# Radiological Assessment of Disposal of Large Quantities of Very Low Level Waste in Landfill Sites

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## ABSTRACT

HPA-RPD and Atkins Ltd were asked by the Department for Environment, Food and Rural Affairs (Defra) and the Environment Agency (EA) to assess the potential radiological impact of disposal of large quantities of very low level solid radioactive waste (VLLW) from the nuclear industry to conventional landfill sites, relating to a proposed change in the definition of VLLW.

The study considered eleven radionuclides likely to be produced in typical waste streams, together with four example waste streams from nuclear sites. Separate assessments were carried out for an inert landfill, a non-hazardous landfill and a hazardous landfill. The assessments considered a number of exposure scenarios: the exposure of operatives at a receiving landfill site, including sorting; the direct disposal to a river of contaminated leachate from the leachate management system at the site; and the radiological impact to members of the public after closure of the site, both from the migration of leachate in groundwater from the site and from possible inadvertent intrusion into the site, including future residence on the site.

A set of 'base case' results were produced and a number of sensitivity runs were carried out to explore the likely range of results. Since this is a generic assessment, the assumptions for the base case were conservative so as not to underestimate the doses.

Results from the study show that for six of the radionuclides the doses from disposal of large quantities of waste ($10^6$ tonnes) at the new definition of VLLW meet the dose criteria that were specified for use in this project. However, for the other five radionuclides, the doses from some scenarios are near to or above the dose criteria and hence a more detailed site-specific assessment would be needed to determine the quantity that could be disposed of and to identify any necessary conditions for disposal.

The regulatory bodies would require Best Practicable Means (BPM) to be demonstrated for a dedicated facility for radioactive waste and would require some form of site-specific assessment. Such an assessment could be based on the results of this study.

This report from HPA Radiation Protection Division reflects understanding and evaluation of the current scientific evidence as presented and referenced in this document.

**The following amendments have been made to this report since its first publication (May 2007)**

**September 2007**

Page number 25  In the fifth line of the last paragraph 'only a factor of 2 below it' has been amended to 'a factor of 4 below it'.

Page number 25  In the seventh line of the last paragraph 'a factor of 20 below' has been amended to 'a factor of 40 below'.

Page number 25  In the eighth line of the last paragraph 'only 4 radionuclides $^{14}$C, $^{36}$Cl, $^{238}$U and $^{241}$Am' has been amended to 'only 5 radionuclides $^{14}$C, $^{36}$Cl, $^{238}$U, $^{239}$Pu and $^{241}$Am'.

Page number 27  In the first line of the first paragraph 'For the non-hazardous site the disposal tonnage for all radionuclides is below the guideline value, apart from $^{14}$C at 40 MBq te$^{-1}$ which is about the same as the guideline value' has been amended to 'For the non-hazardous site the disposal tonnage for all radionuclides is less restrictive than the guideline value'.

Page number 26 Table 11 has been extensively amended.

Page number 27 Table 13 has been extensively amended.

Page number 27 In the fourth line of the first paragraph 'The short-lived radionuclides $^{36}$Cl, $^{60}$Co and $^{90}$Sr' has been amended to 'The short-lived radionuclides with high sorption values $^{36}$Cl and $^{60}$Co'.

Page number 29  In the fifth line of the second paragraph 'is still close to 20 µSv. However, if the $^{14}$C concentration is limited to 4 MBq te$^{-1}$, the peak annual dose will be less than 2 µSv.' has been amended to 'is below 20 µSv. However, if the $^{14}$C concentration is limited to 4 MBq te$^{-1}$, the peak annual dose will be less than 1 µSv'.

Page number 33 Table 17 has been extensively amended.

Page number 61 Appendix B, section 3.3, subsection on drinking water well has been deleted as this methodology was not used in this report: drinking water doses were calculated for discharge to a river, using BIOS.

Page number 74 Table B6 The ingestion dose coefficients for $^{+228}$Th are corrected from 1.1 10$^{-6}$ (infant), 4.3 10$^{-7}$ (child), 1.4 10$^{-7}$ (adult) to 3.7 10$^{-7}$ (infant) 1.5 10$^{-7}$ (child) 7.2 10$^{-8}$ (adult).

# EXECUTIVE SUMMARY

HPA-RPD and Atkins Ltd were asked by the Department for Environment, Food and Rural Affairs (Defra) and the Environment Agency (EA) to assess the potential radiological impact of disposal of large quantities of very low level solid radioactive waste (VLLW) from the nuclear industry to conventional landfill sites, relating to a proposed change in the definition of VLLW. The aim of the study is to determine the quantity of this VLLW that could be disposed of per annum to a landfill site without exceeding the dose criteria that were specified for use in this project.

The proposed definition of VLLW is *"Waste with maximum concentrations of 4 MBq te$^{-1}$ of beta/gamma activity (or 40 MBq te$^{-1}$ in the case of wastes containing solely $^{14}$C or tritium), and no single items of more than 40 kilobecquerels (kBq) of beta/gamma activity (or 0.4 MBq in the case of items containing solely $^{14}$C or tritium), which can, in appropriately controlled quantities, be safely disposed of with domestic refuse".*

The dose criteria agreed with Defra and the EA for use in this project are given in the following table. However it should be noted that these criteria do not constitute formal advice from HPA-RPD or the regulators. The models used in this study are linear and hence the results can be scaled for other dose criteria, if required.

| Scenario | Exposed person | Numerical value for dose limit |
|---|---|---|
| Operational phase | Landfill worker, sorter | 1 mSv y$^{-1}$ |
| Discharge of leachate to river | Member of the public | 300 µSv y$^{-1}$ |
| Natural processes post closure | Member of the public | 20 µSv y$^{-1}$ |
| Inadvertent intrusion post closure | Construction worker, Member of the public | 3 mSv y$^{-1}$ |
| Inhalation of landfill gas post closure | Member of the public | 1 mSv y$^{-1}$ for $^{3}$H and $^{14}$C, 200 Bq m$^{-3}$ for $^{222}$Rn |

The term 'dose' denotes the sum of the committed effective dose and the external dose

The study considered the range of radionuclides likely to be produced in typical waste streams, together with four example waste streams from nuclear sites. Separate assessments were carried out for an inert landfill, a non-hazardous landfill and a hazardous landfill. The assessments took into account the likely role and activities of operatives at a receiving landfill site, including sorting. In addition, they considered the exposure pathways arising from the direct disposal to a river of contaminated leachate from the leachate management system at the site. The study also looked at the radiological impact to members of the public after closure of the site, both from the migration of leachate in groundwater from the site and from possible inadvertent intrusion into the site, including future residence on the site.

A set of 'base case' results were produced and a number of sensitivity runs were carried out to explore the likely range of results. Since this is a generic assessment, the assumptions for the base case were conservative so as not to underestimate the doses.

Results from the study show that in many cases the doses from disposal of large quantities of waste at the new definition of VLLW meet the specified dose criteria.

However, there are several radionuclides which, in some scenarios, give doses near to or well above the specified dose criterion. In these cases the next step would be to review the pessimistic assumptions used in the generic calculations to determine whether the degree of conservatism is appropriate. A more detailed site-specific assessment would then be needed to determine the quantity of waste that could be disposed of, bearing in mind the range of appropriate engineering measures to the site and actual working practices that could be specified.

It was found that for the radionuclides $^3$H, $^{36}$Cl, and $^{90}$Sr disposal of a mass of approximately $10^5$ te y$^{-1}$ (or a total of about $10^6$ tonnes) at the VLLW concentration would meet the dose criteria specified in this study. Disposal of a mass of approximately $10^5$ te y$^{-1}$ (or a total of about $10^6$ tonnes) of waste containing $^{238}$U, $^{239}$Pu or $^{241}$Am at 4 MBq te$^{-1}$ would also meet the dose criteria specified in this study. However, this generic assessment found that some of the dose criteria may not be met if $10^5$ te y$^{-1}$ (or a total of about $10^6$ tonnes) of VLLW waste containing the radionuclides $^{14}$C, $^{60}$Co, $^{137}$Cs, $^{226}$Ra and $^{232}$Th in significant fractions were disposed of at a particular site, depending on how appropriate the pessimistic assumptions were considered to be. (Concentration levels of both 4 MBq te$^{-1}$ and 40 MBq te$^{-1}$ were assessed for wastes containing $^{14}$C).

The study concludes that if Defra formally adopted the dose criteria used in this study then there would be no need to set a limit for the quantity of VLLW containing $^3$H, $^{36}$Cl, or $^{90}$Sr at the concentrations in the proposed definition that could be disposed of in a landfill site to below $10^6$ tonnes. Similarly, there is no need to limit the quantity of waste containing $^{238}$U, $^{239}$Pu or $^{241}$Am at 4 MBq te$^{-1}$ to below $10^6$ tonnes. However, for $^{14}$C, $^{60}$Co, $^{137}$Cs, $^{226}$Ra and $^{232}$Th, the results of the generic assessment and sensitivity analysis indicated that it may be necessary to limit the quantity of waste containing these radionuclides at 4 MBq te$^{-1}$ or 4 Bq g$^{-1}$ to below $10^6$ tonnes, depending on whether the pessimistic assumptions were judged to be appropriate. Hence, for these five radionuclides a more detailed site-specific assessment would be needed to determine the quantity that could be disposed of and to identify any necessary conditions for disposal.

The regulatory bodies would require Best Practicable Means (BPM) to be demonstrated for a dedicated facility for radioactive waste. Thus, even if the waste and site characteristics matched those of the generic assessment given in this report, the regulators would still require some form of site-specific assessment to show that the issues had been considered. However, such an assessment needs to be proportionate and could be based on the results of this study.

# CONTENTS

# 1    INTRODUCTION

## 1.1    Background

This report presents the results of a study carried out by the Health Protection Agency-Radiation Protection Division (HPA-RPD) and Atkins Limited on behalf of the Department for Environment, Food and Rural Affairs (Defra) and the Environment Agency (EA). The aim of the work was to assess the potential radiological impact of disposal of large quantities of Very Low Level solid radioactive Waste (VLLW) from the nuclear industry to conventional landfill sites that could arise as a result of a proposed change in definition of VLLW. The proposed new definition of VLLW, given in the document "A Public Consultation on Policy for the Long Term Management of Solid Low Level Waste in the United Kingdom" (Defra et al, 2006) is:

*"Waste with maximum concentrations of 4 MBq te$^{-1}$ of beta/gamma activity (or 40 MBq te$^{-1}$ in the case of wastes containing solely $^{14}$C or tritium), and no single items of more than 40 kilobecquerels (kBq) of beta/gamma activity (or 0.4 MBq in the case of items containing solely $^{14}$C or tritium), which can, in appropriately controlled quantities, be safely disposed of with domestic refuse".*

The existing definition of VLLW is given in the 1995 Cm2919 White Paper (United Kingdom - Parliament, 1995)  as:

*"each 0.1 m$^3$ of which contains less than 400 kilobecquerels (kBq) of beta/gamma activity, or single items containing less than 40 kBq of beta/gamma activity, which can be safely disposed of with ordinary refuse ('dustbin disposal')".*

In some authorisations for disposal of VLLW, the concentration limits for the radionuclides $^3$H and/or $^{14}$C are relaxed by a factor of ten. Thus the proposed definition effectively incorporates two changes: firstly it formalises the higher concentration limits for $^3$H and $^{14}$C, and secondly it effectively removes the volume limit. This means that, potentially, larger quantities of waste with activity concentrations meeting the new definition could be disposed of to landfill sites whereas previously it was reserved for small quantities. The radiological implications of the existing definition of VLLW were assessed in a previous study (Harvey et al, 1995) and more recently in a parallel study by SNIFFER (Reedha and Wilmot, 2005).

Therefore, the aim of this study was to determine the quantities of VLLW (using the new definition) from nuclear sites that could be disposed of per annum to unspecified sites operated as conventional landfills without exceeding specified dose criteria.   Since the models are linear, these results can be scaled for other dose criteria, as required.

The assessment considers a number of radionuclides that could be present in the waste; four example waste streams from nuclear sites; the way the waste might be dealt with at the disposal site, taking into account the likely role and activities of operatives, including sorting (see Appendix A for details); and the

scenarios by which members of the public could be exposed during operation and post closure of the site, (see Appendix B for details). Separate assessments are provided for three types of landfills, based on the types used for non-radioactive wastes: an inert landfill (which might receive VLLW such as contaminated rubble), a non-hazardous landfill and a hazardous landfill (which might receive VLLW such as incinerator ash). The choice of these three types of landfill is for the purpose of this assessment only and should not be assumed to imply any decision that large volumes of VLLW would be disposed of at existing landfill sites.

## 1.2 Dose criteria chosen for use in this study

Neither the regulators nor HPA-RPD have developed formal guidance on dose criteria for the disposal of VLLW and therefore it was necessary to define dose criteria to be used in this study. The dose criteria agreed with Defra and the EA for use in this work are shown in Table 1. However it should be noted that these criteria do not constitute formal advice from HPA-RPD or the regulators. The following section details the reasoning for selecting these dose criteria. The scenarios they apply to are described in more detail in Sections 4 and 5. The CERRIE report (CERRIE, 2004) concludes that the concept of effective dose is appropriate for prospective studies such as this. The term 'dose' in this report is used to denote the sum of the committed effective dose and the external dose.

For the workers on the landfill sites the annual dose limit for members of the public was adopted ie 1 mSv $y^{-1}$. This was based on (Environment Agency et al, 2002) which states that:

*'the view of the Environment Agencies, NRPB and the FSA is that where such workers do not receive direct tangible benefits from the organisation discharging radioactive waste to the environment (eg employment by that organisation) then these workers should be treated as if they are members of the public for the purposes of authorising the discharges. The dose limit and dose constraint for members of the public would then apply to these groups. The HSE has supported this view for the purposes of authorising discharges'.*

The dose limit for members of the public was chosen for the landfill workers, rather than the dose constraint, as it is enshrined in UK legislation in the Ionising Radiations Regulations (United Kingdom - Parliament, 2000).

The dose criterion used in this study for exposure of members of the public from discharges during the operational phase (ie for the discharge of the collected leachate to river) was an effective dose of 300 $\mu$Sv $y^{-1}$, or a risk of $10^{-5} y^{-1}$. This is consistent with the criteria for disposal of low and intermediate waste (Environment Agency et al, 1996). The use of the dose constraint of 300 $\mu$Sv $y^{-1}$ allows for the potential co-location of other potential sources of exposure to the public which might include discharges from nuclear sites. These would be additive and comparable to the 1 mSv annual dose limit for members of the public referred to above.

For the post closure phase, the approach recommended by the International Commission of Radiological Protection (ICRP) (ICRP, 1998) of using separate criteria for the two categories of post-closure exposure, natural processes and inadvertent human intrusion, was adopted.

The criterion adopted for natural processes post closure (migration with groundwater) was a risk of $10^{-6}$ $y^{-1}$, which corresponds to an effective dose of 20 $\mu$Sv $y^{-1}$ if the process is certain to occur and is consistent with the risk target for disposal of low and intermediate waste (Environment Agency et al, 1996).

Inadvertent intrusion into the site in the future is not certain to occur and therefore has a probability of occurrence. It was decided to follow the approach recommended in ICRP 81 (ICRP, 1998) namely to specify a dose criterion for the scenario were it to occur, based on intervention considerations, rather than to use a risk criterion and a probability of occurrence. Accordingly, a dose criterion of 3 mSv $y^{-1}$ was adopted to be consistent with the dose criterion for the definition of radioactively contaminated land (HPA, 2006). The reasoning was that it would seem appropriate to make sure that, if inadvertent intrusion occurred into an old disposal site and exposures were being received due to the remaining radioactivity on the site, then the land would not be registered under the contaminated land regime. (Planned redevelopment of an old disposal site in the knowledge that it was a disposal site as opposed to unintentional intrusion, would, of course be treated as a practice, with a dose constraint for members of the public of 300 $\mu$Sv $y^{-1}$. In these circumstances, a new assessment would be required as part of the application for redevelopment and the application could be refused if the criteria were not met). It should be noted that, using an annual probability of $10^{-4}$ of site redevelopment (Mobbs, 2007) and a risk per unit dose of 0.06 (ICRP, 1991), the dose criterion of 3 mSv $y^{-1}$ corresponds to a risk of 1.8 $10^{-8}$, well below the risk target used for disposal of low and intermediate waste (Environment Agency et al, 1996).

For exposure of the public from the build up of gas in a building if it were to be located on the landfill site post closure, the dose limit for members of the public was adopted (United Kingdom - Parliament, 2000). For $^{222}$Rn the criterion was taken as the Radon Action Level (NRPB, 2000b) of 200 Bq $m^{-3}$, again following the ICRP 81 approach for inadvertent intrusion.

**Table 1 Dose criteria adopted for selected scenarios**

| Scenario | Exposed person | Numerical value for dose criterion[+] |
|---|---|---|
| Operational phase | Landfill worker, sorter | 1 mSv $y^{-1}$ |
| Discharge of leachate to river | Member of the public | 300 $\mu$Sv $y^{-1}$ |
| Natural processes post closure | Member of the public | 20 $\mu$Sv $y^{-1}$ |
| Inadvertent intrusion post closure | Member of the public, excavation worker | 3 mSv $y^{-1}$ |
| Inhalation of landfill gas post closure | Member of the public | 1 mSv $y^{-1}$ for $^3$H and $^{14}$C, 200 Bq $m^{-3}$ for $^{222}$Rn |

[+] Criteria adopted for this study only; they do not represent formal advice from HPA-RPD or the regulators.

Since the models are linear, the disposal quantities reported in this work may be
scaled linearly if any of the dose constraints are revised.

# 2 RADIONUCLIDES AND WASTES

Information on the arisings and radionuclide content of radioactive wastes from
the nuclear industry is published in the 2004 United Kingdom radioactive waste
inventory (Defra, 2005a). These data were reviewed by Atkins Limited to identify
those waste streams which potentially meet the proposed definition of VLLW.
Several physical forms were identified, such as concrete building blocks,
graphite structures, cellulose, contaminated soil and metal (Atkins Limited,
2006). As a result of this review, Atkins Limited provided the following
information on seven waste streams that could potentially meet the proposed
definition of VLLW: producer, density, annual amount arising, activity
concentration, and radionuclide "fingerprint". Eleven radionuclides were then
selected for this study and these are given in Table 2. For those radionuclides
with radioactive decay progeny, the progeny with a shorter half-life than its
immediate parent was assumed to be in equilibrium with its parent; other
progeny were modelled explicitly if significant ingrowth was likely to occur in the
time period being considered. Although the proposed definition does not
explicitly include alpha activity, some decommissioning waste will contain some
alpha activity and therefore alpha emitting radionuclides were included in this
study for completeness. $^{226}$Ra was also included for completeness, as it can
arise in large quantities in wastes from non-nuclear industries.

Four example waste streams were also selected for the assessment:

- Harwell development laboratory Low Level Waste (LLW) which consists
  of glove boxes, fume cupboards, flasks cells, drains, ductwork, and the
  buildings housing these items

- Sellafield miscellaneous plants final decommissioning LLW which
  consists of building structural materials and miscellaneous soft waste

- Torness concrete reactor and non-reactor LLW which is comprised of
  concrete and reinforced concrete items

- Hartlepool reactor area LLW which consists of rubble and metal waste.

The activity concentration and the total future volumes of these waste streams
waste were taken from the 2004 United Kingdom radioactive waste inventory
(Defra, 2005a), and are given in Table 3.

**Table 2 Radionuclides selected for the study**

| Radio-nuclide | Half life (y) | Associated decay chain considered in secular equilibrium | Chain segments considered with the parent | Emission characteristics[#] |
|---|---|---|---|---|
| $^3$H | 12 | - | | β |
| $^{14}$C | 5730 | - | | β |
| $^{36}$Cl | $3.0\ 10^5$ | - | | β and γ |
| $^{60}$Co | 5.3 | - | | β and γ |
| $^{+90}$Sr | 29 | $^{90}$Y | | β and γ |
| $^{+137}$Cs | 30 | $^{137m}$Ba | | β and γ |
| $^{+226}$Ra | 1600 | $^{222}$Rn, $^{218}$Po, $^{218}$At (0.04%), $^{214}$Pb (99.96%), $^{214}$Bi, $^{214}$Po | $^{+210}$Pb, $^{210}$Po | α, β and γ |
| $^{232}$Th | $1.4\ 10^{10}$ | - | $^{+228}$Ra, $^{+228}$Th | α, β and γ |
| $^{+238}$U | $4.5\ 10^9$ | $^{234}$Th, $^{234m}$Pa (99.8%), $^{234}$Pa (0.2%) | $^{234}$U, $^{+230}$Th | α, β and γ |
| $^{239}$Pu | 24000 | - | | α, β and γ |
| $^{241}$Am | 430 | - | $^{+237}$Np | α, β and γ |
| Progeny | | | | |
| $^{+210}$Pb | 22 | $^{210}$Bi | | |
| $^{+227}$Ac | 22 | $^{227}$Th, $^{223}$Ra, $^{223}$Fr, $^{219}$Rn, $^{215}$Po, $^{211}$Pb. $^{211}$Bi, $^{211}$Po, $^{207}$Tl | | |
| $^{+228}$Ra | 5.7 | $^{228}$Ac | | |
| $^{+228}$Th | 1.9 | $^{224}$Ra, $^{220}$Rn, $^{216}$Po, $^{212}$Bi, $^{212}$Po (64.1%), $^{208}$Tl (35.9%) | | |
| $^{+230}$Th | $7.7\ 10^4$ | $^{+226}$Ra, $^{+210}$Pb, $^{210}$Po | | |
| $^{+235}$U | $7.0\ 10^8$ | $^{231}$Th | | |
| $^{+237}$Np | $2.1\ 10^6$ | $^{233}$Pa | | |

- not applicable

[+] indicates that progeny in secular equilibrium included

[#] α alpha radiation, β beta radiation, γ gamma radiation

**Table 3 Activity concentrations (Bq m$^{-3}$) and waste arisings of four example VLLW waste streams (Defra, 2005a)**

| Radionuclide | Harwell | Sellafield | Torness | Hartlepool |
|---|---|---|---|---|
| $^{3}$H | 0 | $8.3 \; 10^2$ | $1.9 \; 10^5$ | $2.0 \; 10^6$ |
| $^{14}$C | 0 | $3.0 \; 10^3$ | 6.5 | $2.0 \; 10^5$ |
| $^{36}$Cl | 0 | 0 | $2.3 \; 10^2$ | 0 |
| $^{60}$Co | 0 | $1.9 \; 10^3$ | $4.5 \; 10^1$ | $7.0 \; 10^5$ |
| $^{90}$Sr | $2.5 \; 10^6$ | $4.7 \; 10^5$ | 0 | $9.0 \; 10^3$ |
| $^{137}$Cs | $7.4 \; 10^6$ | $2.6 \; 10^6$ | 0 | $2.0 \; 10^4$ |
| $^{226}$Ra | 0 | 0 | 0 | 0 |
| $^{232}$Th | 0 | 0 | 0 | 0 |
| $^{238}$U | $3.0 \; 10^{-6}$ | $9.8 \; 10^2$ | 0 | 0 |
| $^{239}$Pu | $2.0 \; 10^5$ | $9.2 \; 10^4$ | 0 | 0 |
| $^{241}$Am | $1.7 \; 10^4$ | $7.9 \; 10^3$ | 0 | 0 |
| Total | $1.0 \; 10^7$ | $3.2 \; 10^6$ | $1.9 \; 10^5$ | $2.9 \; 10^6$ |
| Total waste volume (m$^3$) | 247 | 69000 | 2700 | 78 |

# 3     CONCEPTUAL MODEL OF LANDFILLS

The EC Landfill Directive (Commission of the European Communities, 1999) defines the class of a landfill site on the basis of the type of waste that the landfill can accept. There are three classes: Inert, Non-hazardous and Hazardous. The design specifications and operating practices are different for each of the landfill classes. Radioactive waste is not classified as hazardous waste, but as 'special waste' and hence it is not clear whether radioactively contaminated rubble would be disposed of in an inert landfill or in a hazardous landfill. Therefore to explore the range of indicative doses, dose assessments have been carried out for all three landfill classes.

Atkins Limited carried out a review of landfill disposal sites in the UK and Defra's interpretation of the Landfill Directive (Defra, 2005b) and used this information to define the design and basic data for three generic landfill classes for the purposes of this study (Atkins Limited, 2006). These three generic landfill sites, shown in Figure 1, do not represent any particular site but the characteristics were chosen so that they are representative of the variety of sites that might be encountered in the UK. Where there is a range of possible values for a site parameter, a conservative value has been selected. The conceptual model and the parameters used for this study were peer reviewed by the British Geological Survey (BGS) and their suggestions were incorporated. Discussions were also held with BGS, Atkins Limited, Defra and EA on the most appropriate value to choose for the distance between the landfill site and the nearest borehole or river into which the aquifer discharges. A distance of 250 m was agreed on as a cautious but representative value (Copplestone, 2006).

