of Low Activity Exemption Order, which is unconditional and fulfils the same role as EC clearance levels.

one headed by a long-lived member of the decay chain. The doses arising from short-lived progeny within the segment are explicitly included in the calculated dose from their longer-lived parent, since secular equilibrium can be assumed to have been reached within that segment. Such segments are prefixed with a '+' so that, for example, $^{+210}$Pb includes ^{210}Pb and ^{210}Bi and $^{+235}$U refers to both ^{235}U and ^{231}Th. In the case of the entire chain, the parent is prefixed with a 'c' for example ^{c235}U refers to the entire ^{235}U chain, where the entire chain is assumed to be in secular equilibrium. The radionuclides considered to be in secular equilibrium with heads of chains and segments are summarised in Table 1 below. The full decay chains are illustrated in APPENDIX A.

The ^{c238}U and ^{c235}U chains are also considered together in their naturally occurring ratio, and this is referred to as ^{na}U. In ^{na}U the activity of ^{235}U is assumed to be 4.5% of that of ^{238}U and each of the two chains is assumed to be in secular equilibrium. This is different to Unat (extracted uranium ore) which only contains the uranium isotopes in their naturally occurring ratio, without all their progeny.

Table 1 Radionuclide chains and segments considered

Chain/segment	Radionuclides considered (secular equilibrium assumed with the parent)
^{c232}Th	^{232}Th, $^{+228}$Ra, $^{+228}$Th
$^{+228}$Ra	^{228}Ra, ^{228}Ac
$^{+228}$Th	^{228}Th, ^{224}Ra, ^{220}Rn, ^{216}Po, ^{212}Pb, ^{212}Bi, ^{212}Po (64.07%), ^{208}Tl (35.93%)
^{c238}U	$^{+238}$U, ^{234}U, ^{230}Th, $^{+226}$Ra, $^{+210}$Pb, ^{210}Po
$^{+238}$U	^{238}U, ^{234}Th, ^{234m}Pa (99.80%), ^{234}Pa (0.20%)
$^{+226}$Ra	^{226}Ra, ^{222}Rn, ^{218}Po, ^{214}Pb (99.98%), ^{218}At (0.02%), ^{214}Bi, ^{214}Po (99.98%), ^{210}Tl (0.02%)
$^{+210}$Pb	^{210}Pb, ^{210}Bi
^{c235}U	$^{+235}$U, ^{231}Pa, $^{+227}$Ac
$^{+235}$U	^{235}U, ^{231}Th
$^{+227}$Ac	^{227}Ac, ^{227}Th (98.62%), ^{223}Fr (1.38%), ^{233}Ra, ^{219}Rn, ^{215}Po, ^{211}Pb, ^{211}Bi, ^{207}Tl (99.72%), ^{211}Po (0.28%)
^{na}U	^{c238}U and ^{c235}U at a head-of-chain activity ratio of 1:0.045

A value following a radionuclide indicates that the chain or segment branches and gives the activity of that radionuclide as a percentage of the head-of-chain's or –segment's activity.

Where a head-of-chain or head-of-segment activity is stated, this is the activity of the parent radionuclide so that 1 Bq of $^{+235}$U means 1 Bq of ^{235}U plus 1 Bq ^{231}Th. In the case of ^{na}U, the activity is that of ^{238}U so that 1 Bq ^{na}U means 1 Bq of ^{238}U (and 1 Bq of each member of the ^{238}U chain) plus 0.045 Bq of ^{235}U (and 0.045 Bq of each member of the ^{235}U chain).

1.2.2 Dose criteria

The term "dose" is taken to mean the sum of the committed effective dose from intakes over a period and the external dose received over that period. The dose criteria used in this study are the same as those in the HV-VLLW assessment where they are discussed comprehensively (Chen et al, 2007). They are reproduced here for convenience.

Table 2 Dose criteria

Scenario	Dose criterion	Key groups
Operational phase	1 mSv y^{-1}	Landfill worker
Inadvertent intrusion (post-closure)	3 mSv y^{-1}	Member of the public (ie resident), construction worker
Inhalation of ^{222}Rn landfill gas (post-closure)	200 Bq m^{-3}	Member of the public (ie resident)
Migration with groundwater (post-closure)	0.02 mSv y^{-1}	Member of the public

1.2.3 Landfill parameters

The basic landfill characteristics are the same as those used in (Chen et al, 2007). The facility is assumed to have a total capacity of 2.2 10^6 tonnes over a lifespan of 15 years. The study considered three types of landfill site, with increasing levels of containment: inert, non-hazardous and hazardous waste landfill sites. It was found that the inert waste site gave rise to the highest doses and hence an inert landfill was selected for this study, see Section 2.2.

Disposal facilities take waste from several consignors, more than one of whom may wish to dispose of NORM waste. For the purposes of this study it has been assumed that there are three consignors of NORM waste but, since the relationship is linear, the final disposable activity and mass simply scale with the number of consignors.

2 METHODOLOGY

The HV-VLLW assessment (Chen et al, 2007) calculated doses to specific groups of people (adults only) from the disposal of radioactively contaminated waste, considering the scenarios and pathways listed in Table 3. The important exposed groups and scenarios were found to be the landfill worker for the operational phase, members of the public from migration with groundwater, members of the public from residence following inadvertent intrusion and members of the public from landfill gases; these were selected for this study.

The specific methodology used in this study is discussed below with details given of any parameters or assumptions that differ significantly from those in the HV-VLLW assessment. Since the HV-VLLW assessment did not consider the entire decay chains in secular equilibrium, it was necessary to derive the appropriate parameters for this study. Hence the dose coefficient for the head of chain or head of segment is the sum of the dose coefficients of the relevant radionuclides assumed to be in secular equilibrium, listed in Table 1. Adults were the only age group considered for the resident and migration scenarios since the exposures would occur over the lifetime of the individual and hence the annual dose averaged over a lifetime would be adequately represented by the annual dose to an adult (ICRP, 2000).

Table 3 Scenarios and exposure pathways considered in the HV-VLLW study

Scenario	Key groups*	Exposure pathways
Operational phase	Sorting worker	External irradiation; inhalation of dust; skin contamination
	Landfill worker	External irradiation; inhalation of dust; skin contamination; ingestion of dust
Leachate discharge	Member of the public	External irradiation; inhalation of re-suspended sediment; ingestion of water, freshwater fish and terrestrial foods
Migration with groundwater (post-closure)	Member of the public	External irradiation; inhalation of dust; ingestion of water, freshwater fish and terrestrial foods
Inadvertent intrusion (post-closure)	Construction worker	External irradiation; inhalation of dust; ingestion of dust
	Member of the public (ie resident)	External irradiation; inhalation of dust; ingestion of dust; ingestion of vegetables grown in the garden
Residential landfill gases (^{3}H, ^{14}C, ^{222}Rn)	Member of the public (ie resident)	Inhalation of radioactive gas

* Adults were the only age group considered in this study

2.1 Landfill workers

The methodology for calculating doses to the landfill workers was taken from the HV-VLLW assessment (Chen et al, 2007), with the modifications described below. The pathways considered were ingestion, inhalation, skin contamination and external dose (all for adults) and the total dose to landfill workers is the sum of the doses from these four pathways.

The HV-VLLW assessment (Chen et al, 2007) assumed that landfill operators spend most of their time outside standing on or next to the contaminated waste, and that their work raises dust that is also contaminated. An informal review of habits at a landfill site suggests that this is not realistic and that a landfill worker spends more than 90% of his time in the closed cab of an excavator with air-conditioning and a dust filtration system (McNulty, 2009). It was assumed that the ambient dust loading levels inside the cab are $1\ 10^{-5}$ g m^{-3} with an occupancy time of 1800 h y^{-1}, and the corresponding levels outside the cab are $1\ 10^{-3}$ g m^{-3} with an occupancy time of 200 h y^{-1}. This means that the time-weighted mean dust loading experienced by the worker (approximately $1\ 10^{-4}$ g m^{-3}) is about a factor of 10 lower than the dust loading outside the cab. It was therefore considered appropriate to apply a reduction factor of 10 to the landfill worker doses from the inhalation pathways calculated using the formula given by Chen et al (2007). It can further be argued that the air conditioning and dust filtration system reduce the dust loading inside the cab to very low levels and hence reduce the opportunity for contaminated dust to adhere to the hands or to be ingested whilst inside the cab. In addition the mass of the dust on the palms of the hand can be assumed to be reduced due to normal removal processes. It was therefore also considered appropriate to apply a reduction factor of 10 for both the inadvertent ingestion of material on the hands and skin contamination pathways calculated using the formula given by Chen et al (2007). In addition, the European Commission exemption calculations apply a shielding factor of 2 to landfill worker external doses to allow for the shielding effect of the cab walls and floor of a vehicle used to move and level waste (European Commission, 2002) and measurements taken in a landfill vehicle cab support the use of this factor (McNulty, 2009). Hence, it was considered appropriate to include this shielding and, since it was not considered by Chen et al (2007), a shielding factor of 2 has been applied to the external dose to landfill workers calculated using the formula given by Chen et al (2007). Further details of the calculations and a comparison with the results of the HV-VLLW study are given in APPENDIX B.

