Radiological Impact on the UK Population of Industries which Use or Produce Materials Containing Enhanced Levels of Naturally Occurring Radionuclides

Part II: The Steel Production Industry

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ABSTRACT

This report contains an assessment of the radiological impact on the UK population of the steel production industry within the UK. The radiological impact of the primary industry, the waste streams produced and the use of by-product slag have been considered. Individual doses from atmospheric releases from all currently operating integrated steel plants in the UK are less than 10 μSv y^{-1} for all age groups. The per caput dose rate in the UK population from 500 years of continuous steel production at the current levels is estimated to be 0.1 μSv y^{-1}. Estimated maximum doses to workers at the steel production plant, landfill workers, and workers manufacturing and using building materials containing slag were generally less than 20 μSv y^{-1}. The estimated radon concentrations in buildings constructed from concrete containing slag depend upon the radon emanation fraction assumed for the material. Experimental data in this area is sparse, and thus a range was considered. The estimated radon concentrations in buildings constructed from concrete containing slag ranged between 7.0 and 10.8 Bq m^{-3}, compared with 9.9 Bq m^{-3} when slag-free concrete is assumed. The estimated dose from radon exposure ranges between 363 μSv y^{-1} and 559 μSv y^{-1}, compared with 510 μSv y^{-1} when slag-free concrete is used. The estimated external dose to an individual in a house constructed using concrete containing slag is 790 μSv y^{-1} compared with 758 μSv y^{-1} for slag-free concrete. The overall effect of the use of the slag in building materials therefore ranges between a reduction in dose of 115 μSv y^{-1} and an increase of 81 μSv y^{-1}. Other scenarios involving exposure of members of the public to slag resulted in doses of less than 5 μSv y^{-1}. The estimated peak individual risk from landfill disposal of steel industry waste is less than approximately 1 10^{-8} y^{-1}. Currently, radiological controls on the operation of steel production sites are confined to the authorisation by the Environment Agency of atmospheric releases from the sinter plant stacks, under the terms of the Radioactive Substances Act 1993. There are no restrictions on the disposal of solid wastes or the use of slag which relate to their radionuclide content. This position is consistent with the low radiological impact of the industry as presented here. The lack of regulation is also consistent with developing EC guidance in this area.

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EXECUTIVE SUMMARY

This report considers the radiological impact on the population of the UK of the steel production industry within the UK. This report is the second in a series of reports on the radiological impact of non-nuclear industries within the UK which process or produce materials containing enhanced levels of naturally occurring radionuclides. For each industry considered the radiological impact of the primary industry, the waste streams produced and, where applicable, the use of by-products are being addressed. The study was funded by NRPB and the Environment Agency.

There are currently three integrated steel production plants in the UK, owned and operated by Corus. At the time of the study there were four such plants operating. Non-integrated steel plants in the UK, which use steel scrap only, were not considered as steel is a low activity material with insignificant radiological impact.

The raw materials used in steel production are iron ore, coal and limestone. They each contain low levels of naturally occurring radionuclides: potassium-40 and those in the uranium-238, uranium-235 and thorium-232 decay chains. The steel production process results in releases to the environment of materials with concentrations of naturally occurring radionuclides that are enhanced with respect to the raw materials used. Because the naturally occurring radionuclides concentrate in the waste streams and by-products, the steel produced contains very low levels of radioactivity and thus its use has an insignificant radiological impact.

The steel making process consists of three stages. The first two of these, sintering and iron making, are of interest regarding the enhancement of naturally occurring radionuclide concentrations. Sintering is the preparation of iron ore and coal into an iron rich porous clinker, called sinter, suitable for addition to the blast furnace. Sintering is an important part of the overall process as it reduces waste and provides an efficient raw material for iron making in the blast furnace. In the blast furnace, the iron ore and sinter are reduced and a pool of molten iron forms in the bottom of the furnace. The limestone combines with impurities forming a liquid slag, which floats on top of the metal. The third stage is conversion of iron into steel in the BOS furnace. The off-gases from all three processes are cleaned, with only a very small fraction of dust escaping to the atmosphere. The waste from the sinter plant gas cleaning system is disposed directly to landfill. Wastes, in the form of slurry, from the blast furnace and BOS furnace wet gas cleaning systems are initially de-watered in lagoons before disposal to purpose built landfills. The slag from the blast furnace and steel making processes is used in road construction and maintenance and in the manufacture of building materials.

Atmospheric releases from sinter plant stacks in the UK are authorised under the Radioactive Substances Act 1993 because the concentrations of lead and

polonium isotopes in the releases are above the exclusion level in Schedule 1 of the Act. There are no restrictions on the disposal of solid wastes or the use of slag which relate to their radionuclide content.

There are a number of potential exposure scenarios created as a result of steel production and its waste streams. These include: the exposure of members of the public to atmospheric releases from the stack, the storage lagoons and stored slag; the exposure of members of the public arising from the disposal of steel industry waste in the landfills; the exposure of members of the public from the use of slag in construction materials; the exposure of workers at the steel plants and landfill sites; and the exposure of workers who manufacture and use building materials containing slag. Individual doses and risks to members of the public and workers from these exposure scenarios have been evaluated as part of this study. Collective doses to the UK population from stack discharges and the disposal of steel industry waste to landfills, from all the integrated steel production plants in the UK, have also been determined.

The aim of this study was to quantify the radiological impact of one year's operation of the steel production industry. The year chosen was 1999, as it was the most recent year for which complete data on atmospheric releases and waste production were available at the time of the study.