Two of the generic landfill classes determined for this study have a cap which is assumed to inhibit the ingress of rainwater but is also assumed to have a

specified lifespan. Once the cap is no longer functional, a fraction of the rainwater will pass through the cap and will enter the waste and hence leaching of the activity from the waste will start. In this generic study, one general form of VLLW is used to represent all the different kinds of waste streams. This general waste is assumed to be readily dissolvable with a release rate at the higher end of the reported range. The effective infiltration rate is determined from the rainfall minus evaporation and surface runoff, and is assumed to be about 320 mm y$^{-1}$ (Atkins Limited, 2006) for all three types of landfill site.
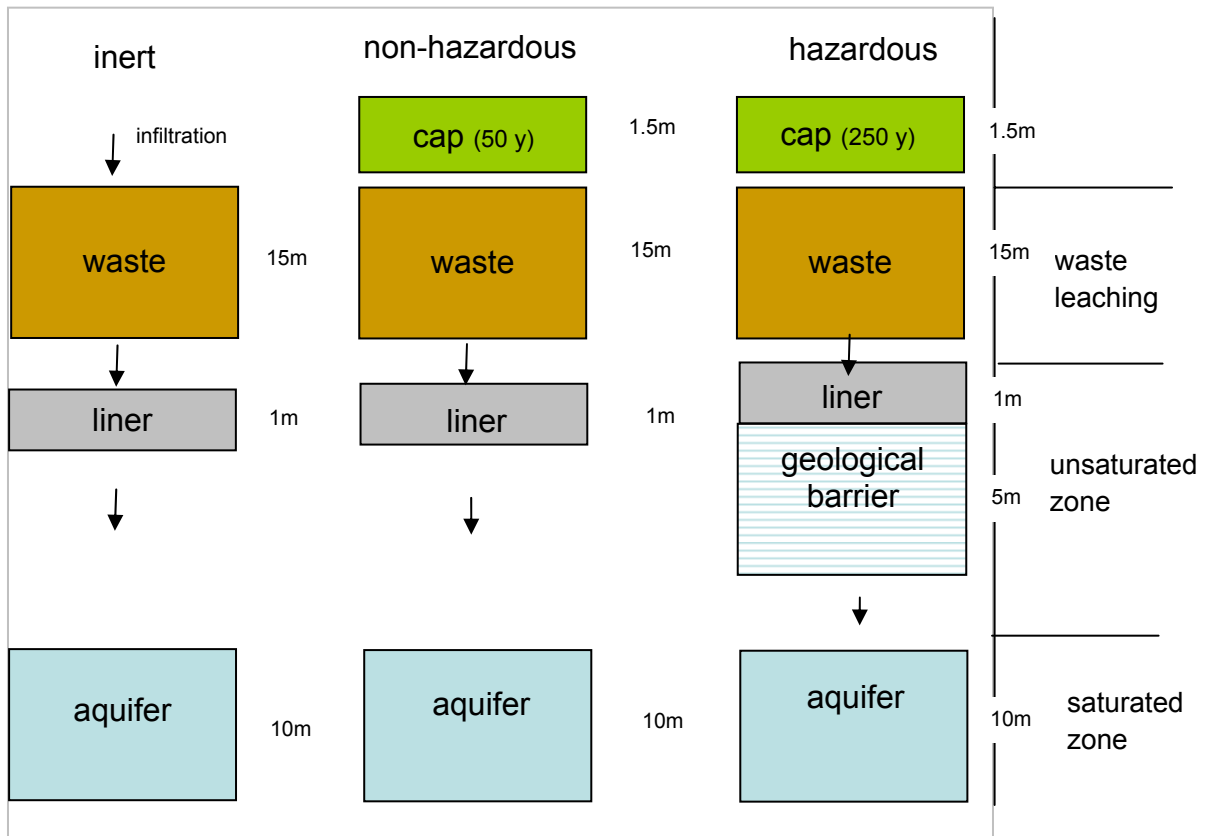


**Figure 1 Schematic diagram of conceptual landfill designs for the 3 landfill categories**

Once leaching has started, the leachate will begin to move downwards towards the aquifer. However all three landfill types have a clay liner system that will reduce the amount of leachate that can eventually reach the aquifer (the saturated zone).  In addition the hazardous type landfill will also have a geological barrier which comprises a low permeability mineral layer and potentially other geological material, above the aquifer. This geological barrier and the clay liner are unsaturated and are therefore modelled as part of the unsaturated zone.

The clay liner has a very low permeability and hence only a fraction of the leachate arising from the waste will seep through the liner and any excess leachate will accumulate above the liner and will need to be pumped out. This pumped leachate may be discharged to the local river after treatment. These two leachate routes were treated separately in the model, using conservative

assumptions. On the one hand, the leachate that penetrates the liner and reaches the aquifer was conservatively assumed to contain all the leached radioactivity, i.e. ignoring the radioactivity that is removed in the extracted leachate. On the other hand, the leachate collected at the landfill site during the operational and post-closure phase was also assumed to contain all the leached activity and to be discharged to a river. The dose arising from this pathway is discussed in Section 3.4.

Note that it is assumed for all landfill types that the liner keeps the same physical properties (permeability, hydraulic conductivity, porosity) over time. This assumption is reasonable for time periods of the order of thousands of years because these are intrinsic properties of the clay liner unlike that of an artificial liner made of man-made material.

Figure 1 also shows the typical geological environment of the three landfill types that were used as the basis for the migration calculations. Further details of the site characteristics and parameter values are given in Appendix B.

## 3.1        Inert type landfill

A landfill site that receives inert waste has a simple design, with a specified minimum permeability and thickness for the liner but with no requirement for a cap. The generic inert landfill site was therefore assumed to have no final cap and a 1 m thick clay liner (hydraulic conductivity of $1 \ 10^{-7} \ \text{m s}^{-1}$) with no additional geological barrier between the liner and the aquifer, i.e. the groundwater table is immediately below the clay liner. The aquifer is assumed to be highly permeable with the permeability of a typical major UK aquifer (Sherwood sandstone). A public borehole and a large river are assumed to be 250 m away from the site. Further details and parameter values are given in Appendix B.

## 3.2        Non-hazardous type landfill

A landfill site that receives non-hazardous waste is required to have a cap, a drainage system to collect leachate, and a specified minimum permeability and thickness for the liner, but there is no requirement for an additional geological barrier between the liner and the aquifer. Therefore the generic site was assumed to have a good plastic liner cap which lasts for 50 years and, as a result, no leachate is produced from the landfill in the first 50 years after closure. After that time, the infiltration rate is assumed to be the same as for the inert waste landfill. The unsaturated zone is assumed to be a 1 m thick clay barrier with a hydraulic conductivity of $1 \ 10^{-9} \ \text{m s}^{-1}$. The saturated zone immediately below it is assumed to be an aquifer with a permeability equivalent to the top of the range for a minor UK aquifer (it was assumed that it would be unlikely that permission will be granted for a non-hazardous landfill to be built over a major aquifer). A well and a small river are assumed to be 250 m away from the landfill site. Further details and parameter values are given in Appendix B.

## 3.3      Hazardous type landfill

A landfill site that receives hazardous waste is required to have a good quality cap, a drainage system to collect leachate, a specified minimum permeability and thickness for the liner, and an additional geological barrier between the liner and the aquifer. Based on the Environment Agency guidance (Environment Agency, 2003), the generic hazardous landfill site was assumed to have a clay cap that lasts for 250 years, after which time it ceases to be an effective barrier to infiltration and the natural infiltration rate applies. Although there may be minimal infiltration during this period, the amount of leachate produced is trivial and has therefore been ignored. The unsaturated zone is assumed to consist of a 1 m thick clay liner and a 5m thick geological clay barrier. The hydraulic properties are assumed to be the same for both clay layers ($1\ 10^{-9}$ m s$^{-1}$), but the densities of the two layers are different, with the compacted sealing clay liner having a higher bulk density (1.5 g cm$^{-3}$) than the natural clay (1.3 g cm$^{-3}$). The saturated zone is assumed to be an aquifer with a permeability equivalent to the top of the range applicable to a minor UK aquifer. Again, a well and a small river are assumed to be 250 m from the site.   Further details and parameter values are given in Appendix B.

## 3.4      Site timeline and capacity

For this study it has been assumed that the landfill site will be operational for 15 years and then closed.  A typical landfill site is assumed to have a full capacity of $1.2\ 10^{6}$ m$^{3}$ x 1.8 te m$^{-3}$ = $2.2\ 10^{6}$ te (see Table B3, Appendix B). Given a life span of 15 years the disposed tonnage per year is approximately $1.4\ 10^{5}$ te y$^{-1}$. These values were used for the calculations. Guideline figures of $10^{5}$ te y$^{-1}$ for the annual disposal rate and $10^{6}$ tonnes for the total capacity of the site were used to evaluate the results and draw conclusions.

Following closure the site will then continue to be managed and monitored (institutional control) for a period of 30 years, after which time inadvertent intrusion or redevelopment of the site can occur.  Thus, in reality, the management of leachate from the landfill will probably only occur while there is institutional control of the site.  Leachate produced during the operational phase of the site was not explicitly considered in this study.  However, the concentration in the leachate from an inert site (calculated immediately after closure) can be assumed to be broadly representative of the maximum concentration that could occur during the operational phase. For the hazardous and non-hazardous landfill type sites it was assumed that leachate is not produced until the cap has failed, i.e. at 50 and 250 years after closure, respectively. Therefore this leachate would arise after the time at which institutional control is assumed to have ended. However calculations were performed assuming that discharge of leachate to a river continues for several hundred years in order to give an indication of the doses that could occur if the leachate was collected and discharged to a river. These calculations can also be used to represent the doses that could be received from chronic overflowing of the site during the time when the liner effectively reduces the movement of the leachate to the aquifer. Thus, the assessment considered discharge of

leachate to river occurring immediately after closure of the site for the inert site, and at 50 years and 250 years after closure of the site for the non-hazardous and hazardous landfill types respectively.


# 4    OPERATIONAL AND MONITORING PHASE


Two scenarios were considered during the operational and monitoring phases of the landfill site.  Firstly the exposure to workers sorting the waste or working at the landfill site and secondly the annual doses to members of the public from the discharge of leachate pumped from the landfill site to a river, during the (extended) institutional control periods.  These scenarios are discussed in turn in the section below.

Two groups of workers were considered for the operational phase in this study: workers involved in sorting the waste and workers disposing of the wastes at the landfill site. Following the approach taken in (Harvey et al, 1995) a simple set of scenarios was developed to represent the exposure situations for each of the two groups of workers.

Sorting of the waste could take place at a dedicated facility or at the landfill site. The large quantities of concrete rubble generated during the dismantling of a nuclear power plant, and the high cost of transportation, suggest that the material would most likely be processed at a dedicated facility on or near the site of the plant being dismantled (US Nuclear Regulatory Commission, 1999). This facility could also be used for VLLW, in which case waste arriving at the landfill site would be disposed of directly with no further sorting. Alternatively the waste could be sent to the landfill site where it was sorted so that items for recycling were segregated and then the remainder disposed of. These processes were also assumed to be relevant to UK situation.  It is assumed that workers on the landfill site will either be involved in the sorting or disposal of the waste but not both hence the two groups of workers were considered separately.


## 4.1    General methodology for workers

The methodology for estimating the annual doses from the handling of concrete rubble at a sorting facility and its disposal at a landfill site is described in Appendix A and outlined here. Annual doses to workers were initially calculated assuming disposal of contaminated rubble material containing 1 Bq g$^{-1}$ of each radionuclide in turn, and then scaled appropriately. No account was made of dilution of the activity in the landfill since it was assumed that there are large volumes of VLLW waste being disposed and it is not mixed with uncontaminated material.

Four exposure pathways were considered:

- external exposure of skin from contamination on skin, and gloves if worn

- inhalation of resuspended material

- inadvertent transfer of particles from hands to mouth

- external exposure (whole body) from non-contact material.

Not all of these exposure pathways are applicable to both types of worker. The annual effective dose for each worker is, therefore, the sum of the doses from the relevant exposure pathways.

The annual effective dose to a worker is given by the following equation:

$$D_{eff} = D_{ext} + D_{skin} + D_{inh} + D_{ing}$$

where

$D_{eff}$ = total annual effective dose to the worker, Sv

$D_{ext}$ = annual effective dose from external exposure to the worker from contaminated material, Sv

$D_{skin}$ = annual effective dose from irradiation of skin by contaminated material in contact with the hands or gloves of the worker, Sv

$D_{inh}$ = annual committed effective dose from inhalation of resuspended contaminated material, Sv

$D_{ing}$ = annual committed effective dose from inadvertent ingestion of contaminated material, Sv

Full details of the exposure pathways and parameter values adopted are given in Appendix A.

## 4.2      Sorting scenario

Prior to reuse or recycling, concrete rubble is crushed and sorted. Any steel in the concrete is separated, monitored and sent to a ferrous metal scrap dealer if appropriate. Prior to being crushed the concrete is likely to be reduced to smaller blocks. During crushing water may be sprayed on the concrete to suppress dust. Concrete not recycled is sent to landfill for disposal. It is assumed that sorting involves large amounts of material that accumulate at one location and remain there during a significant period of time.

It was assumed that the sorting worker was exposed to external irradiation from a sorting stockpile or conveyor for an entire working year (2000 hours). This was represented by a cylindrical 1 m$^3$ source. It was also assumed that the worker spent his time sorting the waste manually and that the palms of the gloves become contaminated leading to irradiation of the skin through the gloves. The protection provided by the gloves against beta radiation was taken into account. For the inhalation pathway it was conservatively assumed the processing of concrete takes place in dry conditions so that during the crushing, sieving and sorting large amounts of airborne dust were created. It was conservatively assumed that this enhanced dust loading was present for the entire time spent

crushing (100 hours per year) and that any mask worn was not effective at stopping inhalation of dust particles. Inadvertent ingestion of dust was not considered as the workers were assumed to be wearing gloves. The exposure pathways and assumptions are summarised in Table 4 and described in detail in Appendix A.

## 4.3 Landfill scenario

It is assumed that a landfill operator distributes waste within the site using a mechanical excavator. It is further assumed that the work raises contaminated dust, and that the worker spends a large fraction of time outside, exposed to irradiation from the disposed waste in the landfill site. It is also conservatively assumed that any material covering the waste does not shield the operator from gamma and beta external irradiation. The landfill worker is assumed to be exposed to high dust levels when he is outside, and lower dust levels when he is inside the cab of the excavator. It is also conservatively assumed that the worker will neither be wearing gloves nor wash his hands before eating or smoking and therefore could have contaminated skin and hence could also inadvertently ingest the dust, which has built up on the skin, whilst smoking or eating during a break.

The highest annual dose will be received when the site is full of waste, or contains sufficient waste that the operator is exposed to it for an entire working year. The exposure pathways and assumptions for landfill workers are summarised in Table 4 and described in detail in Appendix A.

**Table 4 Exposure pathways and assumptions for sorting and landfill[a]**

| Parameter | Sorter | Landfill operator |
|---|---|---|
| Exposure time (h y$^{-1}$) | 100 (crushing) | 2000 (200 inside cab, 1800 outside) |
| | 2000 (handling) | |
| Pathways | External, inhalation of dust, skin | External, inhalation of dust, ingestion of dust, skin |
| External irradiation geometry | 1m from 1m$^3$ cylinder, no shielding, gamma only | 1m above 5m thick slab of infinite extent, no shielding, beta and gamma |
| Dust loading (g m$^{-3}$) | 10$^{-3}$ during crushing | 10$^{-3}$ outside, 10$^{-5}$ inside |
| Inadvertent ingestion rate (g h$^{-1}$) | 0 | 5 10$^{-3}$ |
| Inhalation rate (m$^3$ h$^{-1}$) | 1.69 | 1.69 |

[a] see Appendix A for detailed derivation of assumptions

## 4.4 Discharge of leachate to river

The leachate will be collected at the landfill site during the operational phase and during the institutional control period after closure of the site and it is assumed that this leachate will be discharged to a local river after treatment. The resulting annual doses were estimated by comparing the activity discharged with the Generalised Derived Constraint (GDC) (Bq y$^{-1}$) calculated for release to a small river (ie the release rate, continuing for 50 years, that gives rise to a

dose of 0.3 mSv y$^{-1}$) (NRPB, 2000a); (NRPB, 2002); (HPA, 2006) using the following equation:

$$\text{DPU}_{\text{leachate\_river}} = (\frac{\text{AC}_{\text{leachate}} \; \text{Q}_{\text{inf}}}{\text{GDC}_{\text{river}}}) \; \text{DC}$$

where

DPU$_{\text{leachate\_river}}$ = annual dose rate to members of the public if the collected leachate is discharged to local river (mSv y$^{-1}$)

AC$_{\text{leachate}}$ = activity concentration in the leachate from the waste at the time of discharge, (Bq m$^{-3}$), calculated as described in Section B2.2 in Appendix B.

GDC$_{\text{river}}$ = Generalised Derived Constraints for discharge into rivers (Bq y$^{-1}$). For nuclides for which GDC$_{\text{river}}$ values are not available ($^{36}$Cl and $^{232}$Th), their values were calculated using the biosphere model BIOS (Martin et al, 1991) set up to represent the same river.

Q$_{\text{inf}}$ = infiltration volume in a year (m$^3$ y$^{-1}$) which is assumed to be the same as the amount discharged to river (m$^3$ y$^{-1}$)

DC = Dose Constraint (0.3 mSv y$^{-1}$).

It should be noted that the activity concentration in the leachate varies from one landfill type to another depending on the time at which leaching starts, and it also varies with time. Since the infiltration rate is assumed to remain constant, the highest activity concentration in the extracted leachate will occur in the first year of leaching. For the inert landfill site, AC$_{\text{leachate}}$ is therefore the value calculated at time zero (no delay). For the non-hazardous and hazardous landfill sites, AC$_{\text{leachate}}$ is calculated at year 50 and year 250 respectively. It is conservatively assumed that all the leachate is collected for discharge and that none of the leachate migrates to the aquifer. The discharge during the operational period is not considered explicitly, but the activity concentration would be less than that calculated for the inert site at the post-closure stage because the leaching process takes time (in fact, not all the radioactivity will be instantly available for leaching as is assumed here).

The GDC calculations are based on a constant annual discharge, assumed to continue over 50 years and allows for the build-up of radioactivity in the environment. Hence the doses calculated using this approach implicitly assume that the leachate is collected and discharged into a local river for 50 years and that the concentration of the radionuclide in the leachate remains unchanged over these 50 years. Although this is a reasonable assumption for long-lived radionuclides, it could be a pessimistic assumption for short-lived radionuclides, e.g. $^{60}$Co, since radioactive decay will reduce the concentration in the leachate over this time period. However, a comparison of the doses from $^{60}$Co arising from a constant discharge for 1 year and the doses from the same constant discharge over 50 years, showed that the doses were very similar since $^{60}$Co decays before it has sufficient time to build up in the environment. Hence, this approach is reasonable for $^{60}$Co and may also be valid for other short-lived radionuclides.

# 5 POST-CLOSURE PHASE

Three scenarios that can lead to the exposure of people from the disposal of VLLW in a landfill in the post closure phase were considered in the study and are described below. The first, migration with the groundwater, is likely to occur. The other two scenarios are less likely and hence have an associated low annual probability of occurrence.

## 5.1 Migration with the groundwater

As described in Section 3, once water infiltrates the site and waste leaching starts, the leachate containing dissolved radionuclides will begin to move downwards and eventually it will reach the top of the clay liner system. It is assumed that the clay liner is effectively impermeable for a period of time, specific to each landfill type. Afterwards, the leachate starts to move downwards through the clay liner and any other geological barrier present and eventually it reaches the underlying aquifer. The low permeability of the clay liner will reduce the amount of leachate that can reach the aquifer. Once reaching the aquifer, the radionuclides in the leachate are transported along with the groundwater and are then released into the biosphere. Two exposure situations were considered: one is a well sunk into the aquifer which withdraws contaminated groundwater that is subsequently used for drinking purposes; and the other is the discharge of contaminated groundwater into a local stream or river which is used to irrigate farmland, resulting in the build-up of radionuclides in the surface soil and contamination of the biosphere.

The migration in the unsaturated zone (the clay and geological barriers) was modelled using HYDRUS-1D (Simunek et al, 2005). HYDRUS-1D was used to estimate the time of the initial breakthrough to the aquifer, ie the transit time in the unsaturated zone, and the concentration breakthrough curve for each of the selected radionuclides at the groundwater table. Details of the models and parameters used are given in Appendix B.

The groundwater solute transport model TROUGH-1D (Gilby and Hopkirk, 1985) was used to model the transport of radionuclides in the aquifer. The processes modelled were advection, dispersion, radioactive chain decay and retardation (sorption), see Appendix B for details.

Annual doses from the consumption of drinking water were calculated using the activity concentration in the aquifer at a point 250 m downstream from the landfill site. Annual doses from the discharge of groundwater to river and the subsequent transfer in the biosphere were estimated using the BIOS code (Martin et al, 1991). In both cases, the annual dose initially increases with time rising to a peak (which may last for decades) and then decreases again. The peak annual dose was compared with the dose criterion.

The peak annual doses of some radionuclides (especially the highly sorbed ones) occur at long times in the future, and therefore the results are associated with a considerable degree of uncertainty due to the potential changes to the

biosphere over these long timescales. The approach taken in this study was to run the models for a sufficiently long time to ensure that the peak dose was reached, and the variation of dose over time was explored.

## 5.2 Redevelopment of the site

It is likely that the approved closure plan for a landfill will involve a period of continued control and maintenance until the landfill is considered to no longer pose a pollution hazard to groundwater. It was assumed in this study that this post-closure institutional management would last for 30 years, during which time occupation or use of the site by members of the public will be prevented. Once this active management period is over there is some possibility that the site could be redeveloped in the future, e.g. for a residential housing estate, when people are not aware of the contamination history of the site. Although the annual probability of redevelopment of derelict land in UK is relatively low (approximately $10^{-4}$ per year (Mobbs, 2007)), there is an increasing overall probability that the site will be redeveloped as the time increases since the control of the site is lost. It was conservatively assumed for the purposes of this generic study that the site is redeveloped as soon as the period of management ends and that knowledge of the presence of VLLW at the site has been lost. (Redevelopment of the site with the knowledge of its history would entail a new and specific risk assessment and it would be judged on its overall merits and hence is not relevant to this study). For the calculations detailed in the two sections below, radioactive decay was ignored during the operational phase of the landfill site, assumed to be 15 years. (This is a conservative assumption for the short lived radionuclide $^{60}$Co which has a half life of 5.3 years). However once the landfill site was closed radioactive decay was considered over the 30 year period between the closure of the site and the inadvertent intrusion or redevelopment of the site.

### 5.2.1 Inadvertent intrusion

Two alternative situations were considered: excavation of the site by construction workers; and subsequent residence on the site by members of the public. In both cases, it was assumed that the redevelopment of the site results in contaminated material being disturbed and brought to the surface. The exposure pathways for the construction workers were assumed to be external irradiation, inhalation of dust and inadvertent ingestion of dust. The exposure pathways for the residents were external irradiation, inhalation of dust, inadvertent ingestion of dust and ingestion of vegetables grown in the garden. The annual doses were estimated using the CONLAND model (Oatway and Mobbs, 2003) and further details of the parameters are given in Appendix B.

### 5.2.2 Exposure to radioactive gases in a house on the site

Some radionuclides may escape from the landfill site in gaseous form, e.g. $^{3}$H and $^{14}$C. As waste streams from nuclear power stations contain little organic material, it is unlikely that large quantities of landfill gas such as methane,

carbon dioxide and hydrogen will be generated from the VLLW. Nevertheless, isotopes of hydrogen, oxygen and carbon could be released through corrosion. If $^{226}$Ra or $^{238}$U are disposed of then radon ($^{222}$Rn), a decay product, is a potential contributor to the landfill gas. Note that for this study $^{36}$Cl was not considered to be chemically available for release as a gas.

The potentially exposed group for this scenario is assumed to be the residents of a house that might be constructed on top of the landfill immediately after the end of the site management and control period. The exposure pathway is inhalation of radioactive gas inside the house. The peak indoor activity concentration in air of a radionuclide (Bq m$^{-3}$) was approximated by the total activity released into the house in the first year after it was built, divided by the volume of air passing through the house in a year. Details of the gas model and parameters are given in Appendix B.

The estimated concentration of radon was compared with the radon action level (200 Bq m$^{-3}$, (NRPB, 2000b)) and the annual doses from $^3$H and $^{14}$C were compared with a 1 mSv annual dose criterion, see Table 1.

# 6 QUANTITY OF VLLW CORRESPONDING TO ASSUMED DOSE CRITERIA

The maximum quantity of VLLW that could be disposed of in a landfill was estimated by comparing the annual dose arising from a landfill site that was full of material contaminated at the level given in the proposed definition with the appropriate assumed dose criterion. This was done for each radionuclide in turn, effectively assuming that the waste was only contaminated with that particular radionuclide. It was assumed that the site did not receive any uncontaminated waste or waste containing a lower radionuclide concentration.

## 6.1 Landfill and sorting worker

For the workers, the VLLW disposal rate is more important than the overall capacity of the site since the dose received in any one year depends on the quantity processed and disposed of in that year. Once the quantity of waste received at the site in any one year is sufficiently large that a worker is exposed continuously during an entire working year then the dose received is dependent on the radionuclide concentration in the waste rather than the amount of waste disposed of. (Twice as much waste would expose twice as many workers to the same dose.) The external dose model for the landfill worker implicitly assumes that they are processing at least 36 tonnes of waste per hour since this would give the same external dose rate as the 'semi-infinite source' assumed (see Appendix A). This corresponds to a minimum disposal rate of about half of the average value of 1.4 10$^5$ te y$^{-1}$ given above.

Therefore, if it is assumed that one landfill worker processes the entire 1.4 10$^5$ te y$^{-1}$, then

$$ATE_{vllw} \left(te \ y^{-1}\right) = \left( \frac{Dose \, Criterion \ \left(Sv \ y^{-1}\right) \times \ Disposed \ tonnage \ \left(te \ y^{-1}\right)}{Doseperunitconc\left(Sv \ y^{-1}/(Bq \ g^{-1})\right) \times \ AC_{vllw}\left(Bq \ g^{-1}\right)} \right)$$

where

$ATE_{vllw}$ (te $y^{-1}$) = tonnage per year that can be disposed of while ensuring that the annual doses to a worker do not exceed the assumed dose criterion

Dose Criterion = 1 mSv $y^{-1}$ (value assumed for workers, see Table 1)

Doseperunitconc = dose to worker (Sv in a year) per 1 Bq $g^{-1}$ in waste assuming that a worker processes approximately $10^5$ te $y^{-1}$

Disposed tonnage = 1.4 $10^5$ te $y^{-1}$

$AC_{vllw}$ = activity concentration of radionuclide i in the VLLW in the proposed definition (Bq te$^{-1}$) (4 MBq te$^{-1}$ of beta/gamma activity or 40 MBq te$^{-1}$ in the case of wastes containing solely $^{14}$C or tritium). For the purposes of this work it has been assumed that an activity concentration of 4 MBq te$^{-1}$ for alpha activity applies.