2.2 Migration with groundwater

The HV-VLLW assessment used a combination of tools to model the leaching of radionuclides from waste in a landfill and their flow through the landfill liner, along an aquifer and into the biosphere. Doses due to migration with groundwater were assessed for an inert, non-hazardous and hazardous landfill facility, as defined by EC and UK legislation (European Commission, 1999; Great Britain, 2007). It was found that migration from an inert landfill facility gave rise to the highest doses and so the present study only assesses migration from an inert landfill site.

Since the publication of the HV-VLLW study the tools used to model migration have been aggregated into a single tool known as the Landfill Modelling System (LMS) with improved model interfaces (Mobbs et al, to be published). The present study used the LMS to assess migration doses for the same dose pathways as the HV-VLLW study and using the same parameter values where possible. For radionuclides not assessed in the HV-VLLW study, parameter values were taken from standard sources such as ICRP Publication 72 (ICRP, 1996). As with the HV-VLLW assessment, this study assumed an initial inventory of 1 MBq of each radionuclide considered and then scaled the annual dose accordingly. The peak annual dose from each chain was obtained by summing the peak annual migration doses for each segment. Since the peak annual migration doses from the segments do not coincide in time, this approach overestimates the dose. Comparison of the results of the LMS with the HV-VLLW study is given in Appendix B.

2.3 Post-closure inadvertent intrusion

The HV-VLLW assessment calculated the doses to construction workers building housing on a landfill site 30 years after closure of the site and doses to residents living in this housing, using the contaminated land methodology of Oatway and Mobbs (2003) and taking the values for unburied, uniform contamination. The exposure pathways considered were external irradiation, inadvertent ingestion of dust, inhalation of dust and consumption of vegetables grown on the affected land. The HV-VLLW assessment further assumed that during development 1 m clean soil was mixed with the contamination.

Only the doses to residents were considered in the present study as the doses to construction workers were never limiting in the HV-VLLW study. Although inert landfill facilities are not required to be capped at closure, restoration will include re-landscaping of the site by covering with soil whose depth is unlikely to be less than 1 m. It should be noted that hazardous and non-hazardous sites are required to be capped by 1.5 m of soil. Although some of this covering may have eroded in the future when the site is assumed to be redeveloped, it is reasonable to assume that some covering remains. Indeed, if the covering is less than 1 m then the site excavation work will probably identify the presence of waste. The base case for this study used the contaminated land methodology results for buried, uniform contamination, where the contamination is covered by 15 cm clean soil. The sensitivity of the doses to cover thickness was investigated in a sensitivity analysis that compared these results with those from the contaminated land methodology for unburied, uniform contamination and with the results using the assumptions and parameters of Chen et al (2007), see section 3.3.

2.4 Inhalation of radon gas

The HV-VLLW assessment calculated the activity concentration of ^{222}Rn in a house built on the landfill, assuming that 30 years has elapsed since the ^{226}Ra was disposed of, and compared the concentration with the Radon Action Level (200 Bq m^{-3}). The same formula was used in this study but, in order to calculate the activity concentration of ^{226}Ra that would correspond to the Radon Action Level, the calculation was reversed:

$$C_{Ra-226} = \frac{C_{radon}\Lambda_{house}V_{house}e^{\lambda_{Ra-226}t}e^{h_2/H_2}}{a_H\lambda_{Rn-222}\rho_{waste}\tau H_1} \qquad (1)$$

where all parameters are described in Table 4.

Table 4 Parameters for calculating initial ^{226}Ra concentration

Parameter	Value	Units	Description
C_{Ra-226}		Bq t^{-1}	Initial activity concentration of ^{226}Ra at closure
C_{radon}	200	Bq m^{-3}	Activity concentration of radon in the house
Λ_{house}	8760	y^{-1}	Turnover of air in a house, assumed to be 1 per hour
V_{house}	125	m^3	Representative volume of house
λ_{Ra-226}	4.33 10^{-4}	y^{-1}	Decay constant of ^{226}Ra
t	30	y	Time between landfill closure and residential occupation
h_2	1.5	m	Thickness of cover
H_2	0.2	m	Effective relaxation length of cover
a_H	50	m^2	Horizontal area of the house
λ_{Rn-222}	66.2	y^{-1}	Decay constant of ^{222}Rn
ρ_{waste}	1.8	t m^{-3}	Bulk density of waste
τ	0.1	-	Emanation factor (fraction of radon atoms produced that escape from the solid phase of the waste in pore spaces)
H_1	0.2	m	Effective diffusion relaxation length for waste

The dose from radon gas effectively arises from the ^{226}Ra in the top 2 metres of the land on which the house is built. The sensitivity of the dose to the thickness of cover is discussed in section 3.2.

2.5 Consignor limits

For scenarios where the doses are dependent upon the activity concentration, the total activity capacity of the landfill can be calculated from the limiting activity concentration and the mass capacity of the landfill, ie

$$A_{crit} = MC_{crit} \qquad (2)$$

where A_{crit} is the total landfill activity capacity for that radionuclide, M (t) is the (mass) capacity of the landfill and C_{crit} (Bq t^{-1}) is the activity concentration that gives rise to doses equal to the relevant dose criterion. The annual consignor limit $A_{ann,con}$ is then

$$A_{ann,con} = \frac{A_{crit}}{TN} \qquad (3)$$

where T is the lifetime of the facility (assumed to be 15 years) and N is the number of consignors (assumed to be three). Clearly, the annual consignor limit can be scaled with the number of consignors. Assigning an annual consignor limit means that the consignor can choose to dispose of a smaller mass at a higher concentration or a larger mass at a proportionally lower concentration, whilst ensuring that the activity capacity of the landfill and the relevant dose criteria are not exceeded.

Equation (2) is not strictly true for exposure to radon or for the residence scenario since the dose from these scenarios depends on the near-surface concentration, and is independent of the concentration at greater depths. Hence, if waste at higher activity concentrations was disposed of at greater depths then the equation would underestimate the capacity of the landfill. However, this potential underestimate is not considered further.

The doses from the migration scenario depend on the total activity in the landfill and Equation (2) therefore gives the activity capacity of the landfill directly. Since the exposed groups for the three scenarios (landfill workers, migration and residence) are distinct, the landfill activity capacity was derived for each of the three scenarios and the most restrictive one selected. Application of Equation (2) for situations where the dose is dependent on total activity or on activity concentration is discussed below.

2.5.1 Doses dependent on total activity

If the capacity is limited by the doses resulting from migration in groundwater then the total activity in the site is the limiting factor, and the consignor can dispose of a greater mass of waste (obviously the maximum mass is limited by the physical size of the site) at a proportionately lower concentration should they so choose:

$$M_{con,C} = \frac{A_{ann,con}}{C}$$

where $M_{con,C}$ is the annual consignor limit at concentration C and is limited by the physical size of the site to a value below or equal to M/TN. In this study, the maximum annual consignor limit is assumed to be 5 10^4 t y^{-1} (based on a site capacity of 2.2 10^6 t, 15 year lifetime and 3 consignors).

2.5.2 Doses dependent on activity concentration

When the capacity is limited by the doses to the landfill worker, the doses resulting from inadvertent intrusion or from exposure to radon then the situation is more complicated since the doses are dependent on the activity concentration in the waste. Again, the site can only physically accept a maximum of M tonnes of waste whether it is at an activity concentration equal to C_{crit} or at a lower concentration, ie the amount of material at a concentration below C_{crit} that can be disposed of is limited by the physical size of the site. However, the site can accept waste at concentrations above C_{crit} as long as the average activity concentration in the mass of waste that is relevant to the scenario is C_{crit} or lower. This averaging mass depends on the scenario, as discussed below.

If the capacity is limited by the doses to landfill workers then the average activity concentration in the amount of waste that a worker could process per year ($7\ 10^4$ t waste per year, (Chen et al, 2007)) should be no more than C_{crit}. However, since this study assumes that each consignor can only dispose of $5\ 10^4$ t y^{-1}, then this lower value is used. Hence, for concentrations equal to or below C_{crit},

$$M_{con,C} = \frac{M}{TN}$$

and for waste at a concentration C above C_{crit},

$$M_{con,C} = \left(5\,10^4\ \text{t y}^{-1}\right)\frac{C_{crit}}{C}.$$

If the capacity is limited by the residential scenario then the average concentration in the plot of land considered in the scenario should be no more than C_{crit}. The averaging area used by Oatway and Mobbs, (2003) is about 100 m^2 and the depth is 1 m, giving an averaging volume of 100 m^3 soil, which is equivalent to approximately 150 t (dry weight). Hence, for concentrations equal to or below C_{crit},

$$M_{con,C} = \frac{M}{TN}.$$

Doses at the maximum activity concentration considered in this study, 5 Bq g^{-1}, are discussed in Section 3.3.