NRPB has carried out previous assessments of the radiological impact of the four UK Corus steel plants. These studies considered individual and collective doses to members of the public considered to be most exposed as a result of discharges to atmosphere, and individual doses to members of the public exposed to waste disposed in a landfill site. In both of these studies doses resulting from unit releases or disposal of lead-210 and polonium-210 were evaluated. This report covers a wider scope than these earlier studies and makes use of the most recent monitoring information.

Information on the quantities of waste generated at a steel production site, the proportions released to atmosphere and disposed to landfill and the radionuclide content of the different waste streams was provided by Corus and the Environment Agency. Total quantities of waste released to atmosphere and disposed to landfill from all steel production plant in the UK were obtained from the Environment Agency. Finally, information on the radionuclide content of slag, the quantities produced and sold and its applications were obtained from various sources including Corus and the companies that buy the slag.

Individual and collective doses from stack releases from the four UK steel plants have been determined by scaling the unit release site specific doses derived in previous NRPB studies by the source terms described above. Risks to members of the public and doses to landfill workers arising from the disposal of waste to landfill were determined using a methodology previously developed by NRPB. Methodologies, assumptions and data used in the determination of doses to workers at the steel production plant and workers involved in the manufacture and use of building materials containing slag were developed for this study using information from previous, similar assessments of occupational exposures and

information obtained from Corus. The methodologies, assumptions and data used to estimate doses to members of the public from the use of slag in building materials were developed for this study using information from industry on the different uses of slag and previous, similar assessments. Doses to individuals living in homes constructed from materials containing slag from the inhalation of radon originating from radium in the building materials, were estimated using a standard approach. Doses from external irradiation were determined using a methodology proposed by the European Commission (EC).

The International Atomic Energy Agency (IAEA) has concluded that a level of dose of some tens of microsieverts a year could reasonably be regarded as trivial by regulatory authorities. IAEA also recommends the use of a 10 μSv y^{-1} dose criterion for the derivation of exemption levels. The estimated individual doses from atmospheric releases from all the currently operating integrated steel production plants in the UK are less than 10 μSv y^{-1}. These are therefore below the 'trivial' level, and also well below the dose limit to members of the public of 1000 μSv y^{-1}, and the maximum public dose constraint of 300 μSv y^{-1}.

The estimated doses to most workers at the steel production plant and workers involved in the manufacture and use of products containing recycled slag are in the range of a few, to a few tens of μSv y^{-1}, ie below the IAEA 'trivial' level. Estimated doses to workers at the blast furnace are slightly higher, 84 μSv y^{-1}. Conservative assumptions were made in determining the radionuclide content of the dust at the blast furnace and actual doses are likely to be lower.

Estimated doses to workers from the landfill disposal of wastes from the steel plant gas cleaning systems were determined for two inventories. The first, inventory I, contained only lead-210 and polonium-210 at measured levels. Measurement data for other radionuclides were not available. This inventory may, however, be optimistic for landfill disposal because radionuclides higher up the uranium-238 decay series would result in much higher doses following disposal. Therefore a second disposal inventory, inventory II, was also considered. This contained radionuclides from the uranium-238, thorium-232 and uranium-235 decay chains, with all members of the uranium-238 decay chain having the same activity concentration as that measured for lead-210, and all members of the thorium-232 and uranium-235 decay chains at levels consistent with uranium-238. The first assumption (inventory I) is expected to result in a slight underestimate of the dose that a member of the public may receive as a result of landfill disposal of waste from steel production, as some of the other radionuclides will be present, albeit perhaps at low concentrations. The other (inventory II) is extremely conservative. It is anticipated that the concentrations of these other radionuclides will be found to be closer to those in the raw materials (ie a factor of approximately 40 lower), however, this is difficult to confirm without additional measurement data. The use of inventory II therefore effectively scopes the possible range of risks. The estimated doses to landfill workers from disposal of inventory I were 11 μSv y^{-1}. Those to landfill workers dealing with inventory II were 1.7 mSv y^{-1}. The actual dose is expected to lie in the range between them, closer to that for Inventory I. If, for example,

the concentrations of the radionuclides other than lead-210 and polonium-210 were similar to those in the raw materials, then the dose would be a few tens of microsieverts per year. It is recommended that measurements of these radionuclides in landfill wastes are undertaken to refine these predictions.

In general it is expected that doses to all workers are significantly lower than 1 mSv y^{-1}, which EC guidance indicates is the dose level below which regulation is not necessary for workplaces processing materials with enhanced levels of naturally occurring radionuclides, as is reflected in current UK regulatory guidance.

The estimated doses to individuals using car parks or play areas surfaced using slag containing materials are all well below 5 μSv y^{-1}, ie below the 'trivial' level.

The estimated radon concentrations in buildings constructed from concrete containing slag depend upon the radon emanation fraction assumed for the material. Experimental data in this area is sparse, and thus a range was considered. The estimated radon concentrations in buildings constructed from concrete containing slag ranged between 7.0 and 10.8 Bq m^{-3}, compared with 9.9 Bq m^{-3} when slag-free concrete is assumed. The estimated dose from radon exposure ranged between 363 μSv y^{-1} and 560 μSv y^{-1}, compared with 510 μSv y^{-1} when slag-free concrete is used. The estimated external dose to an individual in a house constructed using concrete containing slag is 790 μSv y^{-1} compared with 758 μSv y^{-1} for slag-free concrete. The overall effect of the use of the slag in building materials therefore ranges between a reduction in dose of 115 μSv y^{-1} and an increase of 81 μSv y^{-1}.