## 6.2       Leachate discharge, migration and redevelopment

For these scenarios, the estimated annual doses depend upon the total activity that has been disposed of to the site.

$$Y^i_{max} = \min\left( \frac{Annual \ dose \ criterion \ (Sv)}{Peak \ Annual \ Dose \ (Sv \ MBq^{-1})} \right)$$

where

$Y^i_{max}$ = activity of radionuclide *i* that ensures that assumed dose criterion is not exceeded (MBq)

Peak Annual Dose = peak annual dose from leachate to river, migration or site intrusion, per MBq disposed of in the site (Sv received per MBq)

Annual dose criteria are given in Table 1.

For $^{222}$Rn the criterion is taken as the Radon Action Level (NRPB, 2000b) of 200 Bq m$^{-3}$. Therefore the calculation is:

$$Y^i_{max} = \left( \frac{Radon \ action \ level \ (Bq \, m^{-3})}{Conc \ per \, MBq \ (Bq \, m^{-3}/MBq)} \right)$$

where

Conc per MBq = concentration of gas per MBq disposed of in the site.

The total mass of VLLW containing only radionuclide *i*, as defined by the proposed definition, that can be disposed of at a landfill site is then obtained by:

$TE_{vllw} = Y^i_{max} /AC_{vllw}$

where

$TE_{vllw}$ = tonnage of VLLW that can be disposed of whilst ensuring that the dose criteria adopted in this study are not exceeded.

# 7 SENSITIVITY ANALYSIS

## 7.1 Sensitivity to changes in parameter values

A number of (generally conservative) assumptions were made in deriving the parameters for the base case calculations and the results of the calculations will be sensitive to the values chosen for many of the parameters. The effect of changes in parameter values on the results was studied for a few important parameters. The approach taken was to vary the value of one parameter at a time. A full uncertainty analysis in which a number of parameter values are varied simultaneously was beyond the scope of this assessment.

Three radionuclides were selected to represent the range of radionuclides that were modelled in the assessment. These were: $^{14}C$ (long radioactive half-life (5700 years), mobile in the environment and appears to be the most restrictive radionuclide for migration); $^{137}Cs$ (representative of beta/gamma emitting radionuclides and has a relatively short half-life (30 years)) and $^{238}U$ (head of a radionuclide chain and has a long radioactive half-life of $4.5 \ 10^9$ years).

The sensitivity studies are summarised in the following table. (The parameter variations for migration were chosen primarily to investigate possible higher estimated annual doses).

**Table 5 Sensitivity studies performed for $^{14}$C, $^{137}$Cs and $^{238}$U**

| Scenario | Sensitivity study | Description |
|---|---|---|
| Landfill workers | Soil covering waste | Shielding of waste by a covering of clean soil. |
| Migration from Inert landfill site | No sorption by waste | $K_d$ for all nuclides set to zero, giving maximum release rate from waste. |
| | No liner or cracked liner | Radionuclides leached from waste go straight into the aquifer. |
| | Shorter distance to well or river | 100 m. |
| | Poor sorption in aquifer | $K_d$ value in the aquifer lowered by one order of magnitude, giving faster movement of the solute in the aquifer. |
| | Coastal site | Aquifer discharges to marine environment. |
| Discharge from Hazardous landfill site | Bathtubbing/flooding | Blockage of the drainage system causes overflow of leachate over the sides of the landfill site onto an area of soil used for crops and livestock. One event in a year. |
| Inadvertent intrusion, gas in house | Time before redevelopment of site | Shorter time (10 years) and longer time (100 years). |
| Discharge to river | Larger rivers | Leachate is discharged to a larger river, with a flow rate of 2.5 m$^3$ s$^{-1}$ for the hazardous and non-hazardous landfill types and 500 m$^3$ s$^{-1}$ for the inert landfill type. |

## 7.2      Impact of climate change and coastal erosion

The impact of climate change and coastal erosion were beyond the scope of this study.  However this section discusses the likely results.  A report for the Committee for Radioactive Waste Management (CoRWM) (Shelton, 2004) on the impact of coastal erosion states that climate change will increase the rates of coastal erosion.  It concludes that it is not possible to arrive at definite conclusions concerning the viability of a shallow radioactive waste disposal facility at any particular site.  However it is clear that some sites will be more prone to the effects of coastal erosion than others.

A study on the assessment of the consequences of on-site disposal at Hinkley Point A (Kelly et al, 2006b) assumes that the major impact of coastal erosion in terms of biosphere modelling is to remove material from the far terrestrial and near terrestrial compartment and transfer them to the foreshore compartment. However it concludes that the annual doses will be negligible due to dilution on the foreshore and that higher annual doses and risks could arise if bathtubbing (see Table 5 for description) of the radionuclides into the terrestrial biosphere could occur.

The results of the sensitivity analysis, see Section 8.3.3 on release of groundwater to the coastal area and Section 8.3.4 on the bathtubbing scenario, can therefore be used to give an indication of the likely annual doses from climate change and coastal erosion. The annual doses obtained from the

redevelopment of the site can also be used to give an indication of the annual doses from direct exposure to exposed wastes following erosion of the site.


# 8    RESULTS


## 8.1   Base case results

### 8.1.1  Annual doses to workers

The estimated annual doses to landfill operators and sorting workers, assuming they spend a working year processing about $10^5$ tonnes of VLLW at an activity concentration based at the proposed levels, are given in Table 6 for each radionuclide.   Details of the calculations are given in Appendix A. It is conservatively assumed that the working practices are the same for inert, non-hazardous and hazardous sites and therefore that these annual doses apply to all three site types.  This may be a pessimistic assumption for hazardous landfill sites where greater precautions are likely to be taken.

The estimated annual doses to sorters from any one radionuclide are less than or equal to 30 μSv and are therefore well below the dose criterion of 1 mSv y$^{-1}$. In fact they are within the dose level normally considered as "trivial", represented by a few tens of μSv per year (IAEA, 1988). The estimated annual doses to the landfill workers are higher and for $^{60}$Co, $^{137}$Cs and $^{226}$Ra they are estimated to be greater than or equal to the dose criterion of 1 mSv y$^{-1}$, with the annual doses from $^{239}$Pu and $^{241}$Am at about 0.4 mSv.

For those radionuclides with annual doses greater than or near the dose criterion of 1 mSv y$^{-1}$, the dominant exposure pathways are external irradiation from the waste ($^{60}$Co, $^{137}$Cs and $^{226}$Ra) and inhalation of contaminated dust ($^{239}$Pu and $^{241}$Am).

The tonnage per year, estimated on the basis that the annual doses to a worker should not exceed the dose criterion of 1 mSv y$^{-1}$, is shown in Table 7.  The methodology for calculation of tonnage from annual dose is given in Section 6. The annual doses in Table 6 are reflected in the tonnage values in Table 7: for $^{60}$Co, $^{137}$Cs, $^{226}$Ra, $^{232}$Th, $^{239}$Pu and $^{241}$Am the annual capacity based on the landfill worker doses is less than or close to the guideline figure of $10^5$ tonnes assumed to be typical of the mass of waste disposed to a site in one year.

It should be noted that the generic methodology for calculating the annual dose to landfill workers is conservative, particularly with respect to the external irradiation pathway, as it assumes that the workers are exposed to external irradiation from an area of exposed waste for their entire working year and that no shielding is provided by the waste. In reality the waste site would be filling gradually and it is common practice at landfill sites to cover the waste with a layer of clean soil. Hence the area of exposed waste would be only a fraction of the area of the site and the landfill workers would probably not spend all their time working on or near the exposed waste. It has also been conservatively assumed that the dust generated from the waste is not sprayed down. Therefore the value for the dust loading used in the calculations is likely to be an

overestimate and the doses from inhalation are likely to be lower in reality. It was assumed that the landfill workers did not wear gloves and that neither sorting nor landfill workers wore effective respiratory protection equipment. Doses to skin and from inhalation may be reduced by use of appropriate PPE (gloves, mask).

The estimated annual doses to landfill workers could be reduced following a re-examination of the conservative assumptions used in the calculations. In practical terms it might be possible to ensure that the annual doses to landfill workers from all radionuclides are below 1 mSv y$^{-1}$ even with an annual capacity close to the guideline value of $10^5$ te y$^{-1}$ by applying some requirements for covering the waste and for working practices (to restrict the time spent by the workers near the waste). Hence it would be necessary to do a more detailed site-specific assessment to determine the amount of VLLW waste that could be disposed of at a particular site if the radionuclides $^{60}$Co, $^{137}$Cs, $^{226}$Ra, $^{239}$Pu and $^{241}$Am are present as a significant fraction and a 1 mSv y$^{-1}$ dose criterion is adopted for landfill workers.

It should be noted that the annual doses to landfill workers would increase by 30% or less if they were also assumed to sort the waste.

**Table 6 Calculation of doses to landfill workers based on activity concentration at proposed levels[#]**

| Radionuclide | Sorting worker (Sv y$^{-1}$) | Landfill worker (Sv y$^{-1}$) | Dominant pathway (based on doses to landfill worker) |
|---|---|---|---|
| $^3$H | 1 10$^{-10}$ | 9.10$^{-9}$ | Inhalation and inadvertent ingestion |
| $^{14}$C[†] | 2 10$^{-8}$ | 5 10$^{-8}$ | Skin, inhalation and inadvertent ingestion |
| $^{14}$C* | 2 10$^{-7}$ | 5 10$^{-7}$ | Skin, inhalation and inadvertent ingestion |
| $^{36}$Cl | 9 10$^{-8}$ | 5 10$^{-7}$ | External |
| $^{60}$Co | 3 10$^{-5}$ | 6 10$^{-3}$ | External |
| $^{+90}$Sr | 2 10$^{-7}$ | 3 10$^{-5}$ | External |
| $^{+137}$Cs | 6 10$^{-6}$ | 1 10$^{-3}$ | External |
| $^{+226}$Ra | 2 10$^{-5}$ | 4 10$^{-3}$ | External |
| $^{232}$Th | 2 10$^{-5}$ | 4 10$^{-4}$ | Inhalation |
| $^{+238}$U | 4 10$^{-6}$ | 7 10$^{-5}$ | Inhalation |
| $^{239}$Pu | 2 10$^{-5}$ | 4 10$^{-4}$ | Inhalation |
| $^{241}$Am | 2 10$^{-5}$ | 4 10$^{-4}$ | Inhalation |

[#] assuming continuous exposure throughout the working year

[+] progeny in secular equilibrium included

[†] 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

**Table 7 Tonnage per year corresponding to 1 mSv y$^{-1}$ to workers (based on 40 MBq te$^{-1}$ for $^3$H and $^{14}$C and 4 MBq te$^{-1}$ for all other radionuclides, including $^{14}$C)**

| Radionuclide | Sorting worker | Landfill worker |
|---|---|---|
| $^3$H | 1 10$^{12}$ | 2 10$^{10}$ |
| $^{14}$C$^\dagger$ | 8 10$^9$ | 3 10$^9$ |
| $^{14}$C* | 8 10$^8$ | 3 10$^8$ |
| $^{36}$Cl | 2 10$^9$ | 3 10$^8$ |
| $^{60}$Co | 5 10$^6$ | 3 10$^4$ |
| $^{+90}$Sr | 6 10$^8$ | 4 10$^6$ |
| $^{+137}$Cs | 2 10$^7$ | 1 10$^5$ |
| $^{+226}$Ra | 6 10$^6$ | 3 10$^4$ |
| $^{232}$Th | 7 10$^6$ | 4 10$^5$ |
| $^{+238}$U | 4 10$^7$ | 2 10$^6$ |
| $^{239}$Pu | 7 10$^6$ | 4 10$^5$ |
| $^{241}$Am | 8 10$^6$ | 4 10$^5$ |

+ progeny in secular equilibrium included

$^\dagger$ 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

### 8.1.2 Base case results for annual doses to public from leachate management

The estimated annual doses to the public resulting from discharge of landfill leachate to a very small river for inert, non-hazardous and hazardous landfill type sites are shown in Table 8. Details of the calculations are given in Appendix B.

It is noted from Table 8 that for the inert site the estimated annual dose from $^{14}$C (at 4 or 40 MBq te$^{-1}$) is at least an order of magnitude greater than the assumed dose criterion of 300 µSv y$^{-1}$.  The annual dose from $^{60}$Co also exceeds the criterion. The annual doses from the other radionuclides are lower than the criterion and, except for $^{36}$Cl, they are also at or below the 20 µSv y$^{-1}$ dose criterion below which optimisation of discharges is not required (IAEA, 1988). The estimated annual doses from the non-hazardous and hazardous sites are less than 300 µSv y$^{-1}$ for all radionuclides, except for $^{14}$C whose value is similar to that of the inert site.  This is because, for the non-hazardous and hazardous sites, leachate is assumed not to be produced until infiltration into the site occurs ie after the cap fails (50 and 250 years after closure for the non-hazardous and hazardous sites respectively).  This delay does not make a significant difference for a long-lived radionuclide such as $^{14}$C but it does reduce the annual doses of short-lived radionuclides such as $^{60}$Co.

These annual dose results are reflected in the calculated tonnage values given in Table 9 where it can be seen that for $^{14}$C (at 4 or 40 MBq te$^{-1}$) and $^{60}$Co the amount of waste that can be disposed of whilst ensuring that the doses remain below the assumed criterion is below the landfill capacity guideline value of 10$^6$ tonnes. For the other radionuclides it is at least one order of magnitude above this guideline value.

Note that these results are based on a conservative calculation in which it is assumed that firstly all the leachate percolating through the waste is collected by the clay liner (none percolates through the liner) and secondly that it is subsequently discharged to a very small river. The first assumption assumes that the clay liner is unsaturated whereas, in reality, the clay layer for the inert landfill is likely to be fully saturated. Table 10 shows the percentage of leachate which would be collected assuming a fully saturated clay liner, calculated using the Darcy equation (see Appendix B). For the inert site, practically all of the leachate will seep through the liner and go directly to the underlying aquifer and hence no leachate would be collected, resulting in no doses from the discharge of leachate. However, for the hazardous and non-hazardous landfill sites, (which have liners with a hydraulic conductivity two orders of magnitude lower than the liner assumed to be used for an inert landfill site), it can be seen that even if the liner was saturated then there would be only a 10% reduction in the amount of leachate that would be collected and discharged.

As far as the assumption of the discharge to a very small river is concerned, the annual doses would be lower if the river flow rate was higher. The sensitivity of the annual dose from leachate discharge to river size is examined in Section 8.3.2. A large river (500 $m^3$ $s^{-1}$ flow rate) would reduce the annual dose by at least two orders of magnitude for the radionuclides considered in the sensitivity study.

Therefore, for the inert site it is likely that the annual doses from leachate discharge to river will be very much smaller than those given in Table 8 since it is expected that the leachate will percolate through the liner rather than be collected. If this is indeed the case then there would be no need for restrictions on the quantity disposed of at an inert site for any of the radionuclides considered, based on the calculations presented here and the dose criteria adopted for this study.

For a non-hazardous or hazardous type of site there is no need to limit the quantity of waste containing radionuclides other than $^{14}$C that is disposed of, based on these calculations and the dose criteria adopted for the study. This work indicates that a more detailed site-specific assessment would be required to determine the amount of VLLW waste containing $^{14}$C (at 4 or 40 MBq te$^{-1}$) that could be disposed of at a particular non-hazardous or hazardous landfill type site. The size of the receiving river and the activity concentration of $^{14}$C in the waste averaged over the whole site are key parameters that would determine whether the resulting annual doses would fall below 300 µSv y$^{-1}$.

**Table 8 Annual doses to public from discharges of leachate to a very small river (Sv), based on disposal of 2 $10^6$ tonnes of waste in landfill types of sites with activity concentration at proposed levels. Loss of leachate through the liner is not considered.**

| Radionuclide | Landfill type | | |
|---|---|---|---|
| | Inert | Non-hazardous | Hazardous |
| $^3$H | 7 $10^{-6}$ | 4 $10^{-7}$ | 6 $10^{-12}$ |
| $^{14}$C$^†$ | 3 $10^{-3}$ | 3 $10^{-3}$ | 3 $10^{-3}$ |
| $^{14}$C* | 3 $10^{-2}$ | 3 $10^{-2}$ | 3 $10^{-2}$ |
| $^{36}$Cl | 7 $10^{-5}$ | 7 $10^{-5}$ | 7 $10^{-5}$ |
| $^{60}$Co | 9 $10^{-4}$ | 1 $10^{-6}$ | 5 $10^{-18}$ |
| $^{+90}$Sr | 2 $10^{-5}$ | 5 $10^{-6}$ | 4 $10^{-8}$ |
| $^{+137}$Cs | 2 $10^{-5}$ | 5 $10^{-6}$ | 5 $10^{-8}$ |
| $^{+226}$Ra | 2 $10^{-5}$ | 2 $10^{-5}$ | 2 $10^{-5}$ |
| $^{232}$Th | 7 $10^{-7}$ | 7 $10^{-7}$ | 7 $10^{-7}$ |
| $^{+238}$U | 2 $10^{-6}$ | 2 $10^{-6}$ | 2 $10^{-6}$ |
| $^{239}$Pu | 8 $10^{-7}$ | 8 $10^{-7}$ | 8 $10^{-7}$ |
| $^{241}$Am | 1 $10^{-6}$ | 9 $10^{-7}$ | 7 $10^{-7}$ |

+ progeny in secular equilibrium included

$^†$ 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

**Table 9 Overall tonnage (tonnes) that can be disposed of in a landfill whilst ensuring doses remain below the assumed criterion based on the annual doses from discharges of leachate to a very small river (waste concentrations are 40 MBq te$^{-1}$ for $^3$H and $^{14}$C and 4 MBq te$^{-1}$ for all other radionuclides including $^{14}$C). Loss of leachate through the liner is not considered.**

| Radionuclide | Landfill type | | |
|---|---|---|---|
| | Inert | Non-hazardous | Hazardous |
| $^3$H | 9 $10^7$ | 2 $10^9$ | 1 $10^{14}$ |
| $^{14}$C$^†$ | 2 $10^5$ | 2 $10^5$ | 2 $10^5$ |
| $^{14}$C* | 2 $10^4$ | 2 $10^4$ | 2 $10^4$ |
| $^{36}$Cl | 1 $10^7$ | 1 $10^7$ | 1 $10^7$ |
| $^{60}$Co | 7 $10^5$ | 5 $10^8$ | 1 $10^{20}$ |
| $^{+90}$Sr | 4 $10^7$ | 1 $10^8$ | 2 $10^{10}$ |
| $^{+137}$Cs | 4 $10^7$ | 1 $10^8$ | 1 $10^{10}$ |
| $^{+226}$Ra | 3 $10^7$ | 3 $10^7$ | 4 $10^7$ |
| $^{232}$Th | 1 $10^9$ | 1 $10^9$ | 1 $10^9$ |
| $^{+238}$U | 4 $10^8$ | 4 $10^8$ | 4 $10^8$ |
| $^{239}$Pu | 8 $10^8$ | 8 $10^8$ | 9 $10^8$ |
| $^{241}$Am | 6 $10^8$ | 7 $10^8$ | 9 $10^8$ |

+ progeny in secular equilibrium included

$^†$ 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

**Table 10 Percentage of leachate migrating through a fully saturated clay liner estimated using Darcy's equation**[*]

| Type of site | % leachate collected | % leachate migrating through the liner |
|:---:|:---:|:---:|
| Inert | 0 | 100 |
| Non-hazardous | 89 | 11 |
| Hazardous | 90 | 10 |

[*] see Appendix B

### 8.1.3 Site Post-closure

Annual doses to members of the public (including construction workers) post site closure were calculated from migration of radionuclides in the groundwater, from intrusion into the site 30 years after site closure and from gas build up in a building on site 30 years after site closure. The peak annual doses from migration and the annual doses to the construction workers and housing residents from intrusion into the site are given in Table 11 and Table 12 respectively. The corresponding capacities of the landfill sites are given in Table 13. Details of the calculations are given in Appendix B.

For the inert site the peak annual dose from migration for $^{14}$C (at 4 or 40 MBq te$^{-1}$) is at least an order of magnitude greater than the criterion of 20 µSv y$^{-1}$ and the peak annual dose from $^{36}$Cl is close to the criterion. The time of the peak dose for these two radionuclides is approximately 20 years after closure. For the other radionuclides the peak annual doses are well below the criterion and are at the level considered as "trivial"; and, for all except $^{3}$H, the peak dose arises far into the future (greater than 1000 years).

The estimated peak annual doses from the non-hazardous and hazardous sites are generally lower than those from the inert site and occur later. Although the estimated peak annual doses from all radionuclides are below the dose criterion of 20 µSv y$^{-1}$, the peak annual dose from $^{14}$C (at 40 MBq te$^{-1}$) for the non-hazardous site is only a factor of 4 below it. However, it is estimated to arise about 100 years after closure. If the $^{14}$C concentration in the waste is limited to 4 MBq te$^{-1}$, the peak annual dose will be a factor of 40 below the dose criterion. For the hazardous site it can be seen that only 5 radionuclides $^{14}$C, $^{36}$Cl, $^{238}$U, $^{239}$Pu and $^{241}$Am (and its daughter $^{237}$Np) are sufficiently long lived and relatively mobile to give non zero doses and that the estimated peak doses from these radionuclides are about 1% or less of the dose criterion, arising several thousand years in the future.

**Table 11  Peak annual doses to adults due to migration based on disposal of $2 \, 10^6$ tonnes with activity concentrations at proposed levels**

| Radionuclide | Inert site | | Non-hazardous site | | Hazardous site | |
|---|---|---|---|---|---|---|
| | Peak annual dose (Sv) | Time of peak (years after closure) | Peak annual dose (Sv) | Time of peak (years after closure) | Peak annual dose (Sv) | Time of peak (years after closure) |
| $^3$H | $2 \, 10^{-6}$ | 20 | $6 \, 10^{-12}$ | 110 | 0.0 | - |
| $^{14}$C$^{\dagger}$ | $8 \, 10^{-4}$ | 22 | $5 \, 10^{-7}$ | 140 | $3 \, 10^{-8}$ | $3 \, 10^3$ |
| $^{14}$C* | $8 \, 10^{-3}$ | 22 | $5 \, 10^{-6}$ | 140 | $3 \, 10^{-7}$ | $3 \, 10^3$ |
| $^{36}$Cl | $1 \, 10^{-5}$ | 15 | $3 \, 10^{-8}$ | 200 | $6 \, 10^{-9}$ | $6 \, 10^3$ |
| $^{60}$Co | 0 | - | 0 | - | 0 | - |
| $^{+90}$Sr | $1 \, 10^{-15}$ | $1 \, 10^3$ | 0 | - | 0 | - |
| $^{+137}$Cs | 0 | - | 0 | - | 0 | - |
| $^{+226}$Ra | $5 \, 10^{-10}$ | $2 \, 10^4$ | 0 | - | 0 | - |
| $^{232}$Th | $2 \, 10^{-7}$ | $2 \, 10^5$ | $8 \, 10^{-10}$ | $1 \, 10^6$ | 0 | - |
| $^{+238}$U | $6 \, 10^{-8}$ | $6 \, 10^4$ | $2 \, 10^{-10}$ | $6 \, 10^5$ | $1 \, 10^{-15}$ | $1 \, 10^6$ |
| $^{239}$Pu | $1 \, 10^{-7}$ | $2 \, 10^5$ | $2 \, 10^{-10}$ | $5 \, 10^5$ | $1 \, 10^{-15}$ | $1 \, 10^6$ |
| $^{241}$Am | $2 \, 10^{-8}$ | $8 \, 10^3$ | $8 \, 10^{-11}$ | $1 \, 10^5$ | $2 \, 10^{-10}$ | $1 \, 10^6$ |

+ progeny in secular equilibrium included

$^{\dagger}$ 4 MBq te$^{-1}$

*40 MBq te$^{-1}$

Cut-off value for zero is $1 \, 10^{-20}$

**Table 12 Annual doses to construction workers and residents (adults) (Sv) based on waste activity concentration at proposed levels and exposure 30 years after closure of site**

| Radionuclide | Construction workers | Housing residents |
|---|---|---|
| $^3$H | $9 \, 10^{-14}$ | $4 \, 10^{-9}$ |
| $^{14}$C$^{\dagger}$ | $1 \, 10^{-7}$ | $2 \, 10^{-6}$ |
| $^{14}$C* | $1 \, 10^{-6}$ | $2 \, 10^{-5}$ |
| $^{36}$Cl | $6 \, 10^{-7}$ | $2 \, 10^{-4}$ |
| $^{60}$Co | $9 \, 10^{-5}$ | $3 \, 10^{-5}$ |
| $^{+90}$Sr | $6 \, 10^{-6}$ | $6 \, 10^{-5}$ |
| $^{+137}$Cs | $5 \, 10^{-4}$ | $1 \, 10^{-4}$ |
| $^{+226}$Ra | $3 \, 10^{-3}$ | $2 \, 10^{-3}$ |
| $^{232}$Th | $5 \, 10^{-3}$ | $2 \, 10^{-3}$ |
| $^{+238}$U | $7 \, 10^{-5}$ | $2 \, 10^{-5}$ |
| $^{239}$Pu | $5 \, 10^{-4}$ | $2 \, 10^{-5}$ |
| $^{241}$Am | $4 \, 10^{-4}$ | $3 \, 10^{-5}$ |

+ progeny in secular equilibrium included

$^{\dagger}$ 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

Table 13 shows the corresponding tonnage values for the migration scenario. For the inert site, $^{14}$C (at 4 or 40 MBq te$^{-1}$) is the only nuclide that gives a capacity that is more restrictive than the guideline figure of $10^6$ tonnes, representing the mass of waste disposed to the site over its lifetime.  For any one radionuclide the allowable tonnage values increase from the inert site to the

non-hazardous site and to the hazardous site. For the non-hazardous site the disposal tonnage for all radionuclides is less restrictive than the guideline value, apart from $^{14}$C at 40 MBq te$^{-1}$ which is about the same as the guideline value. For the hazardous site all radionuclides have a tonnage which exceeds the guideline value by at least a factor of 100. The short-lived radionuclides $^{36}$Cl and $^{60}$Co are estimated to have zero doses for all three types of site and therefore effectively there is no limit on the capacity for VLLW in all three types of site for these radionuclides.