If the capacity is limited by the radon scenario then the average activity concentration in the soil below the house should be C_{crit}. The house area is 50 m^2 and the relevant depth is 2 m, giving an averaging volume of 100 m^3 soil, which is equivalent to approximately 150 t. Hence, for concentrations equal to or below C_{crit},

$$M_{con,C} = \frac{M}{TN}.$$

The sensitivity of the concentration of radon in air to the depth of soil covering the waste is discussed in Section 3.2.

Since both the residential and radon scenarios are dependent on the concentration of the radionuclides in the waste in the top one or two metres of the site, another option would be to dispose of wastes at concentrations above C_{crit} at greater depths. This would ensure that the dose criteria were still met. However, this has not been considered further in this study as it could be addressed in specified conditions for a UK exemption order.

3 RESULTS AND DISCUSSION

3.1 Capacities and concentration levels

Table 5 shows the calculated head-of-chain/segment activity capacities of a landfill site and the activity concentration level, C_{crit}, that meets the dose criteria for the landfill worker scenario (since C_{crit}, was found to be determined by the doses to workers, for all radionuclides). All values are shown to 1 significant figure. The minimum value for C_{crit} is 1 Bq g^{-1}, confirming that landfill disposal of NORM wastes containing 0.5 Bq g^{-1} (the EC unconditional clearance level) will also meet the dose criteria.

Four chain segments have activity concentration levels (C_{crit}) below 5 Bq g^{-1}, and the activity concentration level (C_{crit}) for each of the three full chains is at or below 2 Bq g^{-1}. The activity concentration level (C_{crit}) for $^{+226}$Ra calculated from the landfill worker scenario is 2 10^6 Bq t^{-1} (2 Bq g^{-1}), from the radon dose criterion in Equation (1) is 3 10^9 Bq t^{-1} (ie 3000 Bq g^{-1}), and from the residence scenario is 2.5 10^7 Bq t^{-1} (25 Bq g^{-1}). The radon dose contribution is therefore not limiting for this chain segment.

In three cases the migration doses are the ones that limit the activity capacity, and in all other cases it is the doses to landfill workers that are limiting.

Table 5 Head of chain/segment activity capacities and concentration levels

Radionuclide	From landfill worker doses (Bq)	From residential intrusion doses (Bq)	From migration doses (Bq)	Activity concentration level C_{crit} calculated for doses to landfill workers (Bq g^{-1})
^{c232}Th	**$3\ 10^{12}$**	$3\ 10^{13}$	$1\ 10^{14}$	1
^{232}Th	$2\ 10^{14}$	$2\ 10^{19}$	**$1\ 10^{14}$**	100
^{+228}Ra	**$7\ 10^{12}$**	$4\ 10^{15}$	large	3
^{+228}Th	**$4\ 10^{12}$**	$2\ 10^{18}$	large	2
^{c238}U	**$4\ 10^{12}$**	$5\ 10^{13}$	$9\ 10^{12}$	2
^{+238}U	**$2\ 10^{14}$**	$5\ 10^{15}$	$3\ 10^{15}$	90
^{234}U	$1\ 10^{15}$	$3\ 10^{19}$	**$2\ 10^{14}$**	500
^{230}Th	$2\ 10^{14}$	$6\ 10^{18}$	**$1\ 10^{13}$**	100
^{+226}Ra	**$4\ 10^{12}$**	$5\ 10^{13}$	$2\ 10^{16}$	2
^{+210}Pb	**$1\ 10^{15}$**	$8\ 10^{20}$	large	600
^{210}Po	**$2\ 10^{15}$**	$1\ 10^{43}$	large	1000
^{c235}U	**$5\ 10^{12}$**	$8\ 10^{14}$	$8\ 10^{13}$	2
^{+235}U	**$3\ 10^{13}$**	$6\ 10^{15}$	$9\ 10^{13}$	10
^{231}Pa	**$5\ 10^{13}$**	$1\ 10^{16}$	$2\ 10^{15}$	20
^{+227}Ac	**$6\ 10^{12}$**	$2\ 10^{15}$	large	3
^{na}U	**$4\ 10^{12}$**	$5\ 10^{13}$	$9\ 10^{12}$	2

Note: figures in **bold** are limiting capacities.

For capacities marked as 'large', activity decays completely before reaching biosphere so theoretical capacity is infinity.

3.2 Sensitivity of radon doses to cover thickness after intrusion

Radon (^{222}Rn) is the short-lived product (half life = 3.8 days) of the decay of ^{226}Ra and is parent to ^{218}Po. Both parent and progeny are solid, therefore in order to be present in air above a landfill site, radon has to migrate out of the waste and any covering before decaying. The concentration of radon in air is therefore very dependent on the concentration of ^{226}Ra in the near-surface soil and as a result is also very dependent upon the depth of clean soil covering the waste. From Table 5, the activity concentration level (C_{crit}) of ^{226}Ra in the waste that meets the landfill worker dose criteria is $2\ 10^6$ Bq t^{-1} ($1.8\ 10^6$ Bq t^{-1} to 2 significant figures). Given this initial activity concentration of ^{226}Ra in the waste at disposal, Figure 1 shows the indoor air concentration of radon from ^{226}Ra in the waste, calculated using Equation (1), against depth of clean soil above the waste.

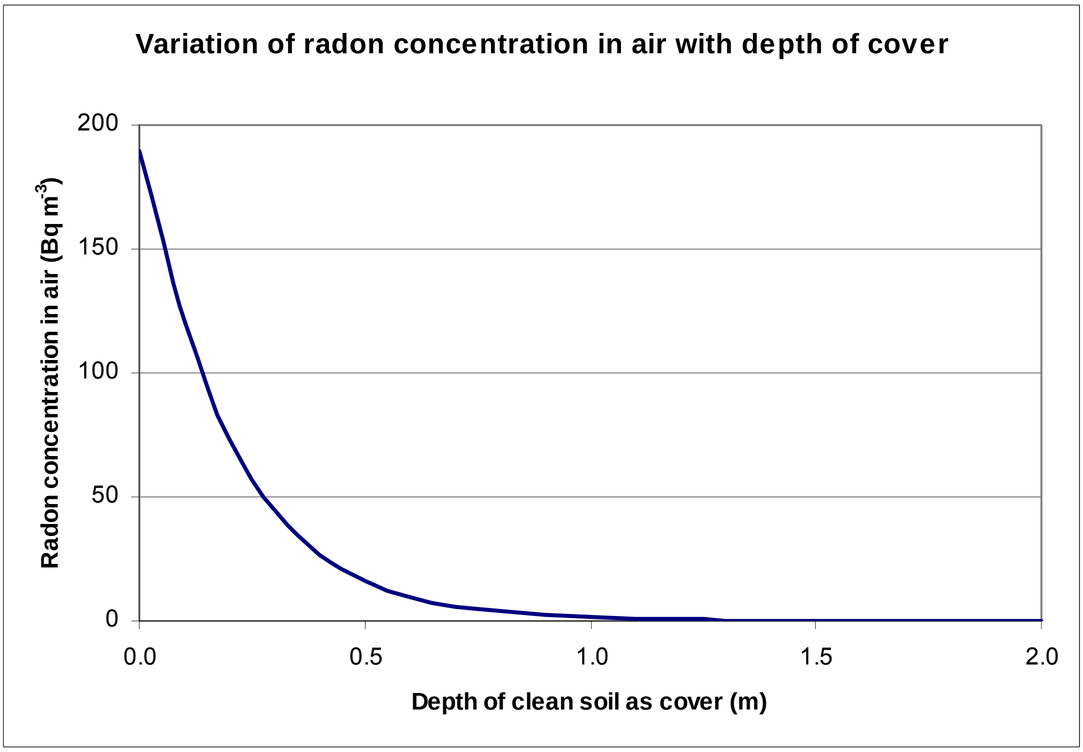

Figure 1 Graph showing radon concentration in air by depth of clean soil covering waste with a ^{226}Ra concentration of 1.8 Bq g^{-1}

It can be seen from the graph that even with no covering of clean soil over waste containing ^{226}Ra at an activity concentration of 1.8 Bq g^{-1}, the indoor concentration of radon in air is below the Radon Action Level of 200 Bq m^{-3}. Recent advice on the impact of radon on health (AGIR, 2009) suggests that a reduction in the Radon Action Level, to 100 Bq m^{-3}, would be cost effective; this concentration level would be met by a covering of about 20 cm of soil. With a covering of 30 cm clean soil, the dose due to radon from the waste will be less than the average dose due to radon from clean soil (1.2 mSv y^{-1}) (Watson et al, 2005) using the convention that 1 mSv corresponds to a concentration of about 20 Bq m^{-3}. With a 1.5 m covering of clean soil, the radon concentration due to waste containing ^{226}Ra at 1.8 Bq g^{-1} is 0.1 Bq m^{-3}. As discussed in section 2.3, landscaping or capping of landfill facilities will result in covering the site with soil to a depth unlikely to be less than 1 m. Although some of this covering may have eroded in the future when the site is assumed to be redeveloped, if the covering is less than 1 m then the site excavation work will probably identify the presence of waste and hence remedial measures could be implemented if necessary. Hence, the dose from radon emanating from 1.8 Bq g^{-1} of ^{226}Ra in the waste is unlikely to be greater than that from ^{226}Ra in normal clean soil.