The estimated radon concentrations in buildings constructed from materials containing slag, originating from radionuclides within the structure are within the range 7.0 to 10.8 Bq m^{-3}. EC guidance recommends that the amount of radium in building materials should be restricted at least to a level where it is unlikely that it would be a major cause for exceeding the design level for indoor radon introduced in the EC Recommendations (200 Bq m^{-3}). The estimated indoor radon activity concentrations are clearly below this level.

The estimated external dose arising from building materials is 758 μSv y^{-1} for standard building materials and 790 μSv y^{-1} when the building materials contain slag. It should be noted that the exposure scenario used in this study is conservative, assuming concrete walls, floors and ceilings, ie bulk quantities of concrete used. Subtracting a typical value for external irradiation outdoors, as used in EC (1999b), gives doses of, respectively, 460 μSv y^{-1} and 490 μSv y^{-1}. These are within the range of 0.3 mSv y^{-1} to 1 mSv y^{-1} (excess external irradiation dose to that received outdoors) within which EC guidance indicates that controls on the use of such building materials should be instituted. This conclusion is in agreement with a general evaluation produced by EC of the possibility of exceeding 0.3 mSv y^{-1} because of the use of certain building materials. This EC study concluded that it was possible that the use of concrete could result in exposures above 0.3 mSv y^{-1} almost anywhere where bulk amounts are used. The study also concluded that exposures above 1 mSv y^{-1}

from concrete were possible if bulk amounts are used and the concrete contained large amounts of blast furnace slag, fly ash or natural sand or rock rich in natural radionuclides.

The EC has recently produced guidance on exemption levels for materials containing naturally occurring radionuclides. Exemption levels were determined using a set of exposure scenarios. The scenarios included the use of materials containing naturally occurring radionuclides, such as slag, in building materials. The building material scenario adopted more realistic assumptions than those used in this study. The resulting recommended exemption levels are 0.5 Bq g^{-1} for uranium-238 and thorium-232 in secular equilibrium. Exemption levels were also derived for segments of the decay chains. The activity concentrations of the radionuclides in slag are well below these recommended exemption levels; thus providing a further illustration of the low radiological impact of the use of slag in building materials.

In order to put the above doses into context it is worthwhile noting that the average annual dose in the UK from all sources is 2.6 mSv, with a wide variation depending on the location. Most of this variation is due to differences in radon concentrations in homes. An exposure review recently conducted by NRPB on the UK population estimated that the dose due to radon accounted for up to 50% of the total dose, and that the dose received from exposure to radon had a range of 0.3 – 100 mSv y^{-1}. The majority of this dose arises from the inhalation of radon emitted from the ground beneath homes.

In the UK the acceptability of purpose built disposal facilities for radioactive waste would be judged against a risk target of 10^{-6} y^{-1} which is equal to the design target recommended by NRPB for such facilities. These criteria all relate to purpose built repositories for radioactive waste, and therefore do not necessarily apply to burial at landfill sites. However, risks below 10^{-6} y^{-1} are considered to be 'broadly acceptable', and this therefore seems a reasonable choice of criterion for judging such disposals. The estimated peak individual risk from landfill disposal of steel waste is less than approximately 10^{-8} y^{-1}, clearly below the 10^{-6} y^{-1} risk criteria.

As outlined above, currently, radiological controls on the operation of steel production sites are confined to atmospheric releases from the sinter plant stacks. There are no restrictions on the disposal of solid wastes or the use of by-products, which relate to their radionuclide content. This position seems entirely consistent with the low radiological impact of the industry as presented above.

CONTENTS

1 INTRODUCTION

This report considers the radiological impact on the population of the UK of the steel production industry within the UK. This report is the second in a series of reports on the radiological impact of non-nuclear industries within the UK which process or produce materials containing enhanced levels of naturally occurring radionuclides. The first report in the series covered coal-fired electricity generation (Smith et al, 2001). For each industry considered the radiological impact of the primary industry, the waste streams produced and, where applicable, the use of by-products will be addressed.

The raw materials used in steel production are iron ore, coal and limestone. The naturally occurring radionuclides present in these raw materials are potassium-40 and those in the uranium-238, uranium-235 and thorium-232 decay chains, see Figure 1. The steel production process results in releases to the environment of materials containing concentrations of naturally occurring radionuclides that are enhanced in relation to those of the raw materials used.

At the time of this study the Corus group operated four integrated steel production plants in the UK; at Scunthorpe, Redcar, Port Talbot and Llanwern. Llanwern has since ceased operating. These sites produce steel from iron ore; all other steel production plants in the UK produce steel by recycling scrap metal. Since radionuclides are released from the raw materials during initial processing of the iron ore, and in the production of iron, the steel that is produced contains extremely low concentrations of radionuclides. It was therefore considered unnecessary to assess the radiological impact of steel works producing steel from recycled scrap metal and this study has concentrated on the four integrated steel plants producing steel from raw materials.

Figure 2 illustrates schematically the main processes occurring at Scunthorpe steel works. These processes are common to all four integrated steel production plants. The steel making process can be broken down into three main stages. The two stages that are of main concern regarding the enhancement of concentrations of naturally occurring radionuclides are sintering and iron making. The third stage, conversion of iron into steel, also produces off-gas dust and slag, but the sintering and blast furnace processes together remove nearly all the naturally occurring radionuclides from the iron, so the radioactivity in releases from the conversion process is unlikely to be significant.