The annual doses from the intrusion scenarios are shown in Table 12. For the construction workers the annual doses from $^{226}$Ra and $^{232}$Th are equal to and greater than the dose criterion for inadvertent intrusion of 3 mSv y$^{-1}$, respectively. The annual doses from $^{137}$Cs, $^{239}$Pu and $^{241}$Am are a factor of 6 below the dose criterion (at about 0.5 mSv), with other radionuclides giving annual doses below 0.1 mSv. For housing residents, again the $^{226}$Ra and $^{232}$Th annual doses are the highest with doses slightly lower than the dose criterion of 3 mSv y$^{-1}$. All other radionuclides give annual doses below 0.3 mSv y$^{-1}$. The tonnages corresponding to the dose criterion of 3 mSv y$^{-1}$ in Table 13 reflect these estimated annual doses.

**Table 13 Tonnage (te) corresponding to the assumed dose criteria based on peak annual migration doses and annual doses to construction workers and residents (adults) 30 years after closure of the site (assuming 40 MBq te$^{-1}$ for $^3$H and $^{14}$C and 4 MBq te$^{-1}$ for all other radionuclides, including $^{14}$C)**

| Radionuclide | Migration (Inert) | Migration (Non-hazardous) | Migration (Hazardous) | Intrusion (Construction workers) | Intrusion (Housing residents) |
|---|---|---|---|---|---|
| $^3$H | $3\ 10^7$ | $7\ 10^{12}$ | No need for limit | $7\ 10^{16}$ | $1\ 10^{12}$ |
| $^{14}$C† | $6\ 10^4$ | $9\ 10^7$ | $2\ 10^9$ | $5\ 10^{10}$ | $2\ 10^9$ |
| $^{14}$C* | $6\ 10^3$ | $9\ 10^6$ | $2\ 10^8$ | $5\ 10^9$ | $2\ 10^8$ |
| $^{36}$Cl | $3\ 10^6$ | $1\ 10^9$ | $7\ 10^9$ | $1\ 10^{10}$ | $5\ 10^7$ |
| $^{60}$Co | No need for limit | No need for limit** | No need for limit** | $8\ 10^7$ | $2\ 10^8$ |
| $^{+90}$Sr | $3\ 10^{16}$ | No need for limit** | No need for limit** | $1\ 10^9$ | $5\ 10^7$ |
| $^{+137}$Cs | No need for limit** | No need for limit** | No need for limit** | $1\ 10^7$ | $5\ 10^7$ |
| $^{+226}$Ra | $9\ 10^{10}$ | No need for limit | No need for limit** | $2\ 10^6$ | $4\ 10^6$ |
| $^{232}$Th | $2\ 10^8$ | $5\ 10^{10}$ | No need for limit | $1\ 10^6$ | $4\ 10^6$ |
| $^{+238}$U | $7\ 10^8$ | $2\ 10^{11}$ | $4\ 10^{16}$ | $9\ 10^7$ | $3\ 10^8$ |
| $^{239}$Pu | $4\ 10^8$ | $2\ 10^{11}$ | $4\ 10^{16}$ | $1\ 10^7$ | $1\ 10^8$ |
| $^{241}$Am | $2\ 10^9$ | $5\ 10^{11}$ | $2\ 10^{11}$ | $2\ 10^7$ | $1\ 10^8$ |

+ progeny in secular equilibrium included

† 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

** Doses are zero so the quantity is unlimited.

The annual doses to the public residing directly over the landfill site from the radioactive gas pathway, and the corresponding capacity of the site, were also derived and these are given in Table 14. The annual dose from $^{14}$C (at 40 MBq te$^{-1}$) is approximately the same as the dose criterion of 1 mSv y$^{-1}$, whilst the annual dose from $^3$H is orders of magnitude below the criterion. The activity concentration of $^{222}$Rn (ingrown from $^{226}$Ra) was estimated to be 0.3 Bq m$^{-3}$ which is well below the action level of 200 Bq m$^{-3}$. The tonnage corresponding to the assumed dose criteria ("capacity of the site") for $^{14}$C (at 40 MBq te$^{-1}$) is close to the guideline figure of $10^6$ tonnes; if the allowed $^{14}$C concentration in waste is lowered to 4 MBq te$^{-1}$, the capacity of the site increases to ten times the guideline value; for $^3$H and $^{226}$Ra the capacity of the site is three orders of magnitude greater than the guideline figure. Hence a more detailed site-specific assessment would be advisable to determine the amount of waste containing $^{14}$C at 40 Bq te$^{-1}$ that could be disposed of, but would not be necessary for waste containing $^{14}$C at 4 Bq te$^{-1}$, $^3$H and $^{226}$Ra. It should be noted that the calculations do not take account of the effect of a landfill cap on the landfill gas release rate. This is a pessimistic assumption and will result in an over-estimate of the annual doses from landfill gas for non-hazardous and hazardous landfills when the cap is intact.

**Table 14 Annual dose to residents from landfill gas (based on exposure 30 years after disposal and activity concentrations at proposed levels) and tonnage corresponding to assumed criteria (assuming 40 MBq te$^{-1}$ for $^3$H and $^{14}$C, and 4 MBq te$^{-1}$ for $^{226}$Ra and $^{14}$C)**

| Radionuclide | Annual dose (Sv) | Tonnage (tonnes) | Limiting age group |
|---|---|---|---|
| $^3$H | 6 10$^{-7}$ | 4 10$^9$ | Infant |
| $^{14}$C$^\dagger$ | 2 10$^{-4}$ | 1 10$^7$ | Infant |
| $^{14}$C* | 2 10$^{-3}$ | 1 10$^6$ | Infant |
| $^{226}$Ra ($^{222}$Rn) | - | 2 10$^9$ | Tonnage not dependent on age group |

$^\dagger$ 4 MBq te$^{-1}$

* 40 MBq te$^{-1}$

### 8.1.4 Summary of base case results

The key points from the base case calculations based on the proposed definition of VLLW are summarized below.

For the landfill workers, assuming a disposal rate of 10$^5$ te y$^{-1}$, the radionuclides giving annual doses above a dose criterion of 1 mSv y$^{-1}$ are $^{60}$Co, $^{137}$Cs and $^{226}$Ra, and the main exposure pathway is external irradiation from the waste. For $^{232}$Th, $^{239}$Pu and $^{241}$Am the annual doses are of the order of 1 mSv in a year, with the exposure pathway being inhalation of contaminated material. However, these annual doses could be reduced to below the dose criterion by requiring that the waste is covered by 30 cm of topsoil. Based on the calculations presented here and the dose criterion adopted for this study it is indicated that a more detailed site-specific assessment would be advisable to determine the quantity of these five radionuclides that could be disposed of. For the other radionuclides, disposal of a mass of VLLW of approximately 10$^5$ te y$^{-1}$ would meet the assumed dose criterion.

For the leachate discharge scenario the annual dose from disposal of 10$^6$ tonnes of waste containing $^{14}$C (at 4 or 40 MBq te$^{-1}$) is estimated to exceed the assumed dose criterion of 300 µSv y$^{-1}$, for all three types of site. These annual doses could be reduced to meet the criterion by choosing a site where the discharge occurs to a larger river. For the inert site, the annual dose from $^{60}$Co is also estimated to exceed the dose criterion. All other annual doses are estimated to be less than 300 µSv y$^{-1}$. However, it is expected that very little leachate could in fact be collected from an inert site (see Table 10) and therefore that the annual doses from disposal of the guideline value of 10$^5$ te y$^{-1}$ of waste to an inert site would be below 300 µSv y$^{-1}$ for all radionuclides. Nevertheless, based on the calculations presented here and the dose criterion adopted for this study, the work indicates that a more detailed site-specific assessment would be advisable to determine the quantity of VLLW waste containing $^{14}$C (at 4 or 40 MBq te$^{-1}$) that could be disposed of to a particular site.

The estimated peak annual doses from migration following disposal of 10$^6$ tonnes of VLLW containing $^{14}$C (at 4 or 40 MBq te$^{-1}$) in an inert site exceed the dose criterion of 20 µSv in a year and the peak annual dose from $^{36}$Cl is

close to the criterion. The corresponding peak annual dose for $^{14}$C (at 40 MBq te$^{-1}$) for the non-hazardous site is below 20 μSv. However, if the $^{14}$C concentration is limited to 4 MBq te$^{-1}$, the peak annual dose will be less than 1 μSv. For all other radionuclides and landfill types the estimated peak annual doses from disposal of $10^6$ tonnes of VLLW are "trivial". This work indicates that a more detailed site-specific assessment would be advisable to determine the amount of VLLW waste containing $^{14}$C (at 4 or 40 MBq te$^{-1}$) and/or $^{36}$Cl that could be disposed of to a particular inert site. For the other radionuclides and landfill types, based on the calculations presented here and the assumed dose criterion, disposal of a mass of approximately $10^5$ te y$^{-1}$ (or a total of about $10^6$ tonnes) would meet the dose criteria.

For the inadvertent intrusion scenarios $^{226}$Ra and $^{232}$Th give estimated annual doses equal to or greater than the dose criterion of 3 mSv y$^{-1}$ for both construction workers and housing residents. This work indicates that a more detailed site-specific assessment would be advisable to determine the disposable quantity of VLLW waste containing these two radionuclides. For the other radionuclides, based on the calculations presented here and the assumed dose criterion, disposal of a mass of approximately $10^5$ te y$^{-1}$ (or a total of about $10^6$ tonnes) would meet the assumed dose criteria.

For the landfill gas scenario the estimated annual dose to the public residing on the site from disposal of $10^6$ tonnes of VLLW containing $^{14}$C (at 40 MBq te$^{-1}$) exceeds the assumed dose criterion of 1 mSv y$^{-1}$, whilst the corresponding annual dose from $^3$H and the activity concentration of $^{222}$Rn (ingrown from $^{226}$Ra) are orders of magnitude below the criteria. Disposal of a total of $10^6$ tonnes of VLLW containing $^{14}$C at 4 MBq te$^{-1}$ would meet the dose criterion of 1 mSv y$^{-1}$. Based on the dose criterion adopted for this study the calculations indicate that a more detailed site-specific assessment would therefore be advisable to determine the disposable quantity of VLLW waste containing $^{14}$C at 40 MBq te$^{-1}$. For the other radionuclides, again based on the calculations presented here and the assumed dose criteria, disposal of a mass of approximately $10^5$ te y$^{-1}$ (or a total of about $10^6$ tonnes) would meet the assumed dose criteria.

## 8.2   Selected waste streams

### 8.2.1  Results

The four example waste stream types selected from the Atkins Limited data (Atkins Limited, 2006) are those containing mixed waste and building materials that are likely to arise from decommissioning.

It should be remembered that the methodologies used for calculating dose in this study assume a uniform waste type in which the radionuclides are available for dissolution in water and subsequently migrate in the environment. For the mixed types of waste selected for the waste stream calculations, the activity may be bound or contained in a material which may not be soluble or may not readily breakdown into gaseous form, e.g. internally contaminated steel or

plastic, and so may not be available for transport. In such a case the doses calculated using the current methodology would be overestimates.

Table 15 gives the estimated annual doses to landfill workers based on the example waste streams. The results presented are calculated assuming a disposal rate of approximately $10^5$ te $y^{-1}$ and do not reflect the total waste volume given in (Defra, 2005a) and Table 3. For the waste streams with small waste volumes, ie the Harwell waste stream with 247 m$^3$ and the Hartlepool waste stream with 78 m$^3$, this means that the annual doses given in this report are a significant overestimate of the doses from these particular waste streams as the workers would not be exposed to these wastes for an entire year.

**Table 15 Annual doses to landfill workers (Sv) for selected waste stream types***

| Radionuclide | Harwell | Sellafield | Torness | Hartlepool |
|---|---|---|---|---|
| $^3$H | 0 | $1\ 10^{-13}$ | $3\ 10^{-11}$ | $4\ 10^{-10}$ |
| $^{14}$C | 0 | $3\ 10^{-11}$ | $6\ 10^{-14}$ | $2\ 10^{-9}$ |
| $^{36}$Cl | 0 | 0 | $2\ 10^{-11}$ | 0 |
| $^{60}$Co | 0 | $1\ 10^{-6}$ | $4\ 10^{-8}$ | $5\ 10^{-4}$ |
| $^{+90}$Sr | $1\ 10^{-5}$ | $2\ 10^{-6}$ | 0 | $4\ 10^{-8}$ |
| $^{+137}$Cs | $2\ 10^{-3}$ | $5\ 10^{-4}$ | 0 | $4\ 10^{-6}$ |
| $^{+226}$Ra | 0 | 0 | 0 | 0 |
| $^{232}$Th | 0 | 0 | 0 | 0 |
| $^{+238}$U | $2\ 10^{-17}$ | $5\ 10^{-9}$ | 0 | 0 |
| $^{239}$Pu | $2\ 10^{-5}$ | $8\ 10^{-6}$ | 0 | 0 |
| $^{241}$Am | $1\ 10^{-6}$ | $6\ 10^{-7}$ | 0 | 0 |
| Total | $2\ 10^{-3}$ | $5\ 10^{-4}$ | $4\ 10^{-8}$ | $6\ 10^{-4}$ |

* Assuming that $10^5$ te $y^{-1}$ are disposed of

+ progeny in secular equilibrium included

It can be seen that the estimated annual doses to the landfill workers are below the assumed criterion of 1 mSv $y^{-1}$ except for the type of waste associated with the Harwell labs. For this type of waste the annual dose was estimated to be 2 mSv with the majority of the dose being due to external irradiation from $^{137}$Cs. From Table 3 it can be seen that $^{137}$Cs is the radionuclide with the highest concentration for this waste stream so it not surprising that it is the most important radionuclide. However, it should be remembered that this is the estimated annual dose assuming that there is sufficient mass of this waste for a worker to be exposed for an entire year, which is not likely for the actual quantity of this particular waste stream. For the Hartlepool waste stream $^{60}$Co and $^{137}$Cs are the dominant radionuclides contributing to annual dose as they have a higher dose per unit activity than $^3$H which makes up around 70% of the activity concentration of the waste.

Table 16 shows the annual doses due to direct discharge of leachate to a river for the four waste streams for each of the three types of landfill. The total annual dose from the Hartlepool waste stream type for an inert site is approaching the dose criterion of 300 µSv $y^{-1}$ due to the presence of $^{14}$C and $^{60}$Co in the waste stream. The total annual dose for Hartlepool waste stream type decreases from the inert landfill value to the non-hazardous and hazardous landfill types respectively and is dominated by the $^{14}$C dose even though it is only 7% of the

total activity concentration. The contribution from $^{60}$Co to the total annual dose from the Hartlepool waste stream type has fallen for the non-hazardous and hazardous sites as shown in Table 8. Hence large quantities of this type of waste may require a more detailed site specific assessment and consideration of a requirement to discharge to a large river.

For all other waste types and landfill types the annual doses can be considered to be "trivial". For other waste types in general, if a radionuclide is dominant in the total activity concentration it is dominant in the total annual dose contribution for the waste stream as a whole. For example, $^{137}$Cs in the Sellafield type waste stream and Harwell type waste streams (see Table 3 and Table 16).

Table 17 shows the peak annual migration doses for the inert, non-hazardous and hazardous sites for the four example VLLW streams. It is noted that for all four different waste streams for all three types of site there is only one case where the peak annual dose is equal to the assumed criterion of 20 μSv y$^{-1}$: the Hartlepool waste stream type disposed of in an inert site. The peak annual dose for this waste stream is dominated by the contribution from $^{14}$C. Apart from the Hartlepool waste stream type in an inert landfill, Table 17 shows that for all other waste streams for all three types of landfill, the peak annual doses are less than 1 μSv, considerably lower than the level normally considered as "trivial".

It should be noted that the timing of the peak annual dose will vary depending on the radionuclide (see Table 11 for peak times for the different radionuclides) and the summation of peak annual doses is a conservative calculation. For example a member of the public who receives the highest doses from $^{3}$H, $^{14}$C and $^{36}$Cl (these radionuclides are estimated to deliver their peak doses at around 20 years) will not be the same person who receives the highest doses from for example $^{226}$Ra where the peak annual dose is estimated to be received at 20,000 years.

**Table 16 Annual doses due to discharge of leachate to river (Sv) from selected waste streams***

| Radionuclide | Inert | | | | Non-hazardous | | | | Hazardous | | | |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
| | Harwell | Sellafield | Torness | Hartlepool | Harwell | Sellafield | Torness | Hartlepool | Harwell | Sellafield | Torness | Hartlepool |
| $^3$H | 0 | $8\ 10^{-11}$ | $2\ 10^{-8}$ | $2\ 10^{-7}$ | 0 | $5\ 10^{-12}$ | $1\ 10^{-9}$ | $1\ 10^{-8}$ | 0 | $6\ 10^{-17}$ | $2\ 10^{-14}$ | $2\ 10^{-13}$ |
| $^{14}$C | 0 | $1\ 10^{-6}$ | $3\ 10^{-9}$ | $8\ 10^{-5}$ | 0 | $1\ 10^{-6}$ | $2\ 10^{-9}$ | $8\ 10^{-5}$ | 0 | $1\ 10^{-6}$ | $2\ 10^{-9}$ | $7\ 10^{-5}$ |
| $^{36}$Cl | 0 | 0 | $2\ 10^{-9}$ | 0 | 0 | 0 | $2\ 10^{-9}$ | 0 | 0 | 0 | $2\ 10^{-9}$ | 0 |
| $^{60}$Co | 0 | $2\ 10^{-7}$ | $6\ 10^{-9}$ | $9\ 10^{-5}$ | 0 | $3\ 10^{-10}$ | $8\ 10^{-12}$ | $1\ 10^{-7}$ | 0 | $1\ 10^{-21}$ | $3\ 10^{-23}$ | $5\ 10^{-19}$ |
| $^{+90}$Sr | $5\ 10^{-6}$ | $1\ 10^{-6}$ | 0 | $2\ 10^{-8}$ | $2\ 10^{-6}$ | $3\ 10^{-7}$ | 0 | $6\ 10^{-9}$ | $1\ 10^{-8}$ | $3\ 10^{-9}$ | 0 | $5\ 10^{-11}$ |
| $^{+137}$Cs | $2\ 10^{-5}$ | $6\ 10^{-6}$ | 0 | $4\ 10^{-8}$ | $5\ 10^{-6}$ | $2\ 10^{-6}$ | 0 | $1\ 10^{-8}$ | $5\ 10^{-8}$ | $2\ 10^{-8}$ | 0 | $1\ 10^{-10}$ |
| $^{+226}$Ra | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{232}$Th | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{+238}$U | $6\ 10^{-19}$ | $2\ 10^{-10}$ | 0 | 0 | $6\ 10^{-19}$ | $2\ 10^{-10}$ | 0 | 0 | $6\ 10^{-19}$ | $2\ 10^{-10}$ | 0 | 0 |
| $^{239}$Pu | $2\ 10^{-8}$ | $1\ 10^{-8}$ | 0 | 0 | $2\ 10^{-8}$ | $1\ 10^{-8}$ | 0 | 0 | $2\ 10^{-8}$ | $1\ 10^{-8}$ | 0 | 0 |
| $^{241}$Am | $2\ 10^{-9}$ | $1\ 10^{-9}$ | 0 | 0 | $2\ 10^{-9}$ | $1\ 10^{-9}$ | 0 | 0 | $2\ 10^{-9}$ | $7\ 10^{-10}$ | 0 | 0 |
| Total | $2\ 10^{-5}$ | $8\ 10^{-6}$ | $3\ 10^{-8}$ | $2\ 10^{-4}$ | $7\ 10^{-6}$ | $3\ 10^{-6}$ | $6\ 10^{-9}$ | $8\ 10^{-5}$ | $9\ 10^{-8}$ | $1\ 10^{-6}$ | $5\ 10^{-9}$ | $7\ 10^{-5}$ |

* assuming $10^6$ te disposed of

+ progeny in secular equilibrium included

**Table 17 Peak annual doses to adults due to migration (Sv) from selected waste streams***

| Radionuclide | Inert | | | | Non-hazardous | | | | Hazardous | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Harwell | Sellafield | Torness | Hartlepool | Harwell | Sellafield | Torness | Hartlepool | Harwell | Sellafield | Torness | Hartlepool |
| $^{3}$H | 0 | $2\ 10^{-11}$ | $4\ 10^{-9}$ | $4\ 10^{-8}$ | 0 | $7\ 10^{-17}$ | $2\ 10^{-14}$ | $2\ 10^{-13}$ | 0 | 0 | 0 | 0 |
| $^{14}$C | 0 | $3\ 10^{-7}$ | $7\ 10^{-10}$ | $2\ 10^{-5}$ | 0 | $2\ 10^{-10}$ | $4\ 10^{-13}$ | $1\ 10^{-8}$ | 0 | $1\ 10^{-11}$ | $2\ 10^{-14}$ | $8\ 10^{-10}$ |
| $^{36}$Cl | 0 | 0 | $5\ 10^{-10}$ | 0 | 0 | 0 | $1\ 10^{-12}$ | 0 | 0 | 0 | $2\ 10^{-13}$ | 0 |
| $^{60}$Co | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{+90}$Sr | $4\ 10^{-16}$ | $8\ 10^{-17}$ | 0 | $2\ 10^{-18}$ | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{+137}$Cs | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{+226}$Ra | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{232}$Th | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{+238}$U | $3\ 10^{-20}$ | $8\ 10^{-12}$ | 0 | 0 | 0 | $3\ 10^{-14}$ | 0 | 0 | 0 | $1\ 10^{-19}$ | 0 | 0 |
| $^{239}$Pu | $3\ 10^{-9}$ | $1\ 10^{-9}$ | 0 | 0 | $5\ 10^{-12}$ | $2\ 10^{-12}$ | 0 | 0 | $3\ 10^{-17}$ | $1\ 10^{-17}$ | 0 | 0 |
| $^{241}$Am | $5\ 10^{-11}$ | $2\ 10^{-11}$ | 0 | 0 | $2\ 10^{-13}$ | $9\ 10^{-14}$ | 0 | 0 | $6\ 10^{-13}$ | $3\ 10^{-13}$ | 0 | 0 |
| Total | $3\ 10^{-9}$ | $3\ 10^{-7}$ | $5\ 10^{-9}$ | $2\ 10^{-5}$ | $5\ 10^{-12}$ | $2\ 10^{-10}$ | $1\ 10^{-12}$ | $1\ 10^{-8}$ | $6\ 10^{-13}$ | $1\ 10^{-11}$ | $2\ 10^{-13}$ | $8\ 10^{-10}$ |

* assuming $10^{6}$ te disposed of

+ progeny in secular equilibrium included

Table 18 gives the estimated annual doses from intrusion into the landfill site after 30 years for construction workers and housing residents based on the inventory for the selected waste streams given in Table 3. The annual doses are all below the assumed intrusion criterion of 3 mSv $y^{-1}$. The highest annual doses were estimated to be for the Harwell waste stream type, at 0.6 mSv in a year and 0.2 mSv in a year for construction workers and housing residents respectively. The majority of the annual dose for both groups was estimated to be from $^{137}$Cs which represents around 75% of the activity concentration of the total waste.

**Table 18 Annual doses from intrusion (Sv) for selected waste streams***

| Radio-nuclide | Construction workers | | | | Housing residents | | | |
|---|---|---|---|---|---|---|---|---|
| | Harwell | Sellafield | Torness | Hartlepool | Harwell | Sellafield | Torness | Hartlepool |
| $^{3}$H | 0 | $1\ 10^{-18}$ | $2\ 10^{-16}$ | $2\ 10^{-15}$ | 0 | $7\ 10^{-14}$ | $2\ 10^{-11}$ | $2\ 10^{-10}$ |
| $^{14}$C | 0 | $5\ 10^{-11}$ | $1\ 10^{-13}$ | $3\ 10^{-9}$ | 0 | $1\ 10^{-9}$ | $3\ 10^{-12}$ | $9\ 10^{-8}$ |
| $^{36}$Cl | 0 | 0 | $2\ 10^{-11}$ | 0 | 0 | 0 | $4\ 10^{-9}$ | 0 |
| $^{60}$Co | 0 | $1\ 10^{-8}$ | $5\ 10^{-10}$ | $8\ 10^{-6}$ | 0 | $4\ 10^{-9}$ | $2\ 10^{-10}$ | $3\ 10^{-6}$ |
| $^{+90}$Sr | $2\ 10^{-6}$ | $4\ 10^{-7}$ | 0 | $7\ 10^{-9}$ | $4\ 10^{-5}$ | $8\ 10^{-6}$ | 0 | $2\ 10^{-7}$ |
| $^{+137}$Cs | $5\ 10^{-4}$ | $2\ 10^{-4}$ | 0 | $1\ 10^{-6}$ | $1\ 10^{-4}$ | $5\ 10^{-5}$ | 0 | $4\ 10^{-7}$ |
| $^{+226}$Ra | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{232}$Th | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| $^{+238}$U | $1\ 10^{-15}$ | $4\ 10^{-7}$ | 0 | 0 | $4\ 10^{-16}$ | $1\ 10^{-7}$ | 0 | 0 |
| $^{239}$Pu | $1\ 10^{-4}$ | $6\ 10^{-5}$ | 0 | 0 | $4\ 10^{-5}$ | $2\ 10^{-5}$ | 0 | 0 |
| $^{241}$Am | $2\ 10^{-7}$ | $8\ 10^{-8}$ | 0 | 0 | $6\ 10^{-8}$ | $3\ 10^{-8}$ | 0 | 0 |
| Total | $6\ 10^{-4}$ | $2\ 10^{-4}$ | $5\ 10^{-10}$ | $1\ 10^{-5}$ | $2\ 10^{-4}$ | $8\ 10^{-5}$ | $4\ 10^{-9}$ | $3\ 10^{-6}$ |

* assuming $10^{6}$ te disposed of

+ progeny in secular equilibrium included

Table 19 gives the annual doses from landfill gas from the disposal of four waste streams. All of the annual doses are lower than the assumed dose criterion of 1 mSv $y^{-1}$. There is no dose from $^{222}$Rn as none of the waste streams contain $^{226}$Ra. The Harwell type waste stream does not contain $^{3}$H or $^{14}$C hence the annual doses are zero.