3.3 Sensitivity of residential intrusion doses to cover thickness

The doses from residence on the site following inadvertent intrusion were calculated from the results for buried, uniform contamination given in the contaminated land methodology (Oatway and Mobbs, 2003). These results assume that the contamination is buried under a layer of clean soil 15 cm thick. The sensitivity of the doses to the thickness of covering soil was investigated using an initial assumed activity concentration of 5 Bq g^{-1}. Doses to residents were calculated for a covering of 15 cm, no covering (using the surface, uniform distribution in the contaminated land methodology) and assuming mixing of waste with 1m clean soil (as assumed in the HV-VLLW assessment). In all cases, it was assumed that 30 years had elapsed between disposal of the waste and intrusion onto the site, during which time the radioactive decay had occurred.

Table 6 Doses to residents 30 years after disposal of waste with a head of chain/segment activity concentration of 5 Bq g^{-1}, with different depths of clean soil cover

Radionuclide	15 cm clean cover, using W36, buried, uniformly distributed contamination (Sv y^{-1})	No clean cover, using W36, surface, uniformly distributed contamination (Sv y^{-1})	Mixed with 1 m clean soil, using HV-VLLW methodology (Sv y^{-1})
^{c232}Th	1 10^{-3}	4 10^{-3}	2 10^{-3}
^{232}Th	1 10^{-9}	5 10^{-5}	3 10^{-5}
$^{+228}$Ra	7 10^{-6}	4 10^{-5}	3 10^{-5}
$^{+228}$Th	2 10^{-8}	4 10^{-8}	3 10^{-8}
^{c238}U	7 10^{-4}	3 10^{-3}	2 10^{-3}
$^{+238}$U	7 10^{-6}	4 10^{-5}	2 10^{-5}
^{234}U	1 10^{-9}	1 10^{-5}	9 10^{-6}
^{230}Th	5 10^{-9}	4 10^{-5}	3 10^{-5}
$^{+226}$Ra	6 10^{-4}	2 10^{-3}	1 10^{-3}
$^{+210}$Pb	4 10^{-11}	4 10^{-4}	3 10^{-4}
^{210}Po	0	0	0
^{c235}U	4 10^{-5}	6 10^{-3}	4 10^{-3}
$^{+235}$U	5 10^{-6}	1 10^{-4}	7 10^{-5}
^{231}Pa	3 10^{-6}	4 10^{-3}	3 10^{-3}
$^{+227}$Ac	1 10^{-5}	4 10^{-4}	3 10^{-4}
^{na}U	7 10^{-4}	4 10^{-3}	2 10^{-3}

It can be seen that for 15 cm of clean soil covering the waste, all doses from waste containing 5 Bq g^{-1} are below the specified dose criterion of 3 mSv y^{-1}. If it is assumed that there is no clean soil covering the waste then the doses from all three chains would exceed the dose criterion (by up to a factor of 2) as does the dose from ^{231}Pa. Assuming that the waste is mixed with 1m of clean soil, as in the HV-VLLW methodology, only the dose from ^{c235}U at 5 Bq g^{-1} exceeds the dose criterion (by less than 50%).

However, using the derived activity concentration levels C_{crit} given in Table 5, with a maximum value of 5 Bq g^{-1}, the doses would all meet the dose criteria, even with no clean soil covering.

3.4 Annual consignor limits

Suggested values for head-of-chain/segment activity capacities are given in Table 7. The table also gives the annual consignor activity capacity, given the assumptions in Section 1.2, and the mass capacity at activity concentrations of 1 Bq g^{-1}, 2 Bq g^{-1} and 5 Bq g^{-1}, respectively. Some of these activity concentrations are above the values given in Table 5. However, as explained above, the dose criteria will not be exceeded as long as the annual consignor limits are not exceeded since these have been set to ensure that the relevant average concentration will be below the level given in Table 5. It should be remembered that 5 Bq g^{-1} is the highest activity concentration considered in this study, and hence disposal of material at higher activity concentrations would need to be considered separately, with a specific dose assessment.

Table 7 Annual consignor mass capacity limits (assuming three consignors of NORM waste over 15 years and head of chain activity concentration of 5 Bq g^{-1} or less)

Radionuclide	Landfill total activity capacity (Bq)	Consignor annual activity limit (Bq y^{-1})	Consignor mass capacity at 5 Bq g^{-1} (t y^{-1})	Consignor mass capacity at 2 Bq g^{-1} (t y^{-1})	Consignor mass capacity at 1 Bq g^{-1} (t y^{-1})
^{c232}Th*	3 10^{12}	6 10^{10}	1 10^{4}	3 10^{4}	5 10^{4} †
^{232}Th	1 10^{14}	2 10^{12}	5 10^{4} †	5 10^{4} †	5 10^{4} †
$^{+228}$Ra	7 10^{12}	2 10^{11}	3 10^{4}	5 10^{4} †	5 10^{4} †
$^{+228}$Th	4 10^{12}	1 10^{11}	2 10^{4}	5 10^{4}	5 10^{4} †
^{c238}U*	4 10^{12}	9 10^{10}	2 10^{4}	4 10^{4}	5 10^{4} †
$^{+238}$U	2 10^{14}	4 10^{12}	5 10^{4} †	5 10^{4} †	5 10^{4} †
^{234}U	2 10^{14}	4 10^{12}	5 10^{4} †	5 10^{4} †	5 10^{4} †
^{230}Th	1 10^{13}	2 10^{11}	4 10^{4}	5 10^{4} †	5 10^{4} †
$^{+226}$Ra	4 10^{12}	9 10^{10}	2 10^{4}	4 10^{4}	5 10^{4} †
$^{+210}$Pb	1 10^{15}	3 10^{13}	5 10^{4} †	5 10^{4} †	5 10^{4} †
^{210}Po	2 10^{15}	5 10^{13}	5 10^{4} †	5 10^{4} †	5 10^{4} †
^{c235}U*	5 10^{12}	1 10^{11}	2 10^{4}	5 10^{4}	5 10^{4} †
$^{+235}$U	3 10^{13}	6 10^{11}	5 10^{4} †	5 10^{4} †	5 10^{4} †
^{231}Pa	5 10^{13}	1 10^{12}	5 10^{4} †	5 10^{4} †	5 10^{4} †
$^{+227}$Ac	6 10^{12}	1 10^{11}	3 10^{4}	5 10^{4} †	5 10^{4} †
^{na}U	4 10^{12}	8 10^{10}	2 10^{4}	4 10^{4}	5 10^{4} †

* Head-of-chain capacity applies when chain is in secular equilibrium. For chains that are not in secular equilibrium the summation rule should be applied as described in Section 3.5 below.

† Annual consignor limit is limited to 5 10^{4} t y^{-1} by the physical size of the site, based on the assumptions in this study

Some of the calculated annual consignor limits (derived from meeting the dose criteria) were greater than the annual consignor limit of 5 10^{4} t y^{-1} assumed in this study to be the maximum amount that can physically be disposed of at the site in one year by each of 3 consignors. In Table 7 these are limited to 5 10^{4} t y^{-1} and are marked with †; this highlights the fact that the assumed characteristics of the site are such that the annual consignor capacity is usually limited by the physical size of the site. In fact, this is the case for all single chains or segments with a head of chain/segment activity concentration of 1 Bq g^{-1}. This confirms that unconditional disposal to landfill of wastes with activity concentrations of 0.5 Bq g^{-1} would also meet the criteria.

Since the activity-dose relationships are linear, the consignor capacities can be scaled for a different number of consignors or a different site lifetime. Similarly, the results of this study can be scaled to find an activity concentration that gives a specific dose or vice versa. Consignor limits at activity concentrations below 5 Bq g^{-1} that are not specified in Table 7 could also be obtained by appropriate scaling.