The off-gases from all three processes are cleaned, with only a very small fraction of dust (~ 0.1%) escaping to the atmosphere. The waste from the wet gas cleaning systems that is not recycled is de-watered in lagoons before final disposal to landfill. The slag from the blast furnace and steel-making processes is generally sold to companies who use it in road construction and maintenance or in the manufacture of construction materials. There are a number of potential exposure scenarios created as a result of these waste streams.

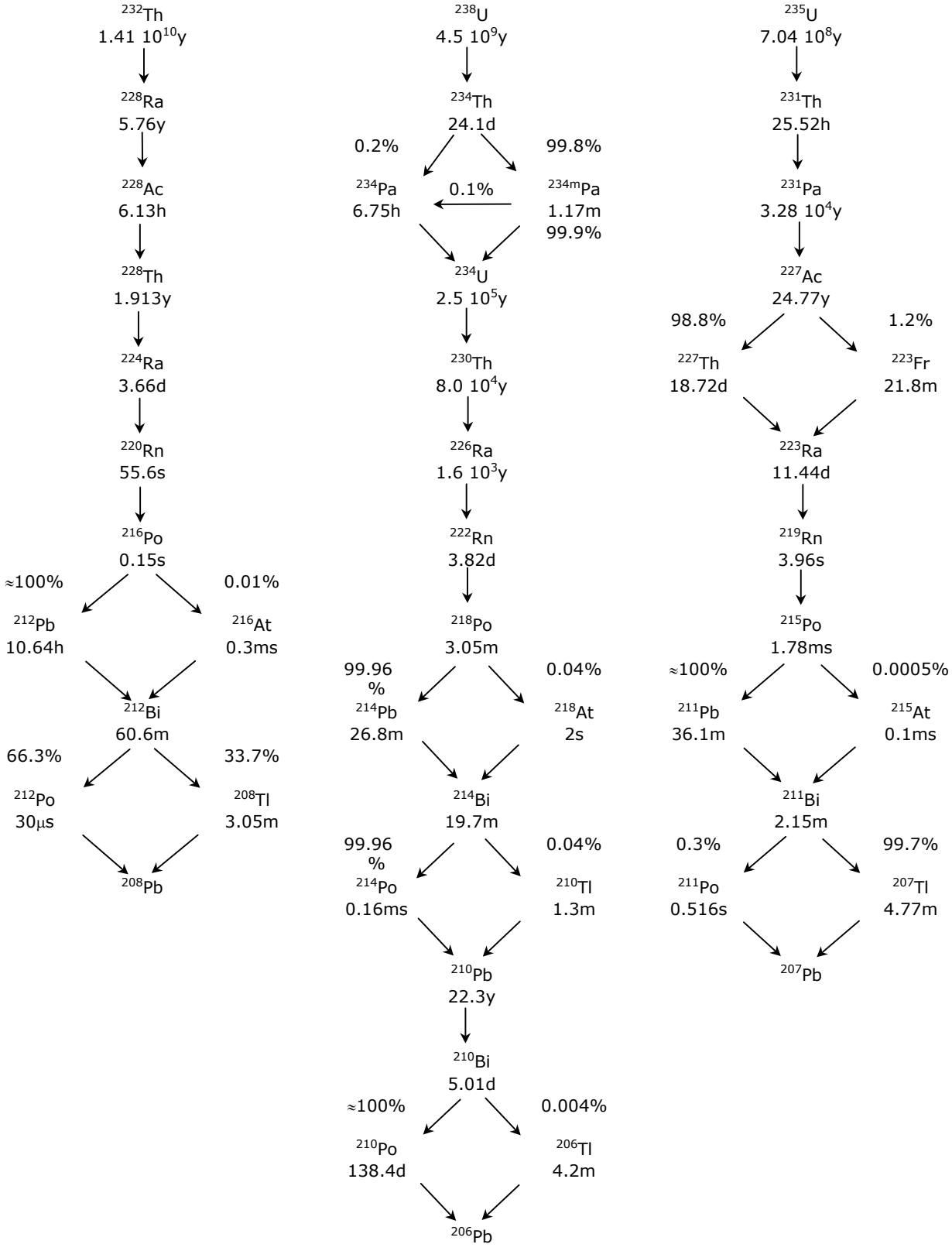
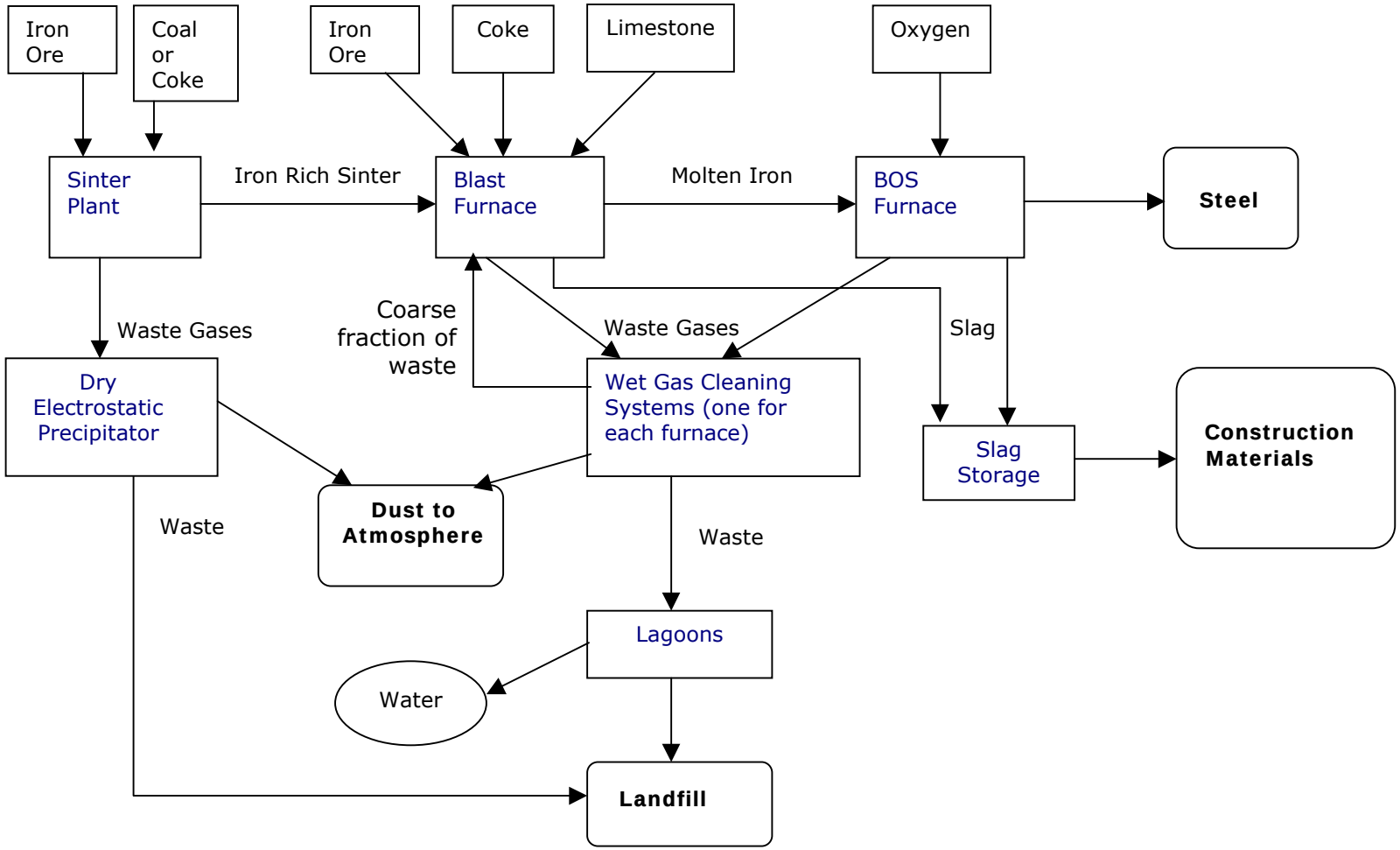