**Table 19 Annual doses from landfill gas (Sv) for selected waste streams***

| Radionuclide | Harwell | Sellafield | Torness | Hartlepool |
|---|---|---|---|---|
| $^{3}$H | 0 | $6\ 10^{-12}$ | $1\ 10^{-9}$ | $1\ 10^{-8}$ |
| $^{14}$C | 0 | $7\ 10^{-8}$ | $1\ 10^{-10}$ | $4\ 10^{-6}$ |

* assuming $10^{6}$ te disposed of

### 8.2.2 Summary of results for different waste streams

Four example waste streams have been chosen. Three of these waste streams gave estimated annual doses below the relevant assumed criteria for all

scenarios, assuming that approximately $10^6$ tonnes of waste with this radionuclide mix are disposed of in a landfill site. The annual dose to workers from the fourth waste stream was estimated to exceed the dose criterion adopted for this study due to the high $^{137}$Cs content of the waste. However, the annual doses from this waste stream for all the other scenarios met the assumed dose criteria. Hence a more detailed site-specific assessment would be advisable for this type of waste to determine the quantity of VLLW waste that could be disposed of to a particular site if the criteria assumed in this study were adopted. Nevertheless, taking into account the available quantities of these waste streams, this study shows that disposal to landfill of each of the four example waste streams would meet the dose criteria adopted for this study.

## 8.3 Sensitivity results

### 8.3.1 Workers

The annual doses to workers will be reduced if additional clean soil is added to cover the waste. For example for $^{226}$Ra, by adding 30 cm of clean top soil the dose rate is reduced by a factor of 20 whilst adding 50 cm would reduce it by a factor of 100. (However, reducing the area of exposed waste would have a small effect unless the area was less than a few metres wide).

### 8.3.2 Discharge of leachate to river

The 'base case' calculations assume that the collected leachate is discharged into a very small river with a volumetric flow rate of 1 $m^3$ $s^{-1}$. A sensitivity study on the river size found that for a small increase in river size, up to a volumetric flow of 2.5 $m^3$ $s^{-1}$, the dose is inversely proportional to the volumetric flow. However, for a large increase in river size, in which the volume flow rate is increased to 500 $m^3$ $s^{-1}$, the corresponding dose did decrease but not in inverse proportion to the river size; the dose was about 100 times smaller than the dose from the very small river (see Table 20). This is due to the addition of extra exposure pathways when the river size increases eg abstraction of water for drinking and extensive irrigation.

**Table 20 Dose rates to the public (Sv y$^{-1}$ per Bq s$^{-1}$) from the discharge of leachate to river for different river sizes and the ratio of the doses**

| | River size | | | Ratio of dose (very small river/small river) | Ratio of dose (very small river/large river |
|---|---|---|---|---|---|
| | Very small river | Small river | Large river | | |
| Volumetric flow rate ($m^3$ $s^{-1}$) | 1.0 | 2.5 | 500.0 | | |
| Radionuclide | | | | | |
| $^3$H | 1.3 10$^{-11}$ | 5.7 10$^{-12}$ | 6.0 10$^{-14}$ | 2.3 | 217 |
| $^{14}$C | 5.0 10$^{-8}$ | 2.0 10$^{-8}$ | 3.8 10$^{-11}$ | 2.5 | 1314 |
| +$^{137}$Cs | 5.0 10$^{-7}$ | 2.1 10$^{-7}$ | 1.9 10$^{-9}$ | 2.4 | 265 |
| $^{238}$U | 4.0 10$^{-8}$ | 1.7 10$^{-8}$ | 3.5 10$^{-10}$ | 2.4 | 113 |

+ progeny in secular equilibrium included

### 8.3.3        Migration scenarios

The peak annual doses resulting from migration of radioactivity for the inert site for the three selected radionuclides ($^{14}$C, $^{137}$Cs and $^{238}$U) and for the sensitivity scenarios are shown in Table 21. The base case migration peak annual doses for the inert site are also shown for comparison.

It can be seen that setting the waste distribution coefficient $K_d$ to zero has no effect on the $^{14}$C peak annual dose as expected (since it has a zero $K_d$ value), and increases the $^{238}$U peak annual dose by a factor of about two. However, $^{137}$Cs still decays before it reaches the biosphere.

For the case when there is no unsaturated zone the peak annual doses from the long-lived radionuclides increase by about an order of magnitude and a non zero peak annual dose is estimated for the short-lived radionuclide $^{137}$Cs. This is because the waste is released and dispersed into the aquifer more quickly and readily, so that even some $^{137}$Cs emerges from the site.

Reducing the aquifer length and aquifer $K_d$ has little effect on peak annual doses or capacity of the site compared to the base case site values. The siting of the landfill at a coastal location rather than an inland one reduces the peak annual doses and increases the capacity of the site due to the fact that the activity leached from the site is diluted by the volume of the local coastal water compartment.

**Table 21 Range of peak annual doses to adults (Sv per MBq disposed of) due to migration from an inert site**

| Radionuclide | Base case | Waste $K_d$ | No unsaturated zone | Aquifer length | Aquifer $K_d$ | Release in to coastal area |
|---|---|---|---|---|---|---|
| $^{14}$C | 9 10$^{-11}$ | 9 10$^{-11}$ | 8 10$^{-10}$ | 9 10$^{-11}$ | 9 10$^{-11}$ | 2 10$^{-12}$ |
| $^{+137}$Cs | - | - | 2 10$^{-15}$ | - | - | - |
| $^{238}$U | 7 10$^{-15}$ | 2 10$^{-14}$ | 7 10$^{-14}$ | 7 10$^{-15}$ | 7 10$^{-15}$ | 4 10$^{-18}$ |

+ progeny in secular equilibrium included

### 8.3.4        Bathtubbing scenario

The annual doses for the bathtubbing scenario are for the first year after the event has been assumed to occur.  The annual doses will continue in subsequent years but will be lower than the first year doses which are presented in Table 22.

For two of the three radionuclides ($^{137}$Cs and $^{238}$U) considered the annual doses are higher than those predicted from the migration of groundwater.

**Table 22 Annual dose to members of the public for bathtubbing scenario (Sv per MBq disposed of)**

| Radionuclide | Bathtubbing |
|---|---|
| $^{14}$C | 8 $10^{-12}$ |
| $^{+137}$Cs | 1 $10^{-13}$ |
| $^{238}$U | 2 $10^{-14}$ |

$^{+}$ progeny in secular equilibrium included

### 8.3.5      Intrusion

Annual doses from intrusion at three different times (10, 30 and 100 years) are shown in Table 23. For the relatively short-lived radionuclides, ie half-lives less than or equal to 30 years ($^{3}$H, $^{60}$Co, $^{90}$Sr and $^{137}$Cs), changing the assumed time of intrusion from 30 years to 10 and 100 years changes the annual doses received by construction workers or housing residents. If intrusion takes place after only 10 years then the annual dose from $^{60}$Co increases to 1 $10^{-10}$ Sv per MBq disposed of whereas it decreases to 1 $10^{-15}$ Sv per MBq disposed of if intrusion occurs after 100 years.  For the long-lived radionuclides ($^{14}$C, $^{36}$Cl, $^{226}$Ra, $^{226}$Ra, $^{238}$U and $^{239}$Pu), varying the intrusion time from 10 to 100 years does not significantly effect the estimated annual doses.

### 8.3.6      Landfill gas

Varying the time at which the public is assumed to reside on the landfill site following closure from 30 years to 10 years and 100 years unsurprisingly influences the annual doses from the radionuclide with the shortest half-life ie $^{3}$H with a half-life of 12.3 years (Table 24).  For $^{14}$C (half-life of 5730 years) and $^{222}$Rn, which is the progeny of the $^{226}$Ra (half-life of 1600 years) varying the time at which exposure to landfill gas starts (to 10 or 100 years) has no effect on the annual dose.

(Crawford et al, 2005) carried out sensitivity studies for two parameters that affect doses from gas release: the thickness of the cap, and the time spent indoors or outdoors which affects the amount inhaled. They found that the annual dose from $^{226}$Ra increased by a factor of 10 if the thickness of the cover was reduced from 1.5m to 1m, but that it was only halved if the cover thickness was increased to 3m. They found no effect of the thickness of the cover on calculated annual doses from the other radioactive gases ($^{3}$H and $^{14}$C) because their assessment methodology conservatively neglected the effect of the cap in mitigating gas migration.

**Table 23 Range of annual doses to construction workers and housing residents (adults) (Sv received in a year per MBq in the landfill site)**

| Time of intrusion (y) | $^3$H | $^{14}$C | $^{36}$Cl | $^{60}$Co | $^{90}$Sr+ | $^{137}$Cs+ | $^{226}$Ra+ | $^{232}$Th | $^{238}$U+ | $^{239}$Pu | $^{241}$Am |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Construction worker doses | | | | | | | | | | | |
| 10 | $3\ 10^{-21}$ | $1\ 10^{-14}$ | $7\ 10^{-14}$ | $1\ 10^{-10}$ | $1\ 10^{-12}$ | $9\ 10^{-11}$ | $4\ 10^{-10}$ | $4\ 10^{-10}$ | $8\ 10^{-12}$ | $5\ 10^{-11}$ | $5\ 10^{-11}$ |
| 30 | $1\ 10^{-21}$ | $1\ 10^{-14}$ | $7\ 10^{-14}$ | $1\ 10^{-11}$ | $7\ 10^{-13}$ | $6\ 10^{-11}$ | $4\ 10^{-10}$ | $6\ 10^{-10}$ | $8\ 10^{-12}$ | $5\ 10^{-11}$ | $4\ 10^{-11}$ |
| 100 | $2\ 10^{-23}$ | $1\ 10^{-14}$ | $7\ 10^{-14}$ | $1\ 10^{-15}$ | $1\ 10^{-13}$ | $1\ 10^{-11}$ | $3\ 10^{-10}$ | $6\ 10^{-10}$ | $8\ 10^{-12}$ | $5\ 10^{-11}$ | $4\ 10^{-11}$ |
| Adult resident doses | | | | | | | | | | | |
| 10 | $2\ 10^{-16}$ | $4\ 10^{-13}$ | $2\ 10^{-11}$ | $5\ 10^{-11}$ | $2\ 10^{-11}$ | $3\ 10^{-11}$ | $2\ 10^{-10}$ | $1\ 10^{-10}$ | $3\ 10^{-12}$ | $6\ 10^{-12}$ | $6\ 10^{-12}$ |
| 30 | $7\ 10^{-17}$ | $4\ 10^{-13}$ | $2\ 10^{-11}$ | $3\ 10^{-12}$ | $1\ 10^{-11}$ | $2\ 10^{-11}$ | $2\ 10^{-10}$ | $2\ 10^{-10}$ | $3\ 10^{-12}$ | $6\ 10^{-12}$ | $6\ 10^{-12}$ |
| 100 | $1\ 10^{-18}$ | $4\ 10^{-13}$ | $2\ 10^{-11}$ | $3\ 10^{-16}$ | $3\ 10^{-12}$ | $3\ 10^{-12}$ | $2\ 10^{-10}$ | $2\ 10^{-10}$ | $3\ 10^{-12}$ | $6\ 10^{-12}$ | $5\ 10^{-12}$ |

+ progeny in secular equilibrium included

**Table 24 Range of annual doses or activity concentrations for landfill gas for members of the public residing on the site**

| Time after closure | $^3$H Adult dose (Sv received in a year per MBq disposed of) | $^{14}$C Adult dose (Sv received in a year per MBq disposed of) | $^{222}$Rn Activity concentration (Bq m$^{-3}$ per MBq disposed of) |
|---|---|---|---|
| 10 year | $2\ 10^{-14}$ | $2\ 10^{-11}$ | $3\ 10^{-8}$ |
| 30 year | $6\ 10^{-15}$ | $2\ 10^{-11}$ | $3\ 10^{-8}$ |
| 100 year | $1\ 10^{-16}$ | $2\ 10^{-11}$ | $3\ 10^{-8}$ |

# 9    DISCUSSION AND CONCLUSIONS

This report gives estimates of the annual doses to landfill operators and members of the public arising from the disposal of large quantities of very low level solid radioactive waste, as defined in the proposed definition (see Section 1.1), to conventional landfill site types. Three types of landfill were considered: inert, non-hazardous and hazardous. The exposure scenarios considered were exposure of workers, leachate discharge to a very small river, migration with groundwater, gaseous release and inadvertent intrusion.

Eleven radionuclides and four example waste streams were selected based on the information supplied by Atkins Limited on wastes that could meet this proposed definition. These waste streams comprise concrete rubble and mixed waste. 'Base case' calculations were performed for each radionuclide and example waste stream. A series of sensitivity analyses were also performed to investigate the possible range of annual doses.

The calculations were cautious, containing conservative assumptions to overestimate the annual doses. The doses were calculated assuming that a typical landfill site would receive approximately $1.4 \ 10^5$ te $y^{-1}$ of waste for 15 years, giving a total disposed quantity of approximately $2 \ 10^6$ tonnes. These results were then used to calculate the quantity of waste containing each radionuclide that could be disposed of at a site and meet the assumed dose criteria. Guideline figures of $10^5$ te $y^{-1}$ for the annual disposal rate and $10^6$ tonnes for the total capacity of the site were used to evaluate the results and draw conclusions. If the quantity that could be disposed of was greater than these values then this indicates that a typical site would meet the dose criteria even when full of VLLW.

## 9.1    Conclusions for each nuclide

For $^3$H, using the proposed definition of VLLW of 40 MBq te$^{-1}$ and assuming that the site is filled with about $10^6$ tonnes of this waste, the estimated doses for all scenarios were below the dose criteria chosen for this project, for all three types of site.  For all the scenarios, except for the inhalation of gas released from the landfill, the annual doses were estimated to be at a level considered to be "trivial", ie of the order few tens of µSv per year (IAEA, 1988).  Hence it is unlikely that limits on the allowable disposal rate would be required to meet the dose criteria adopted for this study.  The results indicate that the total capacity for this waste in an inert landfill site would be around $10^7$ tonnes.  However, for other site types, there would be no need for limits based on the calculations presented here and the dose criteria adopted for this study.

For $^{14}$C, using the proposed definition of 40 MBq te$^{-1}$, the peak annual doses from three scenarios exceeded the dose criteria adopted here: migration of groundwater (inert type landfill only), discharge of extracted leachate to a very small river, and inhalation of gas released from the site. The assumptions in the calculations may need to be re-examined before suggesting limits on the

quantities of waste that could be disposed of based on the calculations presented here and the dose criteria adopted for this study. However, if these were determined to be reasonable for their purpose there may be a case for using 4 MBq te[-1] as the definition of VLLW for [14]C. Nevertheless, even if the definition of VLLW for ^{14}C was lowered to 4 MBq te[-1], the peak annual dose from groundwater migration from the inert type landfill site and from discharge of leachate to a very small river would still exceed the assumed criteria, albeit by a smaller margin. This implies that non-hazardous and hazardous sites could meet the dose criteria if the leachate extracted from the landfill is discharged to a larger river as defined in Table B8. However, based on the calculations presented here, the inert type of landfill sites would not be suitable for large quantities of VLLW containing ^{14}C unless the groundwater migrates to the sea thereby allowing greater dilution, or more relaxed dose criteria were adopted. However consideration would have to be given to the UK's commitment to OSPAR Commission for the Protection of the Marine Environment of the NorthEast Atlantic (OSPAR). The UK is a contracting party to the OSPAR (OSPAR, 1998) whose ultimate aim is to achieve concentrations equivalent to near background values for naturally-occurring substances and close to zero for artificial radioactive substances in the marine environment. In summary, a more detailed site-specific assessment would be required to determine the quantity of VLLW containing [14]C (at 4 or 40 MBq te[-1]) that could be disposed of at a particular landfill site, based on the calculations presented here and the dose criteria adopted for this study.

For [36]Cl, using a definition of 4 MBq te[-1] for VLLW, all the annual doses from disposal of $10^6$ tonnes are below the dose criteria used for this study. The capacity for an inert site may be limited to a tonnage of 3 $10^6$ tonnes based on the peak annual doses from the migration of groundwater and the criteria adopted in this study.

For ^{60}Co the estimated annual doses to landfill workers exceed the dose criterion chosen for this study.  The annual dose to the public if leachate was extracted from an inert landfill site and discharged to a very small river was also estimated to exceed the adopted criterion.  Potential methods for reducing these annual doses and thereby complying with the dose criteria would be by ensuring that the waste was covered more frequently; by putting down more covering material to reduce the annual  doses to the workers; and requiring that the leachate is discharged to a river with a larger flow rate.  However it should be noted that the calculated annual doses to the landfill workers and particularly to the members of the public from the discharge of leachate to a very small river are based on a number of conservative assumptions.  Therefore, based on the calculations presented here and the dose criteria adopted for this study, it would be necessary to do a more detailed site-specific assessment of annual doses before placing any restrictions on the quantity to be disposed of or on the discharge of leachate from a particular site.

For ^{90}Sr, using a definition of 4 MBq te[-1] for VLLW, all the annual doses from disposal of about $10^6$ tonnes are below the dose criteria used for this study.

For ^{137}Cs the annual dose to a landfill worker, assuming that $10^5$ te y[-1] of VLLW waste is disposed of containing 4 MBq te[-1], is estimated to be 1 mSv in a year ie

the value of the dose criterion adopted. Therefore it may be necessary to review the pessimistic assumptions made in the calculations. If they are determined to be satisfactory a more detailed site-specific assessment would be needed to determine if there is a need for measures to ensure that the annual doses to the landfill workers are optimised eg additional covering of the waste. Otherwise, the generic assessment suggests that the allowable disposal rate would be below $10^5$ te y$^{-1}$, based on a dose criterion of 1mSv y$^{-1}$ to landfill workers.

The annual doses from intrusion and to landfill workers from disposal of about $10^6$ tonnes of waste (approximately $10^5$ te y$^{-1}$) containing $^{226}$Ra at 4 MBq te$^{-1}$ would exceed or equal the dose criteria used for this study. Therefore the conservative assumptions made in the calculations could be reviewed in order to consider their appropriateness. In practical terms the annual doses to landfill workers at a particular site could be lower if there is additional covering of the waste. However, due to the long half-life of $^{226}$Ra, it would be difficult to place restrictions that would lessen the impact of an intrusion event. Based on the calculations presented here and the dose criteria adopted for this study, further consideration may need to be given to the assumptions made in the calculations before limiting the quantity of waste that can be disposed of.

For $^{232}$Th, the annual doses for disposal of about $10^6$ tonnes of waste with a concentration of 4 MBq te$^{-1}$ are below the selected dose criteria for all scenarios considered except for intrusion. For intrusion the annual doses to construction workers redeveloping the site 30 years after closure were estimated to be 5 mSv in a year. Again, based on the calculations presented here and the dose criteria adopted for this study, it may be necessary to review the assumptions made in the calculations before limiting the quantity of waste that can be disposed of.

For $^{238}$U, $^{239}$Pu and $^{241}$Am the annual doses from disposal of 2 $10^6$ tonnes of VLLW at a concentration of 4 MBq te$^{-1}$ are all below the selected dose criteria. Although the proposed definition of VLLW does not include mention of alpha-emitting radionuclides, this suggests that $^{238}$U, $^{239}$Pu and $^{241}$Am contaminated waste could be disposed of in large quantities at a landfill site if the definition of VLLW was extended to include them at a concentration of 4 MBq te$^{-1}$.

## 9.2    General discussion

Before placing limits on the quantities of VLLW that can be disposed of at a particular site it may also be advisable to review whether the dose criteria chosen for this study are appropriate (eg use of the dose criterion for workers if the site was on a nuclear licensed site; use of dose rather than risk for intrusion and migration scenarios). Since the models are linear, the results given in this study could be scaled accordingly.

This study shows that many of the estimated annual doses to the operators from disposal of large quantities of VLLW at the proposed definition are above a few tens of μSv per year. For landfill workers the annual doses for $^{60}$Co, $^{90}$Sr,

$^{137}$Cs, $^{226}$Ra, $^{232}$Th, $^{238}$U, $^{239}$Pu and $^{241}$Am were estimated to be greater than 20 µSv based on waste containing an activity concentration of 4 MBq te$^{-1}$. For the public exposed as a result of the migration of radionuclides through groundwater, the peak annual doses assuming disposal of large quantities of waste at activity concentration levels given by the proposed definition were all estimated to be less than 20 µSv except for $^{14}$C disposed of to an inert landfill site. For exposure from landfill gas all annual doses or activity concentration result in annual doses less than 20 µSv, except for $^{14}$C (at 4 or 40 MBq te$^{-1}$). All of the annual doses from inadvertent intrusion are predicted to be equal to or greater than 20 µSv except for $^{3}$H. However, using the probability of redevelopment of about $10^{-4}$ y$^{-1}$, all the estimated risks from inadvertent intrusion would be below the risk target of $10^{-6}$ y$^{-1}$.

The estimated annual doses from disposal of $10^{6}$ tonnes of three of the four example waste streams would meet the selected dose criteria for all the scenarios. The annual dose to workers from disposal of $10^{6}$ tonnes of one waste stream exceeded the dose criterion due to the high $^{137}$Cs content in the waste. Therefore, based on the results of this study and the adopted criteria, it would be advisable to do a more detailed site-specific assessment to determine the amount of waste with this radionuclide fingerprint that could be disposed of, or to identify any measures that would be needed to reduce the annual doses to the landfill workers.

The analysis of sensitive parameters indicates that the annual dose arising from disposal of $^{14}$C at 40 MBq te$^{-1}$ via discharge of leachate to river could be reduced by discharging the leachate to a river larger than the very small river in Table B8, or by placing the landfill site near the coast so that the groundwater migration is to the sea (with due consideration of the UK commitment to OSPAR). Varying the period of institutional control ie the time until the site is redeveloped showed that this is only important for the short-lived radionuclides ($^{3}$H, $^{60}$Co, $^{90}$Sr and $^{137}$Cs).

The CERRIE report (CERRIE, 2004) recommends that the uncertainty in the dose and risk estimates needs to be considered, particularly with respect to internal doses and risks. The Health Protection Agency's view is that the degree of uncertainty associated with ICRP dose coefficients varies with the radionuclide. For some radionuclides, e.g. isotopes of caesium and iodine, the uncertainties are of the order of two to three, but for other radionuclides, e.g. isotopes of plutonium, the uncertainties can exceed a factor of ten.

In summary, based on the calculations presented here and the criteria adopted for this study, a more detailed site-specific assessment would be necessary to determine the quantity of VLLW waste containing the radionuclides $^{14}$C (at 4 or 40 MBq te$^{-1}$), $^{60}$Co, $^{137}$Cs, $^{226}$Ra or $^{232}$Th in significant fractions that could be disposed of at a particular site. For the other radionuclides, $^{3}$H, $^{36}$Cl, and $^{90}$Sr disposal of a mass of approximately $10^{5}$ te y$^{-1}$ (or a total of about $10^{6}$ tonnes) at the VLLW concentration would meet the dose criteria specified in this study. The calculations presented here also show that disposal of a mass of approximately $10^{5}$ te y$^{-1}$ (or a total of about $10^{6}$ tonnes) of waste containing $^{238}$U, $^{239}$Pu or $^{241}$Am at 4 MBq te$^{-1}$ would also meet the dose criteria specified in this study. However, since alpha emitting radionuclides are not covered by the

## 9.3 Overall conclusions

The study concludes that if Defra formally adopted the dose criteria used in this study then there would be no need to limit the quantity of VLLW containing $^3$H, $^{36}$Cl, or $^{90}$Sr at the concentrations in the proposed definition that could be disposed of in a landfill site to below a guideline value of $10^6$ tonnes. Similarly, there would be no need to limit the quantity of waste containing $^{238}$U, $^{239}$Pu or $^{241}$Am at 4 MBq te$^{-1}$ to below $10^6$ tonnes. However, for $^{14}$C (at 4 or 40 MBq te$^{-1}$), $^{60}$Co, $^{137}$Cs, $^{226}$Ra and $^{232}$Th, the results of the generic assessment and sensitivity analysis indicated that it might be necessary to limit the quantity of waste containing these radionuclides at 4 MBq te$^{-1}$ or 4 Bq g$^{-1}$ to below $10^6$ tonnes, depending on whether the pessimistic assumptions adopted in this generic study were judged to be appropriate. Hence, for these five radionuclides a more detailed site-specific assessment would be needed to determine the quantity that could be disposed of and to identify any necessary conditions for disposal.

If Defra formally adopt different dose criteria then the results of this study can be scaled to determine the corresponding quantity of VLLW that can be disposed of at a landfill site.

The regulatory bodies would require Best Practicable Means (BPM) to be demonstrated for a dedicated facility for radioactive waste. Thus, even if the waste and site characteristics matched those of the generic assessment given in this report, the regulators would still require some form of site-specific assessment to show that the issues had been considered. However, such an assessment needs to be proportionate and could be based on the results of this study.