3.5 Mixtures of radionuclides

NORM wastes may not contain decay chains in secular equilibrium, and hence the activity of one segment of the radionuclide decay chain in the waste may be enhanced or reduced in relation to others. Similarly, NORM wastes may contain radionuclides from more than one of the natural decay chains. To determine whether disposal of M tonnes per year of NORM wastes containing mixtures of chains or chains out of secular equilibrium would meet the dose criteria, the summation rule (European Commission, 1996) can be applied:

$$\sum_{i=1}^{n} \frac{A_i}{A_{L,i}} \leq 1.0 \qquad (4)$$

where A_i is the annual activity of radionuclide i in the consigned waste and $A_{L,i}$ is the consignor annual activity limit of radionuclide i in Table 7. (Note that only radionuclides listed in Table 7 need be included in this calculation). A_i is derived from:

$A_i = C_i{*}M$

where C_i is the concentration of radionuclide i in the waste.

The amount of waste with a particular radionuclide composition that can be disposed of and still meet the dose criteria is then derived from:

Annual consignor mass limit (t) = $M/(\text{sum}(A_i/A_{L,i}))$ (5)

In other words, the quantity of waste with that composition that can be disposed of is inversely proportional to the sum of the fractions.

For example, the annual consignor limit for ^{+226}Ra is 9 10^{10} Bq y^{-1} and that for ^{230}Th is 2 10^{11} Bq y^{-1}. If a waste stream contains only these two segments at disposal, at activity concentrations of 1 Bq g^{-1} and 4 Bq g^{-1}, respectively, and it is proposed to dispose of 3 10^{4} t per year then this corresponds to annual activity levels of 3.0 10^{10} Bq y^{-1} and 1.2 10^{11} Bq y^{-1}, respectively. Hence

$$\sum_{i=1}^{n} \frac{A_i}{A_{L,i}} = \frac{A_{Ra+226}}{A_{L,Ra+226}} + \frac{A_{Th-230}}{A_{L,Th-230}}$$
$$= \frac{3.0 \times 10^{10}\,\text{Bq y}^{-1}}{9 \times 10^{10}\,\text{Bq y}^{-1}} + \frac{1.2 \times 10^{11}\,\text{Bq y}^{-1}}{2 \times 10^{11}\,\text{Bq y}^{-1}}$$
$$\approx 0.93 < 1.0$$

and therefore the waste stream meets the dose criteria. However, if the activity concentration of $^{+226}$Ra was 2 Bq g^{-1}, raising the annual activity of $^{+226}$Ra to 6 10^{10} Bq, then the sum of the fractions would be just over unity and the dose criteria would not be met. Nevertheless, a slightly smaller quantity of this waste, 2.3 10^4 t per year, could be disposed of and meet the dose criteria.

If all three decay chains are present with activity concentrations of 5 Bq g^{-1} for each member of the chain (the maximum allowed activity concentration in this study), and the summation rule is applied, then the maximum quantity of this waste that can be disposed of to landfill is 5 10^3 t per year for each consignor. The corresponding value for all three decay chains with activity concentrations of 1 Bq g^{-1} for each member of the chain is 2.6 10^4 t per year, and for concentrations of 0.5 Bq g^{-1} it is 5 10^4 t per year, the assumed capacity of the site.

It should be noted that the summation rule will overestimate the fraction when the head of chain segment limits are derived from doses to different groups at different times (eg landfill workers and migration doses). This becomes particularly obvious when the summation rule is applied to a chain that is in secular equilibrium: in this case the annual activity level in the waste of each member of the chain will be the same throughout the chain, ie $A_1 = A_2 = A_3 = \dots$. For example, if the annual activity in the waste of all the members of the ^{238}U chain is 9 10^{10} Bq y^{-1} and Equation (4) is applied, the sum will exceed unity. This is also the case for members of the ^{235}U chain, if the waste contains 10^{11} Bq y^{-1} of each member of the chain. Therefore the head of chain limit should be applied to a chain in secular equilibrium and the summation rule should only be applied when the chain is not in secular equilibrium i.e. when not all segments are present in the waste stream or when the activities of some segments have been enhanced or reduced by the processes that produced them.

3.6 Generic consignor limit

Table 7 shows that, based on the assumptions in this study, the annual consignor mass capacity varies from 10^4 t to 5 10^4 t for each single chain or chain segment; the value depends on the head of chain/segment activity concentration and this was limited in this study to be below 5 Bq g^{-1}. Given that the range of the annual mass capacities is only a factor of five, it raises the possibility of setting a generic annual consignor mass limit. In addition, depending on the choice of the generic value, it could be applied to landfill sites with more than three consignors, the number assumed in this study. Three options are discussed below.

The first option would be to specify a generic annual consignor mass limit of 10^4 t for NORM wastes containing one or more radionuclides, each with an activity concentration below 5 Bq g^{-1}. If only one natural decay chain or segment is present then, as long as the activity concentration of each member of the chain is below 5 Bq g^{-1}, the generic consignor limit would apply. It would not matter whether the chain was in secular equilibrium or not as the dose criteria would always be met. If more than one natural decay chain is present then the summation rule (European Commission, 1996) would need to be applied to the consignor annual activity limits for the relevant chain segments, as described in Section 3.5. Thus, as long as the maximum activity concentration in each of the decay chains is below 5 Bq g^{-1}, the appropriate quantity of waste containing more than one natural decay chain could be determined using the sum of the fractions, as described in Section 3.5. The generic annual consignor limit of 10^4 t would then be applied as an upper limit on the quantity that could be disposed of to a landfill site. This upper limit has the advantage of making the assumption that there are three consignors of waste containing NORM to the same site much less important.

The second option would be to specify a generic consignor limit of 5 10^3 t per year and this could be applied to all NORM wastes containing individual radionuclides at activity concentrations below 5 Bq g^{-1}. There would be no need to apply the summation rule as the dose criteria would always be met. Again, the lower annual consignor limit would mean that the actual number of consignors to one site was less important.

There is a special case where a generic consignor limit of 10^4 t per year would still apply even if there is more than one decay chain present in the waste: if the maximum activity concentration in each of the decay chains is identified and these values are summed and the sum is found to be below 5 Bq g^{-1}. Hence the third option would be to specify an annual consignor limit of 10^4 t per year for waste containing naturally occurring radionuclide decay chains, where the sum of the maximum activity concentrations identified in each of the three chains is less than 5 Bq g^{-1}. Again, the restriction on the sum of the maximum concentrations in each of the three decay chains would mean that the actual number of consignors was less important.

Although the simple rules offered by the second and third options are intuitively attractive, they do not offer any real practical advantage since the waste consignor has to determine the activity concentration of the chain segments anyway in order to demonstrate that they are all below 5 Bq g^{-1}. Once this has been done, the summation rule is easy to apply. Hence, these options could be viewed as unnecessarily restrictive since the benefit in terms of ease of application or regulation would be negligible. The first option (using the summation rule but with an upper limit of 10^4 t per year for a consignor) could also be viewed as unnecessarily restrictive as disposal of 2 10^4 t per year of NORM wastes containing radionuclide activity concentrations of 1 Bq g^{-1} would still meet the dose criteria. However, since the consignor limits calculated using the summation rule without an upper limit are based on the assumption that there are three consignors at each landfill site, this upper limit means that the actual number of consignors is less important.

3.7 Sensitivity to size of site

This study has assumed that the overall mass capacity of the site is 2.2 10^6 t, with a lifetime of 15 years, leading to an annual capacity of 5 10^4 t. If the site is a similar size, but with a shorter lifetime then the annual capacity would be greater and hence the fraction of the waste received each year by the site that was NORM would be smaller and the annual doses to landfill workers would be lower. Conversely, if the site lifetime was greater than 15 years, then the annual capacity would be correspondingly smaller, the NORM fraction would be higher and the doses to landfill workers could be higher if they still spend all their time at the site. (Doses to future residents would also be higher due to the higher NORM fraction, but this is less important because inadvertent intrusion is not a limiting scenario).

If the site is much larger than that assumed then the overall mass capacity would be greater. If the site lifetime is also greater, then the total amount of NORM waste disposed of in the site could be greater even though the annual capacity was unchanged. This is only important for doses arising from the migration pathway (the dose to landfill workers would be unchanged), and this was only the limiting pathway for two chain members/segments. The site would have to be one hundred times the size assumed in this study before the limiting total activity capacity was reached for these two chain members/segments.

If the site is much smaller than that assumed in the study then the fraction of the waste sent to the site that is NORM waste would be higher. As explained above, the doses to landfill workers (and to future residents) could be proportionally higher.

Hence the results and discussion in this study are valid for landfill sites that have annual mass capacities of around 5 10^4 t or larger.