FIGURE 1 Radioactive decay chains

FIGURE 2 Process diagram for the steel industry



Members of the public may be exposed to atmospheric releases from the sinter plant, the blast furnace or the steel converter. Members of the public may also be exposed following the disposal of wastes in landfill sites. Workers at the steel plant may be exposed when undertaking a variety of activities: working at the blast furnace, transferring slag from the site to the customer, transferring waste from lagoons to on-site landfills and working at the landfill sites. Workers and members of the public may also be exposed through the use of slag in a variety of different scenarios. Individual doses to workers and members of the public have been evaluated as part of this study. Collective doses to the UK population from stack discharges and the disposal of wastes to landfill, from all the integrated steel production plants in the UK, have also been determined. A complete list of the scenarios considered and doses and risks determined is presented in Table 1.

TABLE 1 Summary of exposure scenarios

Releases to atmosphere via the stack and from slurry lagoons and slag heaps
Members of the public
Doses to individuals living in the vicinity of the steel works
Collective dose to UK population from all steel production works in the UK
Landfill disposal of wastes
Members of the public
Individual risks to typical members of hypothetical critical group
Collective doses from migration scenario
Workers
Individual doses to landfill workers
Steel plant workers
Individual doses to worker at the blast furnace
Individual doses to workers at lagoons
Individual doses to workers exposed to slag
Recycled slag
Members of the public
Individual doses from use of a car park surfaced using materials containing slag
Individual doses to children playing on a tarmac area surfaced using slag based materials
Individual doses to children playing on waste ground surfaced with slag (dusty environment)
Individual dose to an individual living in a house constructed from materials containing slag
Workers
Individual doses to workers manufacturing slag products
Individual doses to construction workers using slag products

The aim of this study was to quantify the radiological impact of one year's operation of the steel production industry. The year chosen was 1999 as it was the most recent year for which complete data on atmospheric releases and waste production were available at the time of the study.

The NRPB has carried out previous studies on the radiological impact of the four Corus sites (Mayall et al, 1997a; Mayall and Bexon, 1997). These studies

considered individual and collective doses to members of the public exposed as a result of discharges to atmosphere, and individual doses to members of the public exposed to waste disposed in a landfill site. In both of these studies doses resulting from unit releases of lead-210 and polonium-210 were evaluated. This report covers a wider scope than these earlier studies and makes use of the most recent monitoring information.

In Section 2 a brief description of the steel production industry within the UK is given. In Section 3 the source terms used in the study are discussed and defined. In Section 4 a description is given of the methodology, assumptions and data used to determine the doses and risks. The results are presented in Section 5 and the summary and conclusions in Section 6.