# 10   REFERENCES

Atkins Limited (2006). *Defra/EA VLLW disposal study*. 5039210.181(1).

Baes CF and Sharp RD  (1983 ). A Proposal For The Estimation Of Soil Leaching Or Leaching Constant For Use In Assessment Models. *J. Env. Qual.* **12,** 17.

Bagchi A (1990). *Design, construction and monitoring of a sanitary landfill*.

Batel W (1975).  Measurement of incidence of dust, noise and odour at workplaces in agriculture and ways of reducing it. *Grundl. Landtech* **25**, (5) 135.

CEC (1993). *Principles and methods for establishing concentrations and quantities (exemption values) below which reporting is not required in the European directive*.

Luxembourg, Radiation Protection-65.  Doc XI-028/93.

Chen Q, Degrange J-P, Gerchikov MY et al  (2003). *Effluent and Dose control from European Union NORM industries: Assessment of current situation and proposal for harmonised community approach: Vol I Main report*. Radiation Protection 135.

Chen Q, Kowe R, Jones A et al  (2004). *Development and application of a generic biosphere tool for dose assessments*. Chilton, NRPB EA/13/2004.

Commission of the European Communities  (1999). Council Directive 1999/31/EC of 26 April 1999 on the landfill of waste. *Off J Eur Commun* **L182/1**.

Copplestone D . Water to landfill distance.  13/7/2006.

Crawford MB, Kemp RV and Wilmot RD (2005). *Development of a Framework for Assessing the Suitability of Controlled Landfills to Accept Disposals of Solid Low-Level Radioactive Waste: Assessment methodology*. Galson Sciences Report 0240-2.

Cross W, Freedman N, Wong P (1992), *Tables of beta-ray dose distributions in water.* (AECL-10521). AECL

Defra (2005a). *The 2004 United Kingdom Radioactive Waste Inventory: Main Report*. DEFRA/RAS/05.002, Nirex Report N/090.

Defra (2005b). *Government Interpretation of the Landfill (England and Wales) Regulations 2002 (As Amended)*.

Defra, DOENI, Welsh Assembly et al  (2006). *A Public Consultation on Policy for the Long Term Management of Solid Low Level Waste in the United Kingdom*.

Dockery DW and Sprengler JD (1988). Indoor-outdoor relationships of respirable sulphates and particles. *Atmos. Environ.* **15**, 335.

Environment Agency (2003). *Hydrogeological risk assessment for landfills and the derivation of groundwater control and trigger levels*. Bristol, LFTGN01.

Environment Agency, Scottish Environment Protection Agency, Department of Environment NI et al  (2002). *Authorisation of Discharges of Radioactive Waste to the Environment, Principles for the Assessment of Prospective Public Doses, Interim Guidance*.

Environment Agency, Scottish Environment Protection Agency and Department of the Environment for Northern Ireland (1996). *Disposal facilities on land for low and intermediate level radioactive wastes: Guidance on requirements for authorisation*. London,

Gilby DJ and Hopkirk RJ (1985). *TROUGH-1D A one-dimensional computer code for calculation of radionuclide transport in groundwater*. NTB 85-38.

Harvey M, Mobbs S.F., Cooper JR et al  (1993). *Principles and Methods for Establishing Concentrations and Quantities (Exemption values) Below which Reporting is not Required in the European Directive*. Luxembourg, Radiation Protection 65.

Harvey MP, Barraclough IM, Mobbs SF and McDonnell CE (1995). *Review of the radiological*

*implications of disposal practices for very low level solid radioactive waste*.
DoE/HMIP/RR/95/006.

HMSO (2003). *Landfill (Scotland) Regulations 2003*. S.S.I 2003/235.

HPA  ( 2006). Dose criteria for the designation of radioactively contaminated land: Advice
from the Health Protection Agency. *Doc HPA Radiat Chem Environ Hazards*  **RCE-2,** (*1-3*).

IAEA (1988). *Principles for the exemption of radiation sources and practices from regulatory
control*. Vienna, Safety Series No 89.

ICRP  (1975). Reference Man: Anatomical, Physiological and Metabolic Characteristics.
ICRP Publication 23. *Ann ICRP*.

ICRP  (1983). Radionuclide transformations: energy and intensity of emissions. ICRP
Publication 38. *Ann ICRP* **11-13**.

ICRP  (1991). 1990 Recommendations of the international commission on radiological
protection. ICRP publication 60. *Ann ICRP*  **21,** (1-3).

ICRP  (1996). Age-dependent doses to members of the public from intake of radionuclides:
Part 5 Compilation of ingestion and inhalation dose coefficients. ICRP Publication 72. *Ann
ICRP* **26,** (1).

ICRP  (1998). Radiation Protection Recommendations as Applied to the Disposal of Long-
lived Solid Radioactive Waste.  ICRP Publication 81.   *Ann ICRP* **28,** (4).

Kelly M, Holton D, Lever DA and Williams SJ (2006a). *Assessment of the Consequences of
Onsite Disposal at Hinkley Point A: Data and Assumptions*. Serco Assurance Report
SA/ENV-0836.

Kelly M, Williams SJ, Holton D and Lever DA (2006b). *Assessment of the Consequences of
On-site Disposal at Hinkley Point A: Results and Conclusions*. SA/ENV-0827.

Martin JS, Barraclough IM, Mobbs SF et al  (1991). *User guide for BIOS_3A*. Chilton, NRPB-
M285.

Mobbs SM (2007). *[Personal communication]*.

National Council of Radiation Protection and Measurements (1996). *Screening  models for
releases of radionuclides to atmosphere, surface water and ground*. No. 123.

Negin CA  (1986). MICROSHIELD - A Microcomputer Program for Analysing Dose Rate and
Gamma Shielding. *Trans Am Nucl Soc* **53,** 421-422.

NRPB  (1997). Assessment of skin doses.  *Doc NRPB* **8,** (3).

NRPB (2000a). Generalised derived constraints for radioisotopes of strontium, ruthenium,
iodine, caesium, plutonium, americium and curium; Generalised derived limits for
radioisotopes of polonium, lead, radium and uranium. *Doc NRPB* **11**, (2) 1-71.

NRPB (2000b). *Limitation of Human Exposure to Radon in Homes*. Chilton, Vol 1, No 1.

NRPB . Generalised derived constraints for radioisotopes of polonium, lead, radium and uranium. *Doc NRPB* 13[No 2]. 2002.

NRPB EA (2004). *Models and Data Handbook (version 2)*.

Oatway W and Mobbs S (2003). *Methodology for Estimating the Doses to Members of the Public from the Future Use of Land Previously Contaminated with Radioactivity*. NRPB W-36.

OSPAR (1998), *SINTRA Statement.  Summary Record.* (OSPAR 98/14/1 Annex 45). London: OSPAR

Rau W and Heusser G (2000). $^{222}$Rn emanation measurements at extremely low activities. *Appl Radiat Isot* **53**, (1-2) 371-5.

Reedha D and Wilmot RD (2005). *Assessment model for disposals of solid low-level radioactive waste in controlled landfills: case study*. Galson report 0240-3 version 1.0.

Shelton PD (2004). *Near Surface Disposal – Impact of Coastal Erosion*. CoRWM Document No: 680.

Simmonds JR, Lawson G and Mayall A (1995). *Methodology for assessing the radiological consequences of routine releases of radionuclides to the environment*. Luxembourg, Radiation Protection 72, EUR 15760.

Simunek J, Van Genuchten MT and Sejna M (2005). *The Hydrus-1D Software Package for Simulating the One-Dimensional Movement of Water, Heat, and Multiple Solutes in Variably-Saturated Media*.

Smith GM, Fearn HS, Smith KR et al  (1988). *Assessment of the radiological impact of disposal of solid radioactive waste at Drigg*. Chilton, NRPB-M148.

Smith KR and Jones AL (2003). *Generalised Habit Data for Radiological Assessments*. NRPB-W41.

United Kingdom - Parliament (1995). *Review of radioactive waste management policy*. London, Cmnd 2919.

United Kingdom - Parliament (2000). *The Ionising Radiations Regulations 1999*. London, SI 3232.

UNSCEAR (1977). *Sources and Effects of Ionizing Radiation. Report to the General Assembly, with annexes*. United Nations sales publication E.77.IX.1.

US Nuclear Regulatory Commission (1999). *Radiological assessment for clearance of equipment and materials from nuclear facilities*. Washington, Nureg-1640.

USEPA (2004). *Review of Geochemistry and Available Kd Values for Americium, Arsenic, Curium, Iodine,Neptunium, Radium, and Technetium*. 402-R-04-002C.

Yim M-S, Simonson SA and Sullivan TM (1993). Modeling of gas-phase radionuclides release from low-level waste disposal facilities.

# APPENDIX A  Methodology and data for calculating doses to workers during the operational phase

## A1    INTRODUCTION

Four exposure pathways were considered for the calculation of annual doses to workers handling contaminated rubble at a sorting facility and disposing of rubble to landfill. These are external exposure to gamma emissions, inhalation of dust containing radioactivity, inadvertent ingestion of activity in dust, and beta doses from contamination on skin and gloves. The methodologies for estimating dose from those pathways are detailed in the following sections. As this is a generic assessment, conservative assumptions were made.

Since the aim of this assessment is to estimate the annual doses arising from the disposal of large quantities of VLLW to a landfill site, it was assumed that a landfill site operator or sorter could be in the vicinity of, or processing, VLLW for the entire working year. A typical disposal rate of $1.4 \ 10^5$ tonnes of waste to a landfill site per year was assumed.

It was pessimistically assumed that the worker has contaminated material on his skin/gloves for the entire working day and that any inhaled dust is contaminated. It was also assumed that the landfill worker is exposed to external irradiation from a large source for the entire working year. (The dose rate from standing on a block of waste 2 m x 2 m x 5 m containing $^{60}$Co is practically the same as that from standing on a semi infinite slab 5m thick. If a worker processes a 20 m$^3$ block of waste every hour ie 36 te h$^{-1}$ this implies that over an entire working year of 2000 h, the worker processes approximately $7 \ 10^4$ te).

All references quoted in this Appendix are listed in Section 10 of the main report.

## A2    EXTERNAL IRRADIATION OF SKIN

This represents the beta exposure of some fraction of the UV exposed areas of the hands, face and neck due to irradiation from contaminated material in contact with the skin, or the gloves. The resultant skin equivalent dose was converted into an effective dose. It was assumed that only the palm and one side of the digits are contaminated ie 400 cm$^2$ (area of palm with digits (ICRP, 1975)). The protection provided by the gloves against beta radiation was taken into account for the sorting workers.

It was assumed that there is a film of contaminated material on the skin or glove, which has the same activity concentration (Bq g$^{-1}$) as the contaminated material. It was also assumed that this film of contaminated material is there for the whole of the working day but is removed at the end of the shift. Beta dose rates to skin from material on skin were taken from (Cross et al, 1992). The beta

dose rates taken were those for the basal layer of the skin epidermis (7 mg cm$^{-2}$) for a plane isotropic source uniformly distributed over 100 cm$^2$ circular area as this is approximately the area of a hand.

*Equivalent dose to skin*

The equivalent dose is

$H_{skin}$ (Sv) = C * T * $\tau_{skin}$ * $\rho_{skin}$ * $DF_{\beta}$

where

C = Activity concentration in the waste, Bq g$^{-1}$

T = Time with activity on skin or gloves (Working year 2000 h y$^{-1}$, (Mobbs, 2007)). It was assumed that the activity is washed off at the end of the shift.

$\tau_{skin}$ = Thickness of the deposit on the gloves, cm (= 0.01 cm , (Harvey et al, 1995)).

$\rho_{skin}$ = Density of the deposit on the gloves, g cm$^{-3}$ (0.5 g cm$^{-3}$, for dispersible solids, (Harvey et al, 1995))

$DF_{\beta}$ = Skin equivalent dose rate to the basal layer of the skin epidermis (7 mg cm$^{-2}$) for a source uniformly distributed over 100 cm$^2$ for beta irradiation in contact, Sv h$^{-1}$ per Bq cm$^{-2}$ (Cross et al, 1992) (see Table A1).

*Effective dose from irradiation of skin*

The effective dose from contaminated dust on the skin, $D_{skin}$ (Sv), is given by:

$D_{skin}$ = $H_{skin}$ * $w_t$ * $F_{uv}$

where

$H_{skin}$ = Skin equivalent dose, Sv

$w_t$ = Tissue weighting factor for UV exposed skin ( = 0.01, (NRPB, 1997))

$F_{uv}$ = Fraction of UV exposed skin that is contaminated with dust

$$F_{uv} = \frac{A_{uv\_hfn}}{A_{uv\_tot}}$$

$A_{uv\_hfn}$ = Contaminated area of UVR exposed skin (= 400 cm$^2$ one side of adult hands (NRPB, 1997))

$A_{uv\_tot}$ = Total area of UVR exposed skin (= 3000 cm$^2$ (ICRP, 1975))

## A3     INHALATION OF SUSPENDED DUST/MATERIAL

Contaminated dust is generated as a consequence of the procedures involved in sorting and disposal of the waste rubble. Dust becomes suspended in the air and is then subsequently inhaled. The amount of dust suspended per unit volume of air is termed a dust loading. Ambient dust loadings may become locally enhanced for short periods of time when tasks which produce higher than average dust loadings are conducted, for example, the cutting of blocks of waste rubble to reduce their size. There may be deliberate spraying with water during particularly vigorous processes in order to reduce the dust loading and operatives may wear masks.

The amount of dust inhaled also depends on the nature of the various tasks that an individual might be doing. A more vigorous task, for example manual operations, will have associated with it a higher inhalation rate. Therefore, the amount of contaminated material inhaled is proportional to both the dust loading and the inhalation rate. It is assumed that dust is created as a result of the sorting procedure, or as a result of mechanical disturbance or wind-driven resuspension on the landfill site.

Since this is a generic assessment, cautious estimates of the doses were obtained by assuming that the resuspended dust that is inhaled is 100% contaminated material.

It was assumed that sorting workers are exposed to high dust levels during the crushing process (100 hours) at an average breathing rate and that they are wearing a mask but that this is not effective at stopping inhalation of contaminated dust particles. The landfill worker was assumed to be exposed to high dust levels for the period when he/she is outside, and lower dust levels for the time spent inside the cab of the bulldozer. It was assumed that the landfill operator is breathing at an average rate.

The general equation for the annual committed effective dose from the inhalation of contaminated material, $D_{inh}$ (Sv), for a sorter is:

$$D_{inh} = C * Inh_a * T * Dust_{out} * DC_{inh}$$

and for a landfill worker:

$$D_{inh} = C * Inh_a * (T_{in} * Dust_{in} + T_{out} * Dust_{out}) * DC_{inh}$$

where

$C$ = Activity concentration in the waste, Bq g$^{-1}$

$T$ = Time spent inhaling resuspended material for sorting worker (time spent by worker during concrete crushing 100 h y$^{-1}$, (CEC, 1993))

$T_{in}$ = Time spent by landfill worker inside cab (200 h y$^{-1}$)

$T_{out}$ = Time spent outside by landfill worker (1800 h y$^{-1}$)

$DC_{inh}$ = Dose coefficient for inhalation Sv Bq$^{-1}$ (ICRP, 1996) (see Table A2)

$Inh_a$ = Average breathing rate (1.69 $m^3 h^{-1}$), (Smith KR and Jones AL, 2003)

$Dust_{in}$ = Low dust loading factor for landfill worker inside (1 $10^{-5}$ g $m^{-3}$) (Dockery and Sprengler, 1988)

$Dust_{out}$ = High dust loading factor for sorting worker, landfill worker outside cab (1 $10^{-3}$ g $m^{-3}$) (Batel, 1975)


## A4    INADVERTENT INGESTION OF DUST/MATERIAL

This pathway considers the dose workers receive from inadvertently ingesting contaminated dust or material. Inadvertent ingestion of dust/material was assumed to occur due to the hand to mouth transfer of contaminated dust/material present on the skin. This was assumed to only occur when individuals are not wearing gloves. It was also assumed that any contaminated dust on the hands is removed when people wash their hands.

This pathway was considered for landfill workers only as the sorting workers are expected to wear gloves. The annual committed effective dose from inadvertent ingestion of dust, $D_{ing}$ (Sv), is given by:

$D_{ing} = C * T * Ing * DC_{ing}$

where

C = Activity concentration in the waste, Bq $g^{-1}$

T = Time spent with contaminated hands (Working year for landfill operator 2000 h $y^{-1}$)

Ing = Inadvertent ingestion rate for adults (= 5 x $10^{-3}$ g $h^{-1}$, (Smith KR and Jones AL, 2003)

$DC_{inh}$ = Dose coefficient for ingestion Sv $Bq^{-1}$ (ICRP, 1996) (see Table A2)


## A5    EXTERNAL EXPOSURE FROM A CONTAMINATED OBJECT

This pathway considered the exposure of the worker to external irradiation from radionuclides present in the contaminated material. The worker will be irradiated by different amounts of materials at different distances, dependant on the process: for sorting workers, the blocks or stockpiles of rubble were represented by a cube of waste 1 $m^3$ in size; for landfill workers working above trenches that are full of contaminated wastes, an extended source of an infinite extent and finite thickness was used.

The general equation for the dose from external irradiation, $D_{ext}$ (Sv), is given by:

$D_{ext} = C * T * DF_{ext}$

where

C = Activity concentration in the waste, Bq g$^{-1}$

T = Time spent exposed to contamination (Working year for landfill operator 2000 h y$^{-1}$, time spent by sorting worker during concrete crushing 100 h y$^{-1}$)

$DF_{ext} = DF_{gamma} + DF_{beta}$ (Sv h$^{-1}$ per Bq g$^{-1}$)

For gamma dose rates the following assumptions were made.

At the sorting facility it was assumed that the worker is irradiated at a distance of 1 m from a 1 m$^3$ cylinder of contaminated rubble with no shielding ( (Harvey et al, 1993). At the landfill site it was assumed that the worker is above an infinite slab of contaminated material 5 metres thick with no shielding by clean material or by the bulldozer.

The effective gamma dose rates $DF_{gamma}$ (Sv h$^{-1}$ per Bq g$^{-1}$) at a distance from an infinite and a finite source were obtained using the MICROSHIELD model (Negin, C. A., 1986) using ICRP-38 data (ICRP, 1983). The rubble was assumed to be contaminated concrete (density of 2.35 g cm$^{-3}$). The effective gamma dose rates for the cylinder (anterior/posterior geometry) and infinite slab (rotational geometry) are given in Table A3.

For beta dose rates the following assumptions were made.

Doses from bremsstrahlung were not considered because concrete has a low effective atomic number and hence does not give a significant bremsstrahlung dose in comparison with the beta dose (due to the low efficiency of conversion of beta energy into bremsstrahlung).

The effective beta dose rates for landfill workers were calculated using the following equation:

$DF_{beta} = DF_{slab} * GEOM$

where

$DF_{beta}$ = Beta dose rate (effective dose equivalent) at height of 1 m from a contaminated slab, Sv h$^{-1}$ per Bq g$^{-1}$.

$DF_{slab}$ = (Sv h$^{-1}$ per Bq g$^{-1}$ ) from beta particles 1 m above a semi-infinite slab (Harvey et al, 1993) (see Table A1)

GEOM = Geometry reduction factor from an infinite slab to a finite source. For finite beta sources, such as the 1 m$^3$ cylinder of concrete that the sorting worker is exposed to, GEOM was assumed to be 0.02 (Harvey et al, 1993). For landfill workers GEOM = 1 as it was assumed that they are standing on a contaminated semi-infinite plane.

## A6    TABLES OF PARAMETERS

**Table A1 Radionuclide dependent beta dose rates**

| Radionuclide | Mean energy per disinteration $E_{av}$ (MeV) (ICRP, 1984) | Skin equivalent dose rate (7 mg cm$^{-2}$) $DF_{\beta}$ (Sv h$^{-1}$ per Bq cm$^{-2}$) Cross (1992) | Dose rate at 1 m above semi-infinite slab $DF_{slab}$ (Sv h$^{-1}$ per Bq g$^{-1}$) Harvey et al (1993) |
|---|---|---|---|
| $^{3}$H | $5.68\ 10^{-3}$ | 0 | 0 |
| $^{14}$C | $4.95\ 10^{-2}$ | $3.14\ 10^{-7}$ | 0 |
| $^{36}$Cl | $2.74\ 10^{-1}$ | $1.65\ 10^{-6}$ | $3.19\ 10^{-11}$ |
| $^{60}$Co | $9.65\ 10^{-2}$ | $1.04\ 10^{-6}$ | 0 |
| $^{90}$Sr+ | $1.13\ 10^{0}$ | $3.25\ 10^{-6}$ | $3.95\ 10^{-9}$ |
| $^{137}$Cs+ | $2.48\ 10^{-1}$ | $1.54\ 10^{-6}$ | $1.76\ 10^{-11}$ |
| $^{226}$Ra+ | $1.37\ 10^{0}$ | 0 | $6.85\ 10^{-9}$ |
| $^{232}$Th | $1.25\ 10^{-2}$ | 0 | 0 |
| $^{238}$U | $1.00\ 10^{-2}$ | 0 | 0 |
| $^{239}$Pu | $6.65\ 10^{-3}$ | 0 | 0 |
| $^{241}$Am | $5.19\ 10^{-2}$ | 0 | 0 |

+ indicates that dose rates from progeny are included

**Table A2 Dose\* coefficients for workers for the selected radionuclides (ICRP, 1996)**

| Nuclide | Lung absorption type | $f_1$ | Ingestion dose (Sv Bq$^{-1}$) | Inhalation (Sv Bq$^{-1}$) |
|---|---|---|---|---|
| $^{3}$H | water | 1 | $1.80\ 10^{-11}$ | $1.80\ 10^{-11}$ |
| $^{14}$C | vapour | 1 | $5.80\ 10^{-10}$ | $5.80\ 10^{-10}$ |
| $^{36}$Cl | F | 1 | $9.30\ 10^{-10}$ | $4.90\ 10^{-10}$ |
| $^{60}$Co | S | 0.05 | $2.50\ 10^{-9}$ | $1.70\ 10^{-8}$ |
| $^{90}$Sr+ | F | 0.3 | $3.07\ 10^{-8}$ | $3.17\ 10^{-8}$ |
| $^{137}$Cs+ | F | 1 | $1.30\ 10^{-8}$ | $6.70\ 10^{-9}$ |
| $^{226}$Ra+ | M | 0.2 | $1.20\ 10^{-6}$ | $5.59\ 10^{-6}$ |
| $^{226}$Ra | M | 0.2 | $2.80\ 10^{-7}$ | $2.20\ 10^{-6\#}$ |
| $^{214}$Pb | M | 0.2 | $1.40\ 10^{-10}$ | $4.80\ 10^{-9}$ |
| $^{214}$Bi | M | 0.05 | $1.10\ 10^{-10}$ | $2.10\ 10^{-8}$ |
| $^{210}$Pb | M | 0.2 | $6.80\ 10^{-7}$ | $1.10\ 10^{-6}$ |
| $^{210}$Bi | M | 0.05 | $1.30\ 10^{-9}$ | $6.00\ 10^{-8}$ |
| $^{210}$Po | M | 0.1 | $2.40\ 10^{-7}$ | $2.20\ 10^{-6}$ |
| $^{232}$Th | M | $5.00\ 10^{-4}$ | $2.20\ 10^{-7}$ | $2.90\ 10^{-5}$ |
| $^{238}$U | S | 0.002 | $7.60\ 10^{-9}$ | $5.70\ 10^{-6}$ |
| $^{239}$Pu | M | $5.00\ 10^{-4}$ | $2.50\ 10^{-7}$ | $3.20\ 10^{-5}$ |
| $^{241}$Am | M | $5.00\ 10^{-4}$ | $2.00\ 10^{-7}$ | $2.70\ 10^{-5}$ |

\* Committed effective dose from intakes over a year

\# This value was taken from ICRP 72 Annex B: Inhalation dose coefficients for workers exposed to $^{226}$Ra (ICRP, 1996)

**Table A3 Gamma dose\* rates for concrete objects calculated using MICROSHELD (Negin, 1986)**

| Radionuclide | 1m from 1 m$^3$ cylinder (Sv hr$^{-1}$ per Bq g$^{-1}$) | 1 m above semi-infinite slab (Sv hr$^{-1}$ per Bq g$^{-1}$) |
|---|---|---|
| $^{3}$H | 0.0 | 0.0 |
| $^{14}$C | 0.0 | 0.0 |
| $^{36}$Cl | $4.32\ 10^{-12}$ | $1.17\ 10^{-11}$ |
| $^{60}$Co | $6.68\ 10^{-8}$ | $2.76\ 10^{-7}$ |
| $^{90}$Sr+ | $2.75\ 10^{-19}$ | $1.79\ 10^{-15}$ |
| $^{137}$Cs+ | $1.65\ 10^{-8}$ | $5.10\ 10^{-8}$ |
| $^{226}$Ra+ | $4.80\ 10^{-8}$ | $5.40\ 10^{-7}$ |
| $^{232}$Th | $5.09\ 10^{-12}$ | $5.68\ 10^{-11}$ |
| $^{238}$U | $6.10\ 10^{-13}$ | $5.46\ 10^{-12}$ |
| $^{239}$Pu | $1.82\ 10^{-12}$ | $2.35\ 10^{-11}$ |
| $^{241}$Am | $5.50\ 10^{-10}$ | $4.91\ 10^{-9}$ |

\* Effective dose

+ indicates dose rate from progeny included

# APPENDIX B Post-closure methodology and data

## B1    INTRODUCTION

This appendix describes the detailed methodology used in this study for the 'base-case' assessment of radiation doses following the closure of the site for the following post-closure scenarios.

- Migration to well. Radionuclides leach from the waste, move downward through the unsaturated zone to the groundwater table, followed by migration in the groundwater to a drinking water well.