3.8 Other disposal options

This assessment specifically addresses disposal of NORM wastes to a landfill of at least the standard of an inert landfill as defined under EC and UK legislation (European Commission, 1999). Other disposal options include sorting, on-site disposal, re-use and incineration; since doses to sorting workers are estimated to be around an order of magnitude lower than those to landfill workers (Chen et al, 2007), NORM wastes at or below the levels set in Table 7 can be sorted without the sorting workers incurring doses above the dose criterion (1 mSv y^{-1}). This assessment can also be applied to on-site disposal, provided the waste isolation capability is, at the minimum, of the standard of an inert landfill.

Incineration and re-use of waste are not covered by this assessment. However the same arguments apply as above for sorting of waste streams for re-use. One possible re-use of NORM is as a hardcore layer for a car park or children's play area where the waste will be covered by a layer of concrete and/or tarmac. The contaminated land methodology (Oatway and Mobbs, 2003) considers this scenario specifically, and finds that covering the waste eliminates all exposure pathways except exposure to external irradiation and also provides shielding against external irradiation. (Note that inhalation of radon is not considered in the contaminated land methodology.) For doses from complete chains, assuming secular equilibrium and an activity concentration of 5 Bq g^{-1}, ^{c232}Th is the limiting chain in this scenario (and ^{208}Tl is the dominant radionuclide, emitting gamma radiation with an energy of 2.6 Mev). The contaminated land methodology estimates that the resulting dose rate for a play area (assuming a covering layer over the NORM material 2 cm thick) would be of the order of 500 μSv y^{-1}; and for a car park with a typical covering of 25 cm the dose rate would be of the order of 6 μSv y^{-1}. The corresponding dose rate for a play area with a covering of 25cm of concrete or equivalent over the NORM waste was estimated by scaling the result for the car park to the occupancy time for a play area, resulting in an estimated dose rate of 30 μSv y^{-1}. Since this is an outdoor use, the small amount of radon migrating from the waste will disperse in the open and consequent doses will be insignificant. The reader is referred to the contaminated land methodology (Oatway and Mobbs, 2003) for more detail on the calculation of doses and dose rates in these scenarios.

The attenuation coefficient (μ/ρ) for concrete for gamma energies around 2.5Mev is about 0.04 cm^2 g^{-1} (NIST, 2010). This can be used to calculate a pessimistic measure of the attenuation provided by a particular thickness of concrete above the waste, using the attenuation equation:

Dose rate when covered = dose rate without cover * $e^{-(\mu/\rho .\ t)}$

Where t is the thickness of cover (g cm^{-2}) = d (cm) *ρ (g cm^{-3})

Assuming a density of 2.7 g cm^{-3} for concrete, and a radionuclide activity concentration of 5 Bq g^{-1}, a total of about 40 cm of concrete would reduce the dose rate for a play area to below 10 μSv y^{-1}. Since the dose rate is linearly dependent on the activity concentration, if the activity concentration of each of the radionuclides in the waste was 1 Bq g^{-1} or less, then a covering of 25cm of concrete or equivalent would also reduce the dose rate for a play area to below 10 μSv y^{-1}.

A dose criterion for exposure of the public following reuse of NORM material was not specified for this study. If the exemption criterion used in the EC study on NORM exemption levels (European Commission, 2002) is used (300 μSv y^{-1}), then 25cm of concrete cover over the NORM waste would ensure that the resulting doses were well below the criterion for both a car park and a children's play area, for waste containing radionuclides at an activity concentration of 5 Bq g^{-1}. However, if a dose criterion of 10 μSv y^{-1} is chosen, then a cover of 40cm would be required for a children's play area. Hence NORM waste containing radionuclides at an activity concentration of up to 5 Bq g^{-1} could be re-used as a base for a carpark or recreational area as long as it was covered by an appropriate thickness of concrete or equivalent. In general, any hardcore used for a road or a car park would typically be covered by around 25cm of concrete, whereas this is not necessarily the case for a recreational area. Hence, it is recommended that NORM waste containing radionuclides at an activity concentration of 5 Bq g^{-1} could be re-used as a base for a road or car park, as long as it is covered by a layer of concrete or equivalent with a thickness of around 25cm.

4 RECOMMENDATIONS

The results of this study can be used to propose a simple approach for defining the quantity of waste containing naturally occurring radionuclides that can be disposed of at a landfill site by a consignor, and meet the criteria specified in Table 2. The approach is based on applying the summation rule to the consignor annual activity limits in Table 7 and also applying a generic annual consignor mass limit of 10^4 t. It would operate as follows:

- The maximum activity concentration of any member of the three naturally occurring decay chains in waste that can be disposed of in a landfill is 5 Bq g^{-1}.
- The maximum quantity of waste containing naturally occurring radionuclides with an activity concentration between 0.5 Bq g^{-1} and 5 Bq g^{-1} (any chain member) that can be disposed of by a single consignor to one landfill is 10^4 t per year.
- The landfill site should have an annual mass capacity of at least fifty thousand (5 10^4) t
- Activity concentrations should be measured for each of the chain segments listed in Table 7. (Easy to measure members of the chain segment can be used to determine the activity concentration of that segment).
- The maximum activity concentration of each of the three natural decay chains present in the waste is identified and these maximum values are summed. If this sum is less than 5 Bq g^{-1} then a single consignor can dispose of up to 10^4 t of this waste per year to a landfill site. If the sum is greater than 5 Bq g^{-1} then the summation rule should be applied to the activity capacities in Table 7 (column 3), as described in section 3.5, to see if the proposed combination of activity concentration and quantity (limited to 10^4 t per year) meets the dose criteria. If the result of the summation rule is less than or equal to unity then the dose criteria are met and a single consignor can dispose of 10^4 t per year to a landfill site. If the result of the summation rule is greater than unity then the quantity of that waste that can be disposed of to a landfill site is proportionally lower and can be obtained from the sum of the fractions, as described in section 3.5.

HPA considers that this approach could be used as the basis for an exemption order for the disposal of large quantities of waste containing naturally occurring radionuclides to landfill sites with annual mass capacities of 5 10^4 t or greater. This approach could also be applied to an on-site disposal facility with containment characteristics equivalent to a landfill site for inert waste.

Exempt NORM material containing radionuclide activity concentrations that are below 5 Bq g^{-1} could be re-used as hardcore for a road or car park as long as it is covered by a layer of concrete or equivalent approximately 25 cm thick.

5 SUMMARY

The total activity finally disposed of in a landfill can be controlled, so that the resulting doses are below the dose criteria considered in Table 2, by limiting each consignor to an annual activity capacity and limiting the maximum activity concentration of each radionuclide in the waste to 5 Bq g^{-1}. Assuming that the mass capacity of the landfill site is 2.2 10^6 t and its lifetime is 15 years, each of three consignors can be allocated a "disposable activity" (consignor annual activity limit) for each chain or segment of NORM as shown in Table 7.

Based on these results, HPA recommends a generic consignor annual disposal limit of 10^4 t for NORM waste with a maximum activity concentration of each radionuclide that is below 5 Bq g^{-1}. If only one chain or chain segment is present then the consignor can dispose of 10^4 t of this waste to a landfill site per year. The summation rule (Equations (4) and (5)) should be used to calculate the allowance for wastes containing more than one decay chain.

NORM material containing radionuclide activity concentrations that are below 5 Bq g^{-1} could be re-used as hardcore for a road or car park as long as it is covered with a layer of concrete or equivalent, approximately 25cm thick.

Disposal of NORM wastes containing radionuclide activity concentrations above 5 Bq g^{-1} would require a specific dose assessment.