1.1 Regulation

The Radioactive Substances Act 1993 (RSA93) (RSA, 1993) regulates the accumulation, storage and disposal of radioactive waste, principally to control doses to members of the public. Under the provisions of RSA93 all work activities that use radioactive materials need to be registered and the accumulation and disposal of waste authorised, unless the material is specifically excluded from RSA93. RSA93 covers work involving materials containing naturally occurring radionuclides; however, materials which have concentrations of naturally occurring radionuclides (other than those involved in the nuclear fuel cycle) lower than the values given in Schedule 1 of RSA93 are not considered to be radioactive and are therefore excluded from the provisions of the Act. In addition to exclusion there are a number of Exemption Orders (EOs) made under RSA93 that exempt specific materials from certain provisions of RSA93.

In 1996, as part of a company study into releases of pollutants from its processes, British Steel (now Corus) reported to the Environment Agency that lead-210 and polonium-210 were present in airborne emissions from its sinter plants and in dust collected from electrostatic precipitators (McHugh, 1999). It became clear that the emissions of radionuclides from sinter plants would need to be authorised under the terms of RSA93 as concentrations of lead-210 and polonium-210 in the gases emitted from the sinter plants exceeded the limits for exclusion given in Schedule 1 of RSA93. British Steel therefore applied for authorisations; these were granted by the Environment Agency (EA, 1998; EA, 2000a; EA, 2000b). One of the conditions of the authorisations is that emissions should be monitored and the results reported to the Environment Agency.

The radionuclide content of other materials produced during the steel production process have also been measured. The concentration of radionuclides in the dust collected from the gas cleaning systems, although above the Schedule 1 values, are below the limit for exemption in the Phosphatic Substances, Rare Earths etc. Exemption Order (RSEO, 1962), so no authorisation for disposal is needed. Another by-product of the steel making process is slag. The levels of activity in slag have also been measured and the concentrations of the uranium-238 and

thorium-232 decay chains were found to be well below the RSA93 Schedule 1 levels and as such the material is not considered to be radioactive under the terms of RSA93.

The Ionising Radiations Regulations 1999 (IRR99) (IRR, 1999) deal primarily with regulating the doses that people receive at work. Regulation 3 (Application) of IRR99 makes it clear that the scope of the regulations includes work with radioactive substances containing naturally occurring radionuclides. The associated approved code of practice (HSE, 2000) provides more detailed guidance in this area. In the case of substances containing naturally occurring radionuclides used in work other than a practice*, eg steel plants, the regulations only apply if 'their use is likely to lead to employees or other people receiving an effective dose of ionising radiation in excess of 1 millisievert in a year'.

Further details of the legislation relating to the use of radioactive materials in the UK, and discussion of how the legislation is applied to the steel production industry in the UK is given in Appendix A. Relevant EC legislation and guidance are also discussed in Appendix A.

2 STEEL PRODUCTION IN THE UK

The stages in the production of steel from iron ore are the same at all plants in the UK, and throughout most of the world, although the quantities of steel produced and the resulting quantities of waste differ. In this report the Corus steel plant at Scunthorpe is considered as a typical plant. The first stage is sintering; this is the preparation of iron ore and coal into an iron rich porous clinker, called sinter, suitable for addition to the blast furnace. Sintering is an important part of the overall process as it reduces waste and provides an efficient raw material for iron making in the blast furnace. The process runs automatically and is controlled remotely. The waste from sintering is primarily dust in the off-gases. The gases are cleaned using dry electrostatic precipitators and then discharged to atmosphere. Dust collected from the precipitators is transferred directly to landfill; at a typical UK steel works around 1 10^6 kg y^{-1} of dust is collected and disposed. The total quantity of dust released to atmosphere from the main stack of a typical sinter plant is 3.9 10^5 kg y^{-1}. The dust in escaping gases and collected dust are both routinely analysed to determine their radionuclide content.

* In IRR99 a practice is defined as work involving the production, processing, handling, use, holding, storage, transport or disposal of radioactive substances; or the operation of any electrical equipment emitting ionising radiation and containing components operating at a potential difference of more than 5kV, which can increase the exposure of individuals to radiation from an artificial source, or from a radioactive substance containing naturally occurring radionuclides which are processed for their radioactive, fissile or fertile properties.

The second phase of the process is the production of molten iron in the blast furnace. Iron ore, coke, limestone and the iron rich clinker prepared in the sinter plant are added to the blast furnace. Here, the iron ore and sinter are reduced and a pool of molten iron forms in the bottom of the furnace. The limestone combines with impurities, forming a liquid slag, which floats on top of the metal. Iron making is a continuous process, slag is removed from the blast furnace at regular intervals and the gases are removed through pipes and enter a wet gas cleaning system. The quantity of dust escaping to atmosphere from a typical blast furnace is around 2.3 10^5 kg y^{-1}. The waste collected from the blast furnace wet gas cleaning system is separated into coarse and fine fractions. The coarse fraction, the majority of the waste, is recycled in the furnace. The fine fraction, which is in the form of a slurry, is de-watered in lagoons and is then disposed to landfill. At a typical steel production site there would be two lagoons, each with a capacity of 50 000 m^3, and these would be emptied about three times a year to landfill. Some UK Corus sites have their own landfills and others dispose to landfills off site. The landfills are fully engineered and some are clay lined.

In the third phase of the process molten iron is added to the basic oxygen furnace (BOS) where it is converted into steel. Steel making is a batch process, in which around 300 tonnes of molten iron is charged into a BOS furnace or 'converter'. A water-cooled oxygen lance is lowered into the converter and high purity oxygen is blown on to the metal at very high pressure. The oxygen combines with carbon and other impurities, thus eliminating them from the molten charge. This quantity of molten iron can be converted into steel within about 40 minutes. Gases leaving the converter are predominantly carbon monoxide (70%); after cleaning, this can be collected and used as a fuel. The waste from the BOS furnace wet gas cleaning system is in the form of a slurry and is collected and de-watered in lagoons, with the slurry from the blast furnace wet gas cleaning system, prior to disposal. The floating layer of slag, containing other oxidised impurities, is removed to a cooling pond.