- Migration to river. Radionuclides leach from the waste, move downward through the unsaturated zone to the groundwater table, followed by migration in the groundwater to a river used for drinking water and land irrigation.

- Gas. Inhalation of radioactive gases generated from landfill waste by groups living on the redeveloped site.

- Inadvertent intrusion. Excavation of the landfill site followed by building and occupying residential housing on top of the site.

Disposal of collected landfill leachate to a very small local river is addressed in the main report, together with the variations for the sensitivity analysis.

According to the classification of the EU Landfill Directive (Commission of the European Communities,   1999), three landfill classes were considered in terms of the landfill design requirements (HMSO, 2003): inert waste landfill, non-hazardous waste landfill and hazardous waste landfill. The characteristics of the landfills are described in the main report. All references quoted in this Appendix are listed in Section 10 of the main report.


## B2    LEACHATE GENERATION

### B2.1    Rainwater infiltration into a landfill site
Rainwater follows one of three main routes when it falls on soil. It may:

- "run off" over the surface

- evaporate or be used in the transpiration of plants

- percolate into the ground.

The water balance equation at the landfill is

$I_{rain} = I_{runoff} + I_{evap} + I_{inf}$

and the total amount of annual infiltration $Q_{inf}$ ($m^3$ $y^{-1}$) is

$Q_{inf} = I_{inf}$ AREA                                                                    B1

where

$I_{rain}$      = Annual precipitation rate (m y$^{-1}$)

$I_{runoff}$      = Annual run off (m y$^{-1}$)

$I_{evap}$      = Annual evaporation and transpiration rate (m y$^{-1}$)

$I_{inf}$      = Annual infiltration rate (m y$^{-1}$)

*AREA*   = Landfill area (m$^2$) (see Table B3)


For a typical soil, runoff and evapotranspiration can account for between 55 and 75% of the rainfall (Bagchi, 1990). When a low permeability capping system is constructed over a landfill surface, it can substantially increase the runoff. It is assumed in this study that the capping system prevents the infiltration into the landfill completely until it fully degrades. The infiltration rate ($I_{inf}$) was assumed to be 320 mm y$^{-1}$ (Atkins Limited, 2006). Leaching of radioactivity from the waste takes place when water percolates through the waste.

As shown in Figure 1 in the main text, it is assumed that no cap cover is in place on an inert landfill. Therefore infiltration occurs immediately after the landfill is closed. For a non-hazardous landfill, the cap was assumed to last for 50 years. According to the design criteria given in Defra's interpretation of the Landfill Directive (Defra, 2005b), a 5 mm y$^{-1}$ infiltration rate is allowed for a low permeability cap. Compared with the infiltration rate after the cap degrades, this amount of infiltration is trivial and hence it was assumed that no infiltration occurs before the cap has degraded. For a hazardous landfill, it was assumed that the cap degrades in 250 years time and therefore no infiltration in the first 250 years post-closure was assumed.

There may be some infiltration during the operational period of the site if the landfill surface is not adequately covered. The amount of the radioactivity in the waste lost to the percolating water during this period is difficult to calculate due to the variation of the waste thickness and unknown amount of infiltration. It has been conservatively assumed that no leaching of the waste occurrs during the operational period of the site.


## B2.2     Leach model
*Release from an inert type landfill*

The activity concentration in the leachate ($AC_{leachate}$, Bq m$^{-3}$) decreases with time as the inventory in the landfill reduces due to the combination of radioactive decay and loss through leaching. For an inert waste landfill, the activity concentration in leachate is calculated using the leach rate,

$$AC_{leachate}(t) = \frac{\lambda_L INV(0) e^{-(\lambda + \lambda_L)t}}{Q_{inf}}$$
         B2

where

$\lambda$ = Radioactive decay rate ($y^{-1}$)

$\lambda_L$ = Leach rate ($y^{-1}$)

INV(0) = Initial inventory in the landfill (Bq)

The 'leach rate' constant $\lambda_L$ is a first-order decay rate of the source similar to that of radioactive decay. It is calculated using the distribution coefficient method (National Council of Radiation Protection and Measurements, 1996) (Baes, C. F. and Sharp, R. D., 1983). This method assumes that when the percolating water comes into contact with the waste, equilibrium is achieved instantaneously between the passing water and the activity in the waste. The equilibrium is described by the distribution coefficient $K_d$ which assumes a linear relationship between the concentration in the waste ($AC_{waste}$, Bq $kg^{-1}$) and the concentration in the solution ($AC_{leachate}$, Bq $m^{-3}$), ie

$AC_{waste} = K_d\ AC_{leachate}$                                                B3

The leach rate constant $\lambda_L$ is given by (Baes, C. F. and Sharp, R. D., 1983)

$$\lambda_L = \frac{Q_{inf}}{V_L R \theta} = \frac{I_{inf}}{H_{landfill} R \theta} \qquad\qquad \text{B4}$$

where

R = Retardation coefficient of radionuclides in the waste (-) ( $R = \dfrac{1 + K_d \rho_b}{\theta}$ )

$K_d$ = Distribution coefficient of radionuclides in the waste (Bq $kg^{-1}$ per Bq $m^{-3}$) (see Table B2)

$\theta$ = Volumetric water content of the waste ($m^3 m^{-3}$) (see Table B3)

$\rho_b$ = Bulk density of the waste (kg $m^{-3}$) (see Table B3)

$H_{landfill}$ = Depth of waste (m) (see Table B3)

$V_L$ = Landfill volume ($m^3$) (see Table B3)

*Release from a non-hazardous type landfill*

For a non-hazardous type landfill, it is assumed that no leachate is produced before the cap fails at time t=T where T = 50 y.

After the cap fails, the concentration of the leachate above the clay liner is given by

$$AC_{leachate}(t \geq T) = \frac{\lambda_L INV(0) e^{-\lambda T} e^{-(\lambda + \lambda_L)(t-T)}}{Q_{inf}} \qquad\qquad \text{B5}$$

where

T = Time of cap failure for a non-hazardous landfill (y) (50 y)

The leach rate and the leachate concentration were obtained using equations B4 and B5.

*Release from a hazardous type landfill*

For the hazardous waste landfill site, it was assumed that the reworked clay cap does not degrade within the first 250 years. Infiltration was assumed to start in year 250 when the cap has gone. Assuming no infiltration before the cap fails (at t = 250 y), the leachate concentration is,

$$AC_{leachate}(t < T) = 0$$

The concentration of the leachate from year 250 onwards is,

$$AC_{leachate}(t \geq T) = \frac{\lambda_L INV(0)e^{-\lambda T}e^{-(\lambda+\lambda_L)(t-T)}}{Q_{inf}} \qquad \text{B6}$$

where

T = Time when the cap degrades (y) (250 y)

## B2.3   Fate of the leachate

Leachate containing dissolved radionuclides moves vertically downward towards the aquifer. However, all three landfill types are required to have a liner and leachate management systems (Defra, 2005b) that will reduce the amount of leachate that can eventually reach the groundwater table. The landfill sites are also required to have a leachate management system to collect any leachate that accumulates above the liner barrier.

Landfill leachate will therefore end up in one of two places.  A fraction of the leachate will be collected by the leachate management system and subsequently discharged into a river or sewer system.  The remaining leachate will percolate through the low-permeability liner barrier and migrate downwards into the underlying aquifer. Both cases were considered independently in this study.  The discharge of collected leachate to river scenario is covered in the main text. The migration scenario is discussed here.

## B3   MIGRATION SCENARIO

Three computer models were used for modelling the radionuclide migration scenario. The transport in the unsaturated zone was modelled by Hydrus-1D (Simunek et al, 2005), the horizontal flow in the saturated zone (aquifer) was modelled by Trough-1D (Gilby and Hopkirk, 1985), and the transport in the biosphere and subsequent doses were modelled using BIOS (Martin et al, 1991) (Chen et al, 2004).

### B3.1  Leachate migration in the unsaturated zone

The amount of the leachate that can seep through the clay liners was estimated using Darcy's law. If the infiltration flux exceeds the maximum hydraulic conductibility of the clay layers, the excess leachate will not be able to move downwards. Assuming a 0.5 m leachate head at the top of the clay liner, the maximum leachate rate $q_2$ (m y$^{-1}$) through the clay liners was calculated by,

$$q_2 = \min\left[ I_{inf}, -K\frac{\Delta P}{\Delta l} \right] = \min\left[ I_{inf}, K\left(1.0 + \frac{0.5}{H_{liner}}\right) \right] \qquad \text{B7}$$

$Q_2 = q_2\ AREA$

where

$Q_2$    = Leachate flux to the aquifer (m$^3$ y$^{-1}$)

K    = Hydraulic conductivity for the compacted clay liner (m y$^{-1}$) (3.15 m y$^{-1}$ for inert landfill, 3.15 10$^{-2}$ m s$^{-1}$ for non-hazardous and hazardous landfills (Atkins Limited, 2006)

H$_{liner}$  = Thickness of the compacted clay liner and the underlying natural clay layer (m) (1 m for inert and non-hazardous landfills, 6m for hazardous landfill)

$\frac{\Delta P}{\Delta l}$   = Hydraulic gradient (-), $\Delta P$ is the hydraulic head cross the geological barrier (m), $\Delta l$ is the thickness of the clay liner (m).

The one dimension radionuclide migration in the unsaturated zone was modelled using HYDRUS-1D (Simunek et al, 2005), a one dimensional model for modelling water and solute movement in variably saturated media. This numerical model was developed at the U.S. Soil Salinity Laboratory and has been updated to version 3.0 in 2005 (Simunek et al, 2005). Since 2004, it has been placed in the public-domain. The model was used to model the radionuclide transport through the clay liner and any underlying natural rock above the groundwater table (unsaturated zone). Hydrus-1D was selected because it is able to model radionuclide decay chains.

HYDRUS-1D was used to estimate the breakthrough time to the aquifer (ie the transit time in the unsaturated zone), and the breakthrough concentration profile for the selected radionuclides at the groundwater table.

Constant water flux and constant solute concentration boundary conditions were assumed for the upper boundary of the unsaturated zone, deep drainage and solute zero gradient boundary conditions for the bottom boundary.

The constant concentration at the upper boundary was obtained by approximating the leachate concentration curve (obtained from equations B2-B6) using a single pulse. The total inventory in the pulse was the same as the inventory in the landfill. The pulse concentration was assumed to be equal to the leachate concentration when leaching starts.

Default HYDRUS data for clay and sand such as soil retention and soil water retention function parameters were used, and the hydraulic conductivities were set to the values given in Table B4.

It should be noted that the conceptual model above does not take account of fissures or fractures in the unsaturated zone. The existence of fissures or fractures could lead to a shorter transit time and higher peak concentrations in the aquifer.

### B3.2    Transport in the saturated zone

The transport of radionuclides in the saturated zone is much better understood than that in the unsaturated zone. Seasonal fluctuation of the groundwater table was not considered and the groundwater flow in the aquifer was assumed to be in steady-state.

Advection, dispersion, radioactive chain decay and retardation are important factors affecting radionuclide transport in the aquifer. Although there are three dimensional aspects to the transport of radionuclides in the aquifer, a simple ID groundwater contaminant transport model is sufficient for a generic study. TROUGH-1D  (Gilby and Hopkirk, 1985)  is a one dimensional model that can cope with all the transport factors listed above and was therefore chosen for use in this study. TROUGH-1D was developed by Polydynamics Ltd in Switzerland in 1985 (Gilby and Hopkirk, 1985). TROUGH stands for Transport of Radionuclide Outflows in UnderGround Hydrology. The development of this computer code was partly funded by NRPB. TROUGH-1D was used in this study to model the transport of radionuclide in the aquifer from the landfill to the receptor points.

The aquifer was assumed to consist of homogeneous sandstone and the flow in the aquifer was assumed to be dominated by porous flow. The hydrogeological characteristics used for the aquifer are given in Table B5. It was also assumed that radionuclides reaching the aquifer are instantaneously fully mixed with the groundwater flowing horizontally through the aquifer. Although the mixing zone of the leachate in the aquifer is limited to the top part of the aquifer at the point where the leachate enters the aquifer, the entire thickness of the aquifer was assumed to be fully mixed at the receptor point where the groundwater is abstracted or discharged into a river.

The units of input and output in TROUGH-1D are atom $y^{-1}$. The output from HYDRUS-1D is the leachate flux entering the aquifer, in atoms $cm^{-2}$ $d^{-1}$. The HYDRUS output was converted to the format required by TROUGH using the equation below,

$$\text{Flux}_{\text{trough\_in}} = 3.65 \ 10^{6} \cdot \text{W}_{\text{landfill}} \cdot \text{L}_{\text{landfill}} \cdot \text{Flux}_{\text{hydrus}} \qquad\qquad \text{B8}$$

where

$3.65 \ 10^{6}$ represents the conversion from $cm^{2}$ $d^{-1}$ to $m^{2}$ $y^{-1}$

$\text{Flux}_{\text{trough\_in}}$ = Radionuclide flux into the aquifer (atom $y^{-1}$)

$\text{W}_{\text{landfill}}$ = Width of the landfill (m)

$\text{L}_{\text{landfill}}$ = Length of the landfill (m)

$Flux_{hydrus}$ = Leachate flux to the aquifer (atom cm$^{-2}$ d$^{-1}$)

Parameter values of other TROUGH-1D inputs are given in Table B5.

## B3.3     Radionuclide transfer in biosphere and annual doses to people
*Discharge to river*

Contaminated groundwater is eventually discharged into a river. The river characteristics assumed for this study are given in Table B8. These generic river characteristics are derived from the ones provided by Atkins (Atkins Limited, 2006), with additional information and discussion, particularly in relation to river volumetric flow rates. For the inert landfill, a large river was used and for the non-hazardous and hazardous landfills, a small river was used. The exposure pathways considered are given in Table B1. The compartmental model, PC BIOS (Martin et al, 1991) (Chen et al, 2004), was used to represent the transfer and dispersion of radionuclides through the biosphere and subsequently the peak annual doses to the public.

The biosphere was considered to be a terrestrial biosphere, i.e. an inland site. A coastal site is likely to give rise to lower doses due to the greater dilution obtained in the sea. A coastal site was included in the sensitivity study to examine this assumption.

## B3.4     Selection of distribution coefficient values
The distribution coefficient $K_d$ describes the sorption of an element onto a material. The sorption process is very important as it effectively slows down the migration of radionuclides, especially the strongly sorbed elements such as the actinides. In this study the values used for the soil and rock distribution coefficients are not site-specific. Therefore site conditions, e.g. pH of the groundwater and organic content of the soil or rock, were not taken into account. The $K_d$ values quoted in the literature vary considerably, mainly due to the dependence on the chemical properties of the elements and the chemical and physical composition of the soil or rock. The values were obtained as a result of reviewing a number of reports (Atkins Limited, 2006), (Smith et al, 1988) and (USEPA, 2004).

The general trend of $K_d$ values used in this study are given below,

-     H, C, Cl < Sr < Co, Cs, U < Pu < Am < Th

-     VLLW waste < aquifer = unsaturated sand < clay < river sediment

The $K_d$ values for the waste were conservatively chosen to be near the lower end of the $K_d$ range for the element.

The $K_d$ values for river sediment were taken from (Simmonds et al, 1995). Compared to $K_d$s of clays and sandstone, they are considerably higher. However the impact of this will be small as their usage is limited to the calculation of the concentration in the filtered water. Since the sedimentation load is small ($10^{-2}$ kg m$^{-3}$), the product of $K_d$ and sedimentation load does not have a significant impact on the concentration in the water.

Table B2 lists the $K_d$ values for the selected elements and their progeny in various media.

## B4    DISCHARGES OF LANDFILL GAS

Landfill gas is produced from organic waste in the landfill when the conditions become anaerobic. Landfill gas, eg methane gas, then seeps through the soil above the waste into the environment. For simplicity, the effect of a landfill cap on the landfill gas release rate was not taken into account. This is a pessimistic assumption and will result in the over-estimate of the annual doses from landfill gas for non-hazardous and hazardous landfills when the cap is intact. This gas could enter a building subsequently built on the site, resulting in doses to occupants.

For radioactive gases containing $^{3}$H (in the form of hydrogen, water, or methane) and $^{14}$C (in the form of carbon dioxide or methane), the release rate of gas $\lambda_{,gas}$ (Bq y$^{-1}$) at time $t$ years is given by (Crawford et al, 2005),

$$\lambda_{gas}(t) = \frac{INV(0)\ e^{-\lambda t} \cdot f_{gas}}{\tau_{gas}}$$    B9

where

INV(0)  = Initial activity of radionuclide in the waste (Bq)

$f_{gas}$      = Fraction of the activity associated with each gas (-) (0.039 for $^{3}$H, 0.2 for $^{14}$C)

$\tau_{gas}$    = Average timescale of generation of each gas (y) (50 years for both $^{3}$H and $^{14}$C, representative values)

When waste containing $^{226}$Ra is disposed in a landfill, exposure from radon could occur. The total release rate of radon over the entire site was calculated by,

$$\lambda_{radon}(t) = \lambda_{Rn-222} \cdot AREA \cdot AC_{Ra\text{-}226}\ e^{-\lambda_{Ra-226}t} \cdot \rho_{\text{waste}} \cdot \tau \cdot H_1 \cdot e^{\frac{-h_2}{H_2}}$$    B10

where

$AREA$  = Surface area of the disposal unit (m$^2$) (see Table B3)

$AC_{226Ra}$= Initial $^{226}$Ra concentration in the waste (Bq kg$^{-1}$)

$\rho_{waste}$   = Bulk density of the waste (kg m$^{-3}$) (Table B3)

$\tau$        = Emanation factor, defined as the fraction of the radon atoms produced which escape from the solid phase of the waste into the pore spaces (-) (0.1)

$H_1$      = Effective diffusion relaxation length for the waste (m) (0.2 m)

$h_2$      = Thickness of the cover (m) (1.5 m, minimum burial depth)

$H_2$      = Effective relaxation length of the cover (m) (0.2 m)

It was assumed that a building is constructed on top of the landfill 30 years after closure of the site. The associated air concentration of a radionuclide indoors (Bq m$^{-3}$) can be approximated by the total activity released into the house in the year divided by the volume of air passing through the house in a year (the maximum value will occur in the first year after the house is built). Thus for $^3$H and $^{14}$C,

$$AC_{gas} = \lambda_{gas} \cdot \frac{a_H}{AREA} \cdot \frac{1}{(\lambda_{house} V_{house})} \qquad \text{B11}$$

where:

$\dfrac{a_H}{AREA}$ = Horizontal area of a building divided by the area over which the radioactive gas is being released (-) ($a_H$ is 50 m$^2$)

$\lambda_{house}$ = Turnover rate of air in house (y$^{-1}$) (1 per hour, (UNSCEAR, 1977)

$V_{house}$ = Volume of the house (125 m$^3$, representative value)

Equation B13 is also used to calculate radon indoor concentration ($AC_{radon}$), substituting $\lambda_{radon}$ for $\lambda_{gas}$.

The resident's committed effective dose rate from inhalation of $^3$H and $^{14}$C gases (Sv y$^{-1}$) is given by,

$$DPU_{resident} = DC_{inh} \cdot AC_{gas} \cdot B_{inh} \cdot O_{indoor} \qquad \text{B12}$$

where

$DC_{inh}$ = Dose coefficient for inhalation of radionuclide (Sv Bq$^{-1}$) (Table A2)

$B_{inh}$ = Breathing rate (m$^3$ y$^{-1}$) (Table B7)

$O_{indoor}$ = Indoor occupancy of the residents (-) (Table B7)


## B5    HUMAN INTRUSION SCENARIOS

Inadvertent intrusion of the landfill site will result in radiation exposures. It is envisaged that after the landfill has been closed for a period of time, the site will be re-developed for residential housing. Two cases were considered: excavation/construction and residence. If houses are built on the landfill site, the construction process may disturb buried material and redistribute it on the surface (for example in the gardens). The construction workers and subsequently housing residents will be exposed to contaminated material from the site, albeit in a form where it is diluted with other soil etc. Exposures could be received via external irradiation from the soil or as a result of intakes following inhalation of resuspended contaminated dust, inadvertently ingestion of contaminated soil and vegetables grown in the soil. The annual doses were calculated using the construction and residence scenarios for disturbed buried

contamination given in the CONLAND model (Oatway and Mobbs, 2003), and scaling by the concentrations in the soil.

It is unrealistic to assume the landfill site is redeveloped immediately after its closure. A period of 30 years is the time which is generally used in landfill assessments as it is felt to be representative of the time at which active management will stop and people will use the site either because information of the site has been lost or it is re-claimed after safety assessments. The 30 years period is also the time of one generation.

The dose contribution from progeny was included if the ingrowth of progeny is significant during the 30 years prior to the intrusion. The criterion for significant ingrowth was that the progeny's concentration exceeds 1% of the parent (chain head) concentration at year 30.

## B5.1    Construction site scenario

It was assumed the landfill site is developed after 30 years since its closure. Radioactive decay was considered for this period. No radioactivity loss to leaching or other processes, e.g. plant uptake, were considered. The remaining activity in the landfill was assumed to be distributed evenly in the landfill. The activity concentration in the soil ($C_{soil}$, Bq g$^{-1}$) is,

$$C_{soil}(30) = \frac{INV(30)}{V_L \cdot \rho_{waste}}$$    B13

where

$INV$(30) = Remaining activity out of the initial inventory at the time of excavation (Bq)

$\rho_{waste}$    = Waste density, equal to 1.8 10$^6$ g m$^{-3}$

$V_L$    = Volume of landfill (m$^3$) (see Table B3)

This soil concentration was then used to scale the annual doses given in the CONLAND model (Oatway and Mobbs, 2003). The annual dose to a worker comes from the exposure pathways listed below:

- external from ground when on-site

- external from contaminated soil on the skin

- inhalation of suspended contaminated material when on-site

- inadvertent ingestion of contaminated soil from the site.

The construction worker's dose per unit activity was calculated,

$$DPU_{intrusion\_worker} = D_{LM} \cdot C_{soil}(30)$$    B14

where

DPU$_{intrusion\_worker}$ = Annual dose to a construction worker (Sv received in a year)

$D_{LM}$ = Workers' dose per unit concentration given in CONLAND model (Oatway and Mobbs, 2003) (Sv received in a year per Bq g$^{-1}$ in soil)

## B5.2   Residential housing scenario

The residential housing scenario considered the doses to residents living in a housing estate built on the landfill site. It was assumed that:

- the residents move into the housing after the landfill site has been closed for 30 years,

- the area of the house is 200 m$^2$ plus 100 m$^2$ garden area,

- root vegetables and fruit are grown in the garden for self consumption.

The annual dose to a resident comes from the exposure pathways below:

- external irradiation from ground

- external irradiation from contaminated soil on the skin

- inhalation of suspended contaminated material in the garden and house

- ingestion of vegetables grown in the garden

- inadvertent ingestion of contaminated dirt from the garden.

The annual dose to a housing resident was calculated using the following equation,

$$DPU_{intrusion\_resident} = DIL \cdot C_{soil}(30) \cdot D_{housing} \qquad \text{B 15}$$

where

$DPU_{intrusion\_resident}$ = Annual dose to housing residents (Sv received in a year)

$DIL$ = Dilution factor (-) (2/3)

$D_{housing}$ = Residents' dose per unit concentration given in CONLAND model (Oatway and Mobbs, 2003) (Sv received in a year per Bq g$^{-1}$ in soil)

## B6    TABLES OF PARAMETERS

**Table B1 Terrestrial pathways considered in BIOS**

Exposure pathways considered for terrestrial biosphere

Drinking filtered water from river

Consumption of freshwater fish caught in the river

Consumption of beef from cows grazing on land contaminated by irrigation water

Consumption of cow liver

Consumption of cows milk

Consumption mutton from sheep grazing on pasture contaminated by irrigation water

Consumption of sheep liver

Consumption of green vegetable grown on arable land irrigated by river water

Consumption of orchard and soft fruit irrigated by river water

Consumption of grain

Consumption of root vegetable

Consumption of poultry and eggs produced by chickens fed on contaminated water and cereal

Consumption of pork fed on contaminated water and cereal

Inhalation of airborne dust suspension from pasture land and arable land

External gamma radiation from land

Freshwater fish consumption

**Table B2 Geosphere distribution coefficients ($K_d$, m$^3$ kg$^{-1}$) for radionuclides\***

| Element/Isotope | Waste\*\* | Clay Barrier[†] | Unsaturated[‡] | Aquifer[§] | River Sediment\*\*\* |
|---|---|---|---|---|---|
| H | 0 | 0 | 0 | 0 | $3\ 10^{-5}$ |
| C | 0 | 0 | 0 | 0 | 2 |
| Cl | 0 | 0 | 0 | 0 | 1 |
| Co | 0[††] | 1 | 0.1 | 0.1 | 20 |
| Sr | 0.01 | 0.1 | 0.01 | 0.01 | 2 |
| Cs | 0.1 | 1.8 | 0.2 | 0.2 | 2 |
| Ra | 0.5 | 1 | 0.50 | 0.50 | 0.5 |
| Th | 0.5 | 5.4 | 1 | 1 | $5\ 10^3$ |
| U | 0.1 | 1.5 | 0.01 | 0.01 | 0.05 |
| Pu | 0.2 | 4.9 | 0.6 | 0.6 | 100 |
| Am | 0.1[‡‡] | 8.1 | 0.7 | 0.7 | 400 |
| Pb | - | 0.5 | 0.2 | 0.2 | 10 |
| Po | - | 0.2 | 0.01 | 0.01 | 10 |
| Np | - | 0.2 | 0.03 | 0.03 | 30 |

\*  $K_d$ values are taken from various sources (Atkins Limited, 2006), (Crawford et al, 2005), (Mobbs, 2007),  (Reedha and Wilmot, 2005) and (NRPB, 2004).