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APPENDIX A

Naturally-occurring radionuclide decay chains

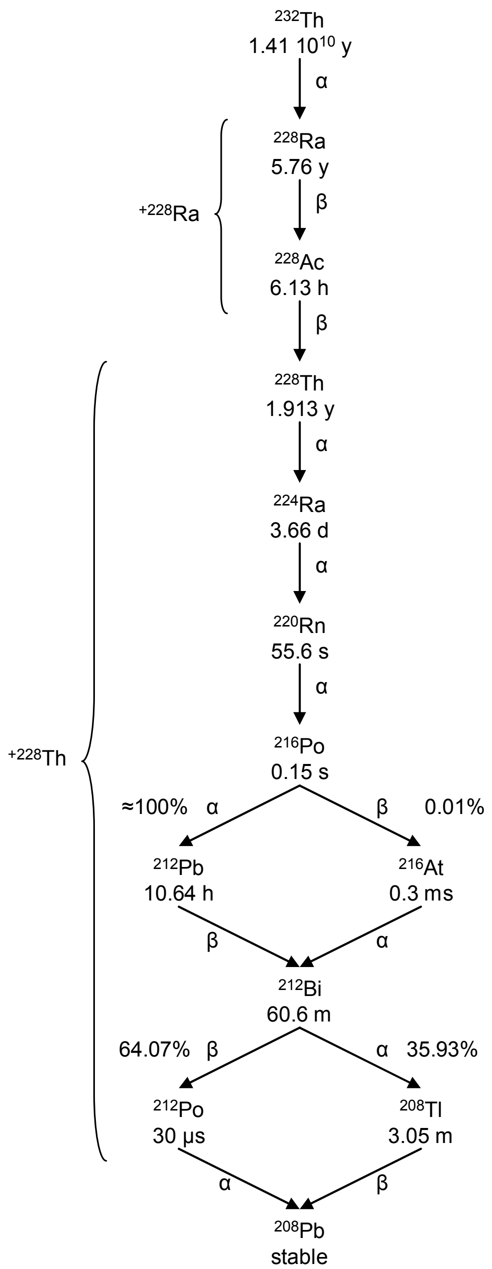

Figure 2 ^{232}Th decay chain

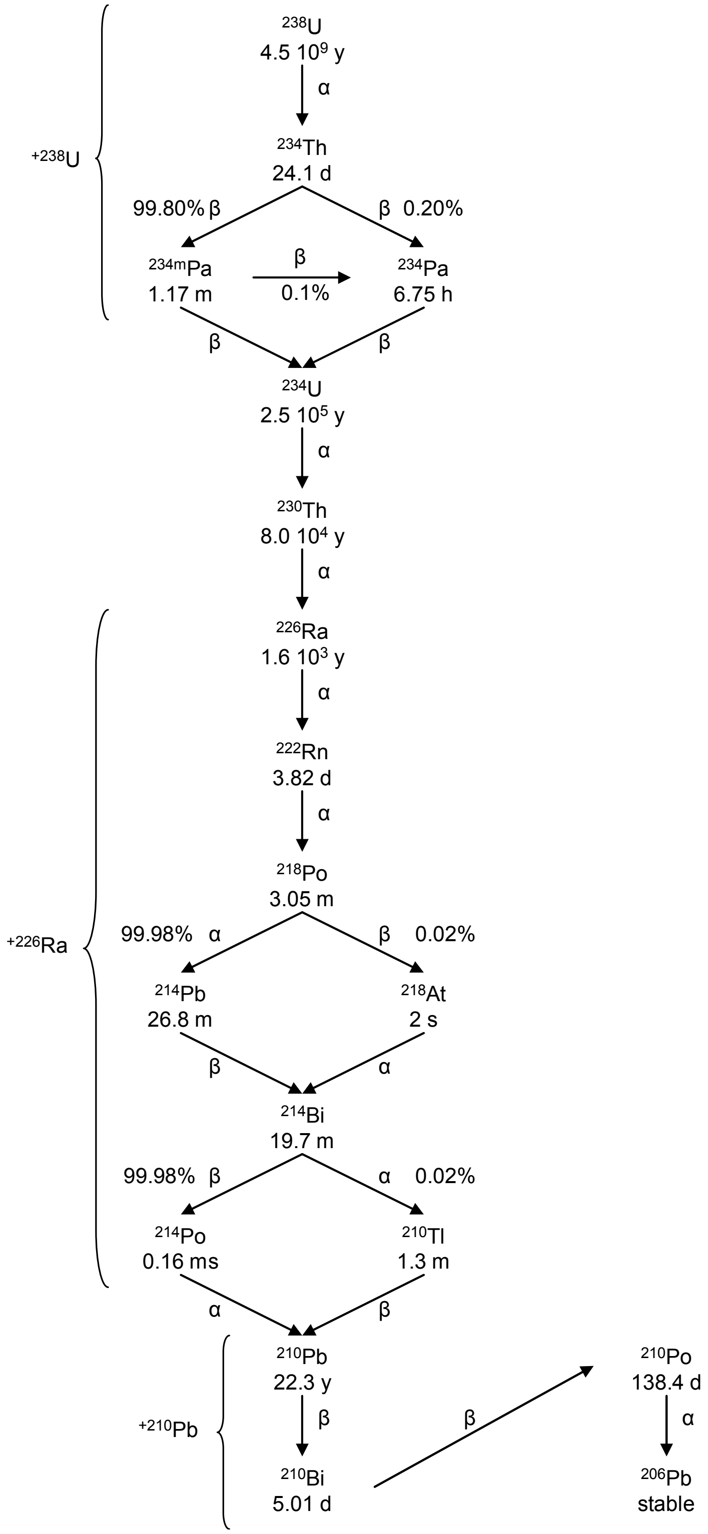

Figure 3 ^{238}U decay chain

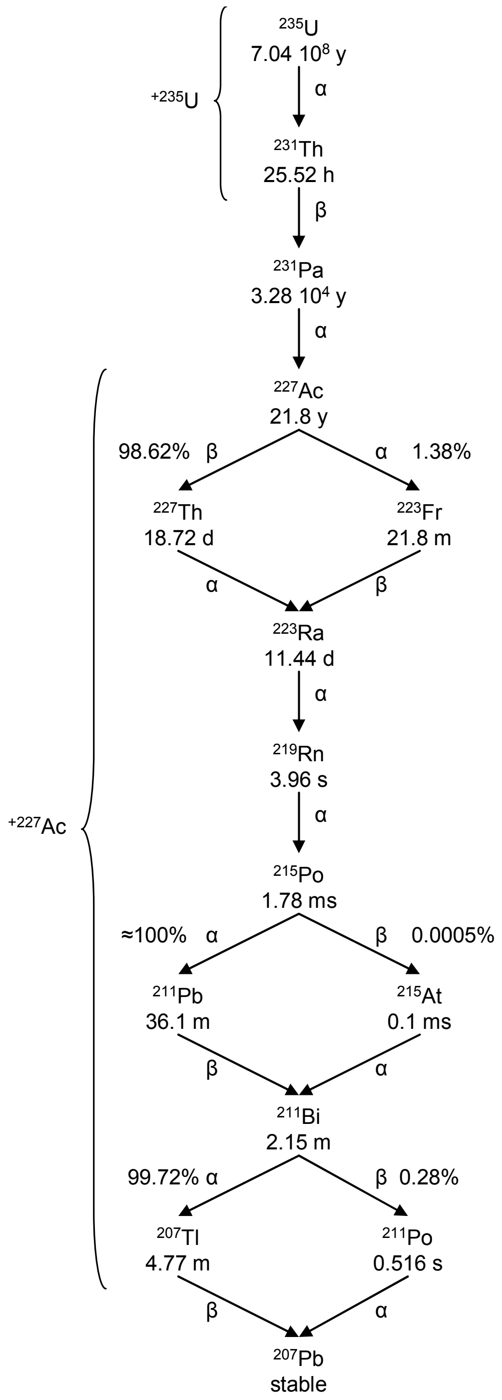

Figure 4 ^{235}U decay chain

APPENDIX B

Methodology and Data

B1 DOSES TO WORKERS

The methodology for calculating landfill worker doses was taken from the HV-VLLW assessment (Chen et al, 2007). The pathways considered were ingestion, inhalation, skin contamination and external dose (all for adults). Details of the ingestion rates, inhalation rates and other parameter values are given in Chen et al (2007). The total dose to a landfill worker is the sum of the doses from these four pathways.

B1.1 Ingestion

As in the HV-VLLW assessment, the committed effective dose coefficients for ingestion for each radionuclide were taken from ICRP68 (ICRP, 1995) and the coefficients summed to give a dose coefficient for each segment or chain. Where more than one dose coefficient was listed, the most restrictive value was used. The summed dose coefficients are given in Table B1. As discussed in Section 1.2.3, the doses calculated using this methodology were multiplied by 0.1 (ie a reduction factor of 10 was applied) to account for the worker spending 90% of their time in the cab.

B1.2 Inhalation

The inhalation dose coefficients for each radionuclide were taken from ICRPDOSE2, an ICRP-copyrighted software package that calculates doses to workers and members of the public based upon data in ICRP68 and ICRP72 (ICRP, 1995; ICRP, 1996). As with ingestion dose coefficients, the per-radionuclide inhalation dose coefficients were summed to provide per-segment and per-chain inhalation dose coefficients. It was assumed that an adult worker inhales contaminated particulate matter of 5 μm activity mean aerodynamic diameter (AMAD). The effective dose to age 70 years (ie integrated over 50 years) per unit intake was then used as the dose coefficient. The dose coefficients are given in Table B1. As discussed in Section 1.2.3, the doses calculated using this methodology were multiplied by 0.1 (ie a reduction factor of 10 was applied) to account for the worker spending 90% of their time in the cab.