All of the slag that is produced is sold. The uses of slag vary and can depend on how it was cooled (quenched). Slag may be air cooled, or sprayed with water to solidify it and then granulated. The main areas of use are in road construction and maintenance, and housing construction, i.e. manufacture of cement, concrete and insulation materials.

At a typical UK steel works the total amount of coal used in 1999 was 2.5 10^9 kg. The coal originated from Australia, Canada and the USA. The amount of iron ore used was about 6 10^9 kg, and was imported from South America, Australia, South Africa and Canada. In addition around 8.5 10^8 kg of limestone was also used, this came from the UK. These raw materials produced about 3.7 10^9 kg of liquid iron and 3.9 10^9 kg of liquid steel. The total amount of dust released to atmosphere from the operation of a typical UK steel works in 1999 was 1.4 10^6 kg. The total dry weight of dust and slurry from the sinter plant, blast furnace and steel making furnace, disposed to landfill was 5.7 10^7 kg and the total mass of slag produced by the blast furnace and steel making furnace was 1.6 10^9 kg.

2.1 Radioactivity in the steel production process

Iron ore and coal entering the sintering process contain trace amounts of naturally occurring uranium and thorium radioisotopes and their decay products. The average activity of the uranium-238 series decay products in the iron ore input to the sinter plant is 15 Bq kg^{-1} (Harvey, 1999) similar to the levels in coal (Smith et al, 2001). These levels are very low and it is only in recent times that the significance of radioactivity in the sintering process was discovered. The sintering process volatilises some minor constituents of the ore and coal, including the radioisotopes lead-210 and polonium-210, which become concentrated in the off-gases from the sintering process. As mentioned earlier the levels are such that authorisations are required for these discharges under RSA93.

Concentration of activity was also found to occur during iron-making in the blast furnace. The wet gas cleaning system removes most of the dust from the off-gases and the final emissions to atmosphere are very low in radioactivity. The emissions from the blast furnace do not require authorisation. The sintering and blast furnace processes together remove nearly all the naturally occurring radioactivity from the iron (Harvey, 1999).

The slag from both the iron and steel-making processes contains low levels of naturally occurring radionuclides from the uranium-238, uranium-235 and thorium-232 decay series. The steel that is finally produced is known as a material that contains extremely low levels of radioactivity.

3 RADIONUCLIDE CONCENTRATIONS AND SOURCE TERMS

All of the exposure scenarios described in Table 1 result from radionuclides present in the raw materials used in steel production. The raw materials originate from several different countries and the initial activity concentration in each of the materials is very low. During the stages of steel production, the concentrations of these radionuclides are enhanced to differing extents. The activity concentrations of the materials involved and the source terms that have been assumed in this study have been derived from measurements made by Corus and information on atmospheric releases from the Environment Agency, including Martin (1998) and Sandalls (1999), and are described below.

3.1 Raw materials

The main materials used in steel production are iron ore, coal and limestone. Coal contains trace quantities of uranium-238, unranium-235, thorium-232 and potassium-40 and (where relevant) their daughters. The naturally occurring radionuclides in the decay chains are likely to be in, or very close to, secular

equilibrium. Activity concentrations of the different natural decay chains can vary significantly; however, they are within the usual range of concentrations in soil and many other minerals. Reports by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) produced since 1982 (UNSCEAR, 1982, 1988) have assumed that the average concentrations of uranium-238 and thorium-232 in coal are 20 Bq kg^{-1} in both cases and that the decay products are in radioactive equilibrium with their precursor. This was based on an analysis of coal samples from 15 countries. UNSCEAR noted that although the reported concentrations ranged over two or three orders of magnitude, the averages from the various countries were in fairly good agreement. The radionuclide concentrations of iron ore are also low. The average concentration of the uranium-238 decay series in the iron ore input to a typical UK sinter plant is 15 Bq kg^{-1} (Harvey, 1999).

3.2 Atmospheric releases

3.2.1 Stack

In order to determine the radiological consequences of stack emissions from a steel production site the quantity of each radionuclide released (the source term) is required. In the sinter plant dry electrostatic precipitators are used to clean the gases leaving the sinter plant stack. The quantities of dust released to atmosphere from the sinter stack at Scunthorpe and the radionuclide concentrations (Giles and Harvey, 2000) are presented in Table 2. Waste gases from the blast furnace are extracted and taken to a wet gas cleaning system. The quantity of dust released to atmosphere from the blast furnace operations at Scunthorpe and the radionuclide concentrations in the dust (Giles and Harvey, 2000) are also presented in Table 2. The final stage of steel making is the basic oxygen furnace. As the sintering and blast furnace processes together remove nearly all of the naturally occurring radioactivity from the raw materials, releases of activity from the BOS furnace are assumed to be negligible (Harvey, 1999).

As discussed in Section 1.1, the emissions of radionuclides from the sinter stacks of each of the integrated steel production sites in the UK are authorised by the Environment Agency under RSA93 (RSA, 1993). The maximum authorised releases of lead-210 and polonium-210 from the Scunthorpe site sinter stack are also presented in Table 2.

The source term defined in Table 2 includes only lead-210 and polonium-210. The Environment Agency conducted a study to measure the activity concentrations of other radionuclides in the uranium-238 and thorium-232 decay series in sinter plant emissions. Measurements were made at Scunthorpe, Llanwern and Port Talbot. At Scunthorpe an analytical blank was included with particulate samples and filters. As there were no significant differences between the samples and the blank it was therefore assumed that the concentrations of all radionuclides except lead and polonium were insignificant at Scunthorpe. No blanks were included at either Port Talbot or Llanwern. However, in almost all cases the amounts of radionuclides on the particulate filters were below the

lower limits of detection of the radiometric analysis. It was therefore not possible to calculate the concentrations of actinium, radium, thorium or uranium. It has been assumed in this study that since the majority of members of the uranium-238 and thorium-232 decay series could not be measured that the concentrations were effectively zero.

TABLE 2 Atmospheric stack emissions from Scunthorpe*

	Sinter plant		Blast furnace		Total	
	^{210}Pb	^{210}Po	^{210}Pb	^{210}Po	^{210}Pb	^{210}Po
Activity concentration of dust (Bq kg^{-1})	1.13 10^4	9.98 10^4	8.00 10^3	2.80 10^3		
Mass of dust released (kg y^{-1})	3.90 10^5		2.26 10^5			
Activity released (Bq y^{-1})	4.41 10^9	3.89 10^{10}	1.81 10^9	6.33 10^8	**6.22 10^9** +	**3.96 10^{10}** +
Authorisation limit (Bq y^{-1})					2.9 10^{10}	5.3 10^{10}

* These are the highest measured for the four UK steel plants.

\+ Value used in assessment.

A scoping calculation was undertaken to investigate the potential significance of the inhalation of radon released from the stack. The calculation was undertaken using the total quantities of coal, iron ore and limestone used by a steel production plant in one year and assuming the activity concentrations in the raw materials were those discussed in Section 3.1. It was further assumed that the uranium-238 and thorium-232 decay chains are in secular equilibrium and that all radon present would be released from the stack. Doses to the most highly exposed group were of the order of 10^{-3} μSv y^{-1}. Even allowing for the range in activity concentrations reported by UNSCEAR, doses from this pathway are very small and have therefore not been considered further.

The source terms for the other steel plants were derived from the Scunthorpe source term. The scaling of source terms was undertaken by considering the total mass of particulates released to atmosphere from each of the plants. This information was taken from data reported to the Environment Agency (EA, 2001). The resulting source terms are presented in Table 3. The maximum authorised releases of lead-210 and polonium-210 from each of the steel production sites are also presented in Table 3.

TABLE 3 Atmospheric emissions from the stack at all UK steel works

Steel production plant	Mass of dust released to atmosphere (kg y^{-1})	Activity released to atmosphere, ^{210}Pb (Bq y^{-1})		Activity released to atmosphere, ^{210}Po (Bq y^{-1})	
Scunthorpe	$3.6\ 10^{6}$	**$6.2\ 10^{9}$** *	$2.9\ 10^{10}$ †	**$4.0\ 10^{10}$** *	$5.3\ 10^{10}$ †
Redcar	$2.8\ 10^{6}$	**$4.7\ 10^{9}$** *	$2.0\ 10^{10}$ †	**$3.0\ 10^{10}$** *	$3.6\ 10^{10}$ †
Llanwern¶	$2.9\ 10^{6}$	**$5.0\ 10^{9}$** *	$2.8\ 10^{10}$ †	**$3.2\ 10^{10}$** *	$4.9\ 10^{10}$ †
Port Talbot	$3.1\ 10^{6}$	**$5.3\ 10^{9}$** *	$3.0\ 10^{10}$ †	**$3.3\ 10^{10}$** *	$5.4\ 10^{10}$ †

* Value used in assessment.

† Authorisation limit.

¶ Llanwern ceased iron production in 2001.

3.2.2 Slurry lagoons

The waste that is collected from the blast furnace and BOS furnace wet gas cleaning systems is a wet slurry. This material is de-watered in lagoons before disposal to landfill. Atmospheric releases from the lagoons are assumed to occur through resuspension of the surface material. This is low because of the wet state of the waste. The radionuclide concentrations in the dust at the lagoons have been estimated from measurements made by Giles and Harvey (2000) of the activity concentrations in the dry dust from the lagoons when it is disposed to landfill. The activity concentrations used to assess the radiological impact of the dust lagoons are presented in Table 4. Further description of the lagoons and the method of calculating releases from them can be found in Appendix B.

TABLE 4 Activity concentrations in dust released from storage lagoons

Radionuclide	Activity concentration (Bq kg^{-1})
^{210}Pb	$3.6\ 10^{2}$
^{210}Po	$8.8\ 10^{1}$

3.2.3 Slag storage

Slag that is removed from the blast furnace and the BOS furnace is collected and stored on site until it is taken for recycling. Slag may be air cooled or granulated; depending on how it was cooled the slag has different potential end uses. Steel slag destined for use as an aggregate is stockpiled outdoors for several months to expose the material to moisture from natural precipitation and/or application of water by spraying. The purpose of such storage (ageing) is to allow potentially destructive hydration and its associated expansion to take place prior to use of the material in aggregate applications. There is a wide variation in the amount of time required for adequate exposure to the elements. Up to 18 months may be needed to hydrate the expansive oxides (TFHRC, 2000). It has therefore been assumed that all of the slag produced in one year will be stored outside in a pile, either at the steel production site or at the site of the company buying the slag. Using the total mass of slag produced in one year and the typical density