\*\* from (Atkins Limited, 2006). Co and Am values are derived elsewhere.

[†]  from Table 4-10 in (Kelly et al, 2006a)

[‡]  as in aquifer

[§]  from (Mobbs, 2007)

\*\*\* from Table 2 of MDH aquatic modelling project committee report v2 (NRPB, 2004)

[††]  based on chemical analogy to chlorine

[‡‡]  judgement based on (Crawford et al, 2005) (Mobbs, 2007)

**Table B3 Default values of the 3 landfill types for the 'best case' dose assessments**

| Waste zone | Symbol | Unit | Landfill class | | |
|---|---|---|---|---|---|
| | | | Inert | Non-hazardous | Hazardous |
| Annual rainfall | $I_{rain}$ | m y$^{-1}$ | 1.078 | 1.078 | 1.078 |
| Infiltration rate | $I_{inf}$ | m y$^{-1}$ | 0.32 | 0 before the cap degrades 0.32m y$^{-1}$ afterwards | 0 before the cap degrades 0.32m y$^{-1}$ afterwards |
| Time of capping failure | T | y | N/A | 50 | 250 |
| *Leach rate* | $\lambda_L$ | y$^{-1}$ | nuclide dependent, calculated using equation B4 | | |
| landfill area | AREA | m$^2$ | 80000 | 80000 | 80000 |
| landfill width | $W_{landfill}$ | m | 140 | 140 | 140 |
| landfill length | $L_{landfill}$ | m | 570 | 570 | 570 |
| waste thickness | $H_{waste}$ | m | 15 | 15 | 15 |
| Landfill volume | $V_L$ | m$^3$ | 1.2 10$^6$ | 1.2 10$^6$ | 1.2 10$^6$ |
| waste bulk density | $\rho_b$ | kg m$^{-3}$ | 1.8 10$^3$ | 1.8 10$^3$ | 1.8 10$^3$ |
| waste porosity | $\Theta$ | - | 20% | 20% | 20% |
| initial source inventory | INV | Bq | 1 10$^6$ | 1 10$^6$ | 1 10$^6$ |
| waste-water distribution coefficient | $K_d$ | m$^3$ kg$^{-1}$ | see Table B2 | | |
| leachate rate | $I_{inf}$ | m$^3$ y$^{-1}$ per m$^2$ | 0.32 | 0.32 | 0.32 |
| **liner / geological barrier** | **symbol** | **unit** | **inert** | **non-hazardous** | **hazardous** |
| hydraulic conductivity | K | m s$^{-1}$ | 1 10$^{-7}$ | 1 10$^{-9}$ | 1 10$^{-9}$ |
| thickness | $H_{liner}$ | m | 1 | 1 | 1 |
| bulk density | $\rho_b$ | kg m$^{-3}$ | 1.5 10$^3$ | 1.5 10$^3$ | 1.5 10$^3$ |
| effective porosity explain in text | $\theta_e$ | - | 1% | 1% | 1% |
| total porosity | $\theta_t$ | - | 50% | 50% | 50% |
| diffusivity in clay liner | $D_m$ | cm$^2$ s$^{-1}$ | 10$^{-6}$ | 10$^{-6}$ | 10$^{-6}$ |
| head of leachate | H | m | 0.5 | 0.5 | 0.5 |
| leachate flux to aquifer | $q_2$ | m$^3$ y$^{-1}$ per m$^2$ | 0.32 | 0.04 | 0.03 |
| **unsaturated zone beneath the liner** | **symbol** | **unit** | | | **clay** |
| hydraulic conductivity | K | m s$^{-1}$ | | | 1 10$^{-9}$ |
| thickness | $H_{unst}$ | m | | | 5 |
| bulk density | $\rho_b$ | kg m$^{-3}$ | | | 1.3 10$^3$ |
| effective porosity | $\theta_e$ | - | | | - |
| total porosity | $\theta_t$ | - | | | 50% |
| **aquifer** | **symbol** | **unit** | **inert** | **non-hazardous** | **hazardous** |
| hydraulic conductivity | K | m s$^{-1}$ | 1 10$^{-3}$ | 1 10$^{-5}$ | 1 10$^{-5}$ |
| hydraulic gradient | I | - | 7 10$^{-3}$ | 7 10$^{-3}$ | 7 10$^{-3}$ |
| porosity | $\Theta$ | - | 20% | 20% | 20% |
| **aquifer** | **symbol** | **unit** | **inert** | **non-hazardous** | **hazardous** |
| bulk density | $\rho_b$ | kg m$^{-3}$ | 1.8 10$^3$ | 1.8 10$^3$ | 1.8 10$^3$ |
| thickness of aquifer | $H_{aquifer}$ | m | 10 | 10 | 10 |
| width of aquifer | $W_{aquifer}$ | m | 140 | 140 | 140 |

| | | | | | |
|---|---|---|---|---|---|
| groundwater pore water flow rate | $V$ | m s$^{-1}$ | 3.5 10$^{-5}$* | 3.5 10$^{-7}$** | 3.5 10$^{-7}$ |
| *Length of aquifer* | L | m | 250 | 250 | 250 |
| longitudinal dispersivity | $\alpha_L$ | m | 25 | 25 | 25 |
| *distribution coefficient of aquifer* | $K_d$ | m$^3$ kg$^{-1}$ | radionuclide-dependent, Table B2 | | |
| Landfill gas | symbol | unit | inert | non-hazardous | hazardous |
| Volume of the house | $V_{house}$ | m$^3$ | 125 | 125 | 124 |
| Turnover rate of air in the house | $\lambda_{house}$ | y$^{-1}$ | 8700 | 8700 | 8700 |
| *Time of dose assessment* | T | y | 30 | 30 | 30 |

*italic font* parameters are selected for the sensitivity study

*major UK aquifer

**minor UK aquifer

**Table B4 Hydrus-1D input parameters**

| Main process | solute transport, general solute transport |
|---|---|
| Length Unit | cm |
| Time Unit | day |
| Depth of the Soil Profile | 100 cm for inert, 200 cm for non-hazardous, 600 cm for hazardous |
| Number of Materials | 1 for inert, 1 for non-hazardous, 2 for hazardous (100 cm layer 1, 500 cm layer 2) |
| Number of Subregions | 1 for inert, 1 for non-hazardous, 2 for hazardous |
| Decline from Vertical | 1 for vertical column |
| **Time Discretisation:** | |
| Initial Time | 0 |
| Final Time | 100,000 days [*] |
| Initial Time Step | 1 day |
| Minimum Time Step | 1 day |
| Maximum Time Step | 5,000 day |
| **Iteration Criteria and Time Step Control**: | |
| Iteration Criteria | max number of iteration = 10, water content of tolerance 0.01, pressure head tolerance = 1 |
| Time Step Control | lower optimal iteration range = 3, upper optimal iteration range = 7.0; lower time step multiplication factor = 1.3, upper time step multiplication factor = 0.7 |
| Internal Interpolation Tables | lower limit of the tension interval = $1 \; 10^{-6}$, upper limit of the tension interval = $1 \; 10^{5}$ |
| Model of Hydraulic Properties | use Single porosity model of van Genuchten-Mualem, no hysteresis considered |
| **Water flow parameters:** | |
| Residual soil water content | 0.0792 for clay |
| Saturated soil water content | 0.50 for clay |
| Soil water retention function parameter Alpha ($m^{-1}$) | 0.0158 for clay |
| Parameter in the soil water retention function n | 1.4145 for clay |
| Saturated hydraulic conductivity | $0.864 \; cm \; d^{-1}$ for inert landfill clay and $8.64 \; 10^{-3} \; cm \; d^{-1}$ for non-hazardous and hazardous landfill clay, $86.4 \; cm \; d^{-1}$ for sand |
| Pore connectivity parameter | 0.5 |
| Hydraulic properties are considered to be not temperature dependent. | |
| Water Flow Boundary Conditions (BC): | Constant flux for upper BC |
| | Constant flux for bottom BC |
| Initial condition | pressure head |
| **Water Flow Constant Boundary Fluxes:** | |
| Upper boundary flux | $0.32 \; m \; y^{-1} = 0.09 \; cm \; d^{-1}$ for inert, 0.013 for non-hazardous, and 0.009 for hazardous |
| Bottom boundary flux | $0.09 \; cm \; d^{-1}$ for inert, 0.013 for non-hazardous, and 0.009 for hazardous |
| General Solute Transport Information: | |
| Time Weighing Scheme | 0.5 for a Crank-Nicholson implicit scheme |
| Space Weighing Scheme | Galerkin formulation |
| Number of solutes involved in the decay chain reaction | 1 ~ 3 (nuclide-dependent) |

[*] The largest length unit in HYDRUS-1D is centimetre. The largest time unit is day.

| Main process | solute transport, general solute transport |
|---|---|
| Iteration Criteria | linear sorption isotherm is assumed, therefore the equations are linear, no need for iteration, Absolute Concentration Tolerance = 0, Relative Concentration Tolerance = 0, Maximum Number of Iterations = 1 |
| Pulse Duration | varied in years (this is calculated using leach model then approximated using a single pulse) |
| Stability criterion (Peclet number x Curant number) | 2.0 |
| Bulk density | 1.5 g cm$^{-3}$. For hazardous landfill, layer 1 is 1.5 g cm$^{-3}$, layer 2 is 1.3 g cm$^{-3}$) |
| Longitudinal dispersivity | 10% of the unsaturated zone thickness (10 cm for inert, 10 cm for layer 1 of non-hazardous, and 10 cm for layer 1 and 50 cm for layer 2 of hazardous landfill) |
| Dimensionless fraction of adsorption sites | 1 |
| Immobile water content | 0 |
| Molecular diffusion coefficient in free water | $10^{-6}$ cm$^2$ s$^{-1}$ ≈ 0.09 cm$^2$ d$^{-1}$ |
| | (diffusion could be important for low permeability soil) |
| Molecular diffusion coefficient in soil air (cm$^2$ d$^{-1}$) | 0 |
| **Solute Reaction Parameters:** | |
| Adsorption isotherm coefficient | $K_d$ for radionuclide in clay (cm$^3$ g$^{-1}$), e.g. Pu is 4.9 m$^3$ kg$^{-1}$ = 4900 cm$^3$ g$^{-1}$. See Table B2 |
| Henry equilibrium distribution constant between liquid and gaseous phases | 0 |
| First-order rate constant for dissolved phase 1 | radioactive decay rate of the parent |
| First-order rate constant for solid phase 1 | radioactive decay rate of the parent |
| First-order rate constant for gas phase 1 | no gas phase exist, set to 0 |
| First-order rate constant for dissolved phase 2 | Only required if progeny present |
| First-order rate constant for solid phase 2, representing the chain reaction | Only required if progeny present |
| First-order rate constant for gas phase 2, representing the chain reaction | Only required if progeny present |
| Zero-order rate constant for dissolved phase 1 | Not considered |
| Zero-order rate constant for solid phase 1 | Not considered |
| Zero-order rate constant for solid phase 1 | Not considered |
| First-order rate coefficient for one-site or two-site non-equilibrium adsorption, mass transfer coefficient for solute exchange between mobile and immobile liquid regions | Not considered |
| **Solute Transport Boundary Conditions:** | |
| upper boundary condition: | Concentration boundary condition (leachate concentration calculated using Equations 2, 4-6) |
| lower boundary condition: | Free drainage (zero gradient) |
| soil profile: | material 1 = 100 cm (inert landfill) |
| | material 1 = 100 cm (non-hazardous) |
| | material 1 = 100 cm, material 2 = 500 cm (hazardous). |
| | H = -100 |
| | initial concentration = 0 |
| | temperature = 20 degree centigrade |

## Table B5 TROUGH-1D input data

| Variable | Meaning | Value |
|---|---|---|
| NLAYER | number of geosphere zones | 1 |
| NREST | 0 - new calculation; 1 - restart cal; 2 - restart facility not used | 1 |
| NSTEP | max time steps | 800 |
| KOORD | 0 - linear transport in rectangular; 1- radial transport in cylindrical | 0 |
| KFLOW | Kflowd = 1; Kflowz = 3 | 1 |
| NNUC | members in a chain | nuclide-dependent, e.g. 1 for single radionuclide |
| NCHAIN | number of separate chains to be considered | nuclide-dependent, e.g. 0 indicates only one radionuclide is considered |
| NLINK | solubility structure for multiple chain, not considered | 0 |
| KSOURCE | 1 – constant rate;<br>2 – time dependent rate ARFM();<br>3 – rate or boundary conc. time dependent;<br>4 – rate and boundary conc. from time history | 4 |
| INVENT | Inventory definition:<br>1 - Bq in and Bq out<br>2 - moles in and Bq out<br>3 - Bq in and moles out<br>4 - atom in and atom out | 4 (outputs from HYDRUS is in unit of atom $m^{-3}$) |
| INJECT | no of the cell into which the release takes place<br>1 – release value as inlet boundary concentration | 1 |
| KSORB | 1 – equilibrium sorption, AKD = retard;<br>otherwise, calculate AKD | 1 |
| KSAT | 0 - no precipitation; 1 - precipitation; | 0 |
| KMDIFF | 0 - matrix diffusion; 1 – diffusion into solid slabs; 2 – diffusion into solid spheres | 0 |
| TIMEX | Expansion factor for time step size | 1.05 |
| TIME | absolute time at start of cal. | 0 |
| DY | width of layer | 1 |
| RNAME | radionuclide name | nuclide-dependent e.g. 'I129' |
| ALAM | decay constant ($y^{-1}$) | nuclide-dependent e.g. $4.415 \times 10^{-8}$ for $^{129}$I |
| STINV | inventory at time zero | used for calculating influx concentration, **not to be used** as the influx concentration will be replaced by HYDRUS output |
| KD(LAYER,NNUC) | layer 1, if KSORB =1 , retardation; if K≠1, AKD = Kd | 1 |
| AKD(LAYER,NNUC) | layer 2, as above | 1 |

| Variable | Meaning | Value |
|---|---|---|
| RELRATE | release rate ($y^{-1}$) | used for calculating influx concentration, **not used** as the influx concentration will be replaced by HYDRUS output |
| TREL | release period (y)<br>after TREL, RELRATE =0 | used for calculating influx concentration, **not used** as the influx concentration will be replaced by HYDRUS output |
| WDEN | water density (kg $m^{-3}$) | $1 \times 10^3$ (at $20^oC$ the density of pure water is 1000 kg $m^{-3}$) |
| IFLOWD | integer number | 0 |
| FLOWD | real number | 0.0 |
| CINIT | concentration at time zero | 0 |
| HSL | | 0 |
| HSR | | 0 |
| THETA | solver option; 1 – fully implicit; 0 - explicit; 0.5 - semi implicit | 1 |
| TOL | solver tolerance | $1.0 \times 10^{-7}$ |
| CSTOP | | 0 |
| STOL | | $1.0 \times 10^{-6}$ |
| NOJUMP | jump factor for timestep output. n, only timestep 0, n, 2n will be printed | 1 |
| NHJUMP | | 1 |
| NSJUMP | | 200 |
| NIJUMP | jump factor for iteration | 999 |
| NPJUMP | jump factor for precipitation | 999 |
| NOJ2 | | 1 |
| NHJ2 | | 1 |
| NSJ2 | | 200 |
| KPRINT(I), I=1,10 | printing control parameters | 0  0  0  0  0  0  1  0  0  0 |
| DELAY | delay time before release (y) | used for calculating influx concentration, not to be used as the influx concentration will be replaced by HYDRUS output |
| RHOS | particle density of aquifer material (kg $m^{-3}$) | $2.60 \times 10^3$ |
| LENLAY | aquifer length (m) | 250 |
| VEL | groundwater velocity (m $y^{-1}$) | 11 for a typical minor aquifer;<br>1100 for a typical major aquifer |
| EPS | aquifer porosity | 0.2 |
| DISP | longitudinal dispersivity (m), 10% of travel length | 25 |

**Table B6 Dose coefficients of selected radionuclides (ICRP, 1996; ICRP, 1994)**

Committed effective dose per unit intake by ingestion  (Sv Bq$^{-1}$)

| Nuclide | | $f_1$ | Infant (1y) | Child (10 y) | Adult (20 y) |
|---|---|---|---|---|---|
| $^3$H | | 0.1 | $4.8\ 10^{-11}$ | $2.3\ 10^{-11}$ | $1.8\ 10^{-11}$ |
| $^{14}$C | | 0.1 | $1.6\ 10^{-9}$ | $8\ 10^{-10}$ | $5.8\ 10^{-10}$ |
| $^{36}$Cl | | 0.1 | $6.3\ 10^{-9}$ | $1.9\ 10^{-9}$ | $9.3\ 10^{-10}$ |
| $^{60}$Co | | 0.1 | $2.7\ 10^{-8}$ | $1.1\ 10^{-8}$ | $3.4\ 10^{-9}$ |
| $^{+90}$Sr | | 0.1 | $9.3\ 10^{-8}$ | $6.6\ 10^{-8}$ | $3.1\ 10^{-8}$ |
| $^{+137}$Cs | | 1 | $1.2\ 10^{-8}$ | $1.0\ 10^{-8}$ | $1.3\ 10^{-8}$ |
| $^{+226}$Ra | | 0.1 | $9.6\ 10^{-7}$ | $8.0\ 10^{-7}$ | $2.8\ 10^{-7}$ |
| | $^{+210}$Pb | 0.1 | $3.6\ 10^{-6}$ | $1.9\ 10^{-6}$ | $6.9\ 10^{-7}$ |
| | $^{210}$Po | 0.1 | $8.8\ 10^{-6}$ | $2.6\ 10^{-6}$ | $1.2\ 10^{-6}$ |
| $^{232}$Th | | $5\ 10^{-4}$ | $3.7\ 10^{-7}$ | $1.5\ 10^{-7}$ | $7.2\ 10^{-8}$ |
| | $^{+228}$Ra | $5\ 10^{-4}$ | $5.7\ 10^{-6}$ | $3.9\ 10^{-6}$ | $6.9\ 10^{-7}$ |
| | $^{+228}$Th | $5\ 10^{-4}$ | $3.7\ 10^{-7}$ | $1.5\ 10^{-7}$ | $7.2\ 10^{-8}$ |
| $^{+238}$U | | $5\ 10^{-4}$ | $1.5\ 10^{-7}$ | $7.5\ 10^{-8}$ | $4.8\ 10^{-8}$ |
| | $^{234}$U | $5\ 10^{-4}$ | $1.3\ 10^{-7}$ | $7.4\ 10^{-8}$ | $4.9\ 10^{-8}$ |
| | $^{+230}$Th | $5\ 10^{-4}$ | $4.1\ 10^{-7}$ | $2.4\ 10^{-7}$ | $2.1\ 10^{-7}$ |
| $^{239}$Pu | | $5\ 10^{-4}$ | $4.2\ 10^{-7}$ | $2.7\ 10^{-7}$ | $2.5\ 10^{-7}$ |
| | $^{+235}$U | $5\ 10^{-4}$ | $2.5\ 10^{-7}$ | $2.7\ 10^{-7}$ | $4.2\ 10^{-7}$ |
| $^{241}$Am | | $5\ 10^{-4}$ | $3.7\ 10^{-7}$ | $2.2\ 10^{-7}$ | $2.0\ 10^{-7}$ |
| | $^{+237}$Np | $5\ 10^{-4}$ | $2.2\ 10^{-7}$ | $1.1\ 10^{-7}$ | $1.1\ 10^{-7}$ |

Committed effective dose per unit intake by inhalation  (Sv Bq$^{-1}$)

| Nuclide | | Absorption type | Infant (1y) | Child (10 y) | Adult (20 y) |
|---|---|---|---|---|---|
| $^3$H | | Vapour | $4.8\ 10^{-11}$ | $2.3\ 10^{-11}$ | $1.8\ 10^{-11}$ |
| $^{14}$C | | Vapour | $6.6\ 10^{-9}$ | $2.8\ 10^{-9}$ | $2.0\ 10^{-9}$ |
| $^{36}$Cl | | M | $2.6\ 10^{-8}$ | $1.0\ 10^{-8}$ | $7.3\ 10^{-9}$ |
| $^{60}$Co | | M | $3.4\ 10^{-8}$ | $1.5\ 10^{-8}$ | $1.0\ 10^{-8}$ |
| $^{+90}$Sr | | M | $1.2\ 10^{-7}$ | $5.4\ 10^{-8}$ | $3.7\ 10^{-8}$ |
| $^{+137}$Cs | | F | $5.4\ 10^{-9}$ | $3.7\ 10^{-9}$ | $4.6\ 10^{-9}$ |
| $^{+226}$Ra | | M | $1.1\ 10^{-5}$ | $4.94\ 10^{-6}$ | $3.5\ 10^{-6}$ |
| | $^{+210}$Pb | M | $4.0\ 10^{-6}$ | $1.3\ 10^{-6}$ | $1.2\ 10^{-6}$ |
| | $^{210}$Po | M | $1.1\ 10^{-5}$ | $4.6\ 10^{-6}$ | $3.3\ 10^{-6}$ |
| $^{232}$Th | | S | $5.0\ 10^{-5}$ | $2.6\ 10^{-5}$ | $2.5\ 10^{-5}$ |
| | $^{+228}$Ra | M | $1.0\ 10^{-5}$ | $4.7\ 10^{-6}$ | $2.6\ 10^{-6}$ |
| | $^{+228}$Th | S | $1.1\ 10^{-4}$ | $5.9\ 10^{-5}$ | $4.3\ 10^{-5}$ |
| $^{+238}$U | | M | $9.4\ 10^{-6}$ | $4.1\ 10^{-6}$ | $2.9\ 10^{-6}$ |
| | $^{234}$U | M | $1.1\ 10^{-5}$ | $4.8\ 10^{-6}$ | $3.5\ 10^{-6}$ |
| | $^{+230}$Th | S | $3.5\ 10^{-5}$ | $1.6\ 10^{-5}$ | $1.4\ 10^{-5}$ |
| $^{239}$Pu | | M | $7.7\ 10^{-5}$ | $4.8\ 10^{-5}$ | $5.0\ 10^{-5}$ |
| | $^{+235}$U | M | $1.0\ 10^{-5}$ | $4.3\ 10^{-6}$ | $3.1\ 10^{-6}$ |
| $^{241}$Am | | M | $6.9\ 10^{-5}$ | $4.0\ 10^{-5}$ | $4.2\ 10^{-5}$ |
| | $^{+237}$Np | M | $4.0\ 10^{-5}$ | $2.2\ 10^{-5}$ | $2.3\ 10^{-5}$ |

**Table B7 Ingestion rates (kg y$^{-1}$) (Smith KR and Jones AL, 2003)**

| Intakes | Adult Critical | Adult Average | Child Critical | Child Average | Infant Critical | Infant Average |
|---|---|---|---|---|---|---|
| Green vegetables | 80.0 | 30.0 | 35.0 | 10.0 | 15.0 | 5.0 |
| Soft fruit | 75.0 | 20.0 | 50.0 | 15.0 | 35.0 | 9.0 |
| Potatoes | 120.0 | 50.0 | 85.0 | 45.0 | 35.0 | 10.0 |
| Carrots | 40.0 | 10.0 | 20.0 | 6.0 | 15.0 | 5.0 |
| Milk | 240.0 | 95.0 | 240.0 | 110.0 | 320.0 | 120.0 |
| Cow meat | 45.0 | 15.0 | 30.0 | 15.0 | 10.0 | 3.0 |
| Cow offal | 10.0 | 2.8 | 5.0 | 1.5 | 2.8 | 0.5 |
| Sheep meat | 25.0 | 3.6 | 10.0 | 1.5 | 3.0 | 0.6 |
| Sheep offal | 10.0 | 2.8 | 5.0 | 1.5 | 2.8 | 0.5 |
| Freshwater fish | - | 2 | - | 0.5 | - | 0.1 |
| Drinking water | - | 600 | - | 350 | - | 260 |
| Air inhalation (m$^3$ y$^{-1}$) | - | 8100 | - | 5600 | - | 1900 |
| Soil inadvertent ingestion | $8.30 \ 10^{-3}$ | $3.70 \ 10^{-3}$ | $1.80 \ 10^{-2}$ | $1.10 \ 10^{-2}$ | $4.40 \ 10^{-2}$ | $3.70 \ 10^{-2}$ |
| Indoor occupancy | | 0.5 | | 0.7 | | 0.9 |

**Table B8 Characteristics of the generic rivers**

| River characteristics | large*, ** | medium* | small*, ** | very small stream, *** |
|---|---|---|---|---|
| **River section** | | | | |
| Length (m) | 1000 | 1000 | 1000 | 500 |
| Width (m) | 200 | 50 | 5 | 5 |
| Water depth (m) | 3 | 3 | 1 | 1 |
| Water volume (m$^3$) | 600,000 | 150,000 | 5,000 | 2,500 |
| Bed sediment depth (m) | 1 | 1 | 0.3 | 0.1 |
| Dry sediment density (kg m$^{-3}$) | 1500 | 1500 | 1500 | 1500 |
| River suspended sediment load (kg m$^{-3}$) | 0.04 | 0.04 | 0.04 | 0.04 |
| **River water flows** | | | | |
| Velocity (m s$^{-1}$) | 0.83 | 0.67 | 0.5 | 0.2 |
| Volumetric flow (m$^3$ s$^{-1}$) | 500 | 100 | 2.5 | 1 |
| **Bed sediment flow** | | | | |
| Velocity (m s$^{-1}$) | 0.0001 | 0.0001 | $3.17 \ 10^{-5}$ | $3.17 \ 10^{-5}$ |
| Volumetric flow (m$^3$ s$^{-1}$) | 0.02 | 0.005 | $4.76 \ 10^{-5}$ | $4.76 \ 10^{-5}$ |

* RP135 (Chen et al, 2003) Table 38

** The small size river is used in this assessment as the local river for non-hazardous and hazardous type landfill site, the large size rivers for inert type landfill site.

*** Small river defined in GDCs (NRPB, 2002)