B1.3 Skin contamination

The doses to skin arising from contaminated dust on the skin were calculated using the methodology in the HV-VLLW assessment, using beta dose coefficients taken from (Cross et al, 1992). These dose coefficients correspond to 0.07 mm water depth on the axis of a 1 Bq cm^{-2} plane isotropic source distributed uniformly over a 100 cm^2 circular area on an air-water boundary. The area was assumed to be the approximate surface area of the palm and fingers and the depth was assumed to be the most appropriate for simulating the distance of the basal layer of the epidermis from the contamination. The summed dose coefficients for segments and chains are given in Table B1. As discussed in Section 1.2.3, the doses calculated using this methodology were multiplied by 0.1 (ie a reduction factor of 10 was applied) to account for the worker spending 90% of their time in the cab.

B1.4 External

The external dose was estimated from both beta and gamma radiation and the same methodology used as in the HV-VLLW assessment, but with the addition of shielding by the cab. For beta radiation, the exposure geometry was assumed to be 1 m above a contaminated, semi-infinite slab and the dose coefficients were calculated from the mean beta energy per disintegration of the radionuclide using the algorithm on page 42 of RP65 (European Commission, 1993) and reproduced below:

$$J_\beta < 0.1\,\text{MeV} \qquad DF_{slab} = 0\ \text{Sv}\,\text{h}^{-1}\ \text{per Bq}\,\text{g}^{-1}$$

$$0.1\,\text{MeV} \le J_\beta < 0.4\,\text{MeV} \qquad \ln\left(DF_{slab}\right) = 6\ln\left(J_\beta\right) - 16.4$$

$$J_\beta \ge 0.4\,\text{MeV} \qquad \ln\left(DF_{slab}\right) = 2.86\ln\left(J_\beta\right) - 19.7$$

where J_β is the mean beta energy per disintegration and DF_{slab} is the beta dose rate 1 m above a semi-infinite slab. The mean beta energies were taken from ICRP38 (ICRP, 1983). The beta dose rate was calculated for each radionuclide, and the rates then summed.

For the gamma dose, the software application Microshield (Negin, 1986) was used to calculate the gamma dose rate for a 5 m thick infinite slab of concrete, density 2.35 g cm^{-3} with 1 Bq g^{-1} contamination (ie 2.35 Bq cm^{-3}). Photons below 0.015 MeV are excluded and photon energies are used as group references for radionuclides with fewer than 25 photons. The gamma dose rate is taken as the effective dose equivalent rate for rotational geometry with buildup, where the buildup material reference is the air gap. The beta and gamma external dose rates are given in Table B1. As discussed in Section 1.2.3, the gamma doses calculated using this methodology were multiplied by 0.5 (ie a shielding factor of 2 was applied) to account for the shielding afforded by the walls and floor of the cab. The beta dose rate inside the cab was assumed to be zero.

Table B1 Radionuclide dependent data

Radionuclide	Ingestion dose coefficient, Sv Bq^{-1}	Inhalation dose coefficient, Sv Bq^{-1}	Skin beta dose rate, Sv h^{-1} per Bq g^{-1}	External beta dose rate, Sv h^{-1} per Bq g^{-1}	External gamma dose rate, Sv h^{-1} per Bq g^{-1}
$^{c232}Th^{*}$	$1.03\ 10^{-6}$	$5.82\ 10^{-5}$	$5.25\ 10^{-6}$	$8.45\ 10^{-10}$	$8.01\ 10^{-7}$
^{232}Th	$2.20\ 10^{-7}$	$2.90\ 10^{-5}$	0	0	$5.68\ 10^{-11}$
^{+228}Ra	$6.70\ 10^{-7}$	$1.73\ 10^{-6}$	$1.62\ 10^{-6}$	$3.02\ 10^{-10}$	$3.08\ 10^{-7}$
^{+228}Th	$1.41\ 10^{-7}$	$2.75\ 10^{-5}$	$3.63\ 10^{-6}$	$5.43\ 10^{-10}$	$4.94\ 10^{-7}$
$^{c238}U^{*}$	$1.51\ 10^{-6}$	$4.61\ 10^{-5}$	$6.94\ 10^{-6}$	$2.69\ 10^{-9}$	$5.48\ 10^{-7}$
^{+238}U	$4.74\ 10^{-8}$	$5.71\ 10^{-6}$	$2.12\ 10^{-6}$	$1.57\ 10^{-9}$	$7.95\ 10^{-9}$
^{234}U	$4.90\ 10^{-8}$	$6.80\ 10^{-6}$	0	0	$3.94\ 10^{-11}$
^{230}Th	$2.10\ 10^{-7}$	$2.80\ 10^{-5}$	0	0	$1.32\ 10^{-10}$
^{+226}Ra	$2.80\ 10^{-7}$	$2.23\ 10^{-6}$	$3.28\ 10^{-6}$	$8.50\ 10^{-10}$	$5.40\ 10^{-7}$
^{+210}Pb	$6.81\ 10^{-7}$	$1.16\ 10^{-6}$	$1.55\ 10^{-6}$	$2.61\ 10^{-10}$	$2.46\ 10^{-10}$
^{210}Po	$2.40\ 10^{-7}$	$2.20\ 10^{-6}$	0	0	$2.71\ 10^{-12}$
$^{c235}U^{*}$	$1.97\ 10^{-6}$	$7.38\ 10^{-4}$	$8.15\ 10^{-7}$	$6.63\ 10^{-10}$	$2.47\ 10^{-7}$
^{+235}U	$4.63\ 10^{-8}$	$6.10\ 10^{-6}$	$8.15\ 10^{-7}$	$1.41\ 10^{-12}$	$7.41\ 10^{-8}$
^{231}Pa	$7.10\ 10^{-7}$	$8.90\ 10^{-5}$	0	0	$1.50\ 10^{-8}$
^{+227}Ac	$1.21\ 10^{-6}$	$6.43\ 10^{-4}$	0	$6.62\ 10^{-10}$	$1.58\ 10^{-7}$

* Assuming chain is in secular equilibrium

B2 DOSES TO MEMBERS OF THE PUBLIC

Doses to members of the public were calculated for three scenarios: migration with groundwater, residence on the landfill site following inadvertent intrusion and inhalation of radon following redevelopment of the site. The methodology was taken from the HV-VLLW assessment (Chen et al, 2007). The pathways considered were ingestion, inhalation, skin contamination and external dose and details of the parameter values are given in Chen et al (2007). In order to calculate the migration activity capacity for each chain, the peak migration doses for each segment were summed and a capacity calculated from the summed dose. Since the peak migration doses from the segments do not coincide in time, this method gives a conservative estimate.

B3 COMPARISON OF RESULTS

The HV-VLLW study included three segments in the naturally occurring decay chains: ^{232}Th, ^{+238}U and ^{+226}Ra. Landfill activity capacities were derived from the landfill worker, residential inadvertent intrusion and migration doses reported in Chen et al (2007) and compared with the values obtained in this study. Table B2 shows the results of the comparison of landfill activity capacities.

Table B2 Comparison of derived activity capacities with the HV-VLLW assessment

Radio-nuclide	Landfill worker derived from:		Residential intrusion derived from:		Migration derived from:		Limiting capacity derived from:	
	this study (Bq)	HV-VLLW study (Bq)	this study (Bq)	HV-VLLW study (Bq)	this study (Bq)	HV-VLLW study (Bq)	this study (Bq)	HV-VLLW study (Bq)
^{232}Th	2 10^{14}	2 10^{13}	1 10^{15}	1 10^{13} *	1 10^{14}	9 10^{14}	1 10^{14}	1 10^{13} *
$^{+238}$U	2 10^{14}	1 10^{14}	1 10^{15}	1 10^{15}	3 10^{15}	3 10^{15}	2 10^{14}	1 10^{14}
$^{+226}$Ra	4 10^{12}	2 10^{12}	2 10^{13}	1 10^{13}	2 10^{16}	4 10^{17}	4 10^{12}	2 10^{12}

* The capacity calculated for ^{232}Th from residential intrusion doses in the HV-VLLW study, on further investigation, proved to be the capacity calculated from the entire chain, ^{c232}Th. When compared to the capacity derived above for the entire chain, the capacities are the same (1 10^{13} Bq).

Since the methodology used in this study is taken from the HV-VLLW study, the two sets of values should agree well. Differences would be expected due to the additional shielding and reduction factors used for the doses to landfill workers and this is seen above. The derived capacities based on the doses to future residents (once the entire ^{232}Th chain is considered) agree well, as expected. The derived capacities from the migration doses are similar or smaller than those in Chen et al (2007). The difference is due to the improvement in the interfaces between the series of models that were used in the calculations. For example, the activity concentration in leachate from a landfill over time follows an exponential decay curve (Chen et al, 2007) and in the HV-VLLW assessment, model limitations resulted in this being approximated as a rectangular pulse. One of the improvements in the LMS is that this approximation is no longer needed and an exponentially decaying flux is used. Despite these differences in the migration results, the overall capacities calculated by these two systems agree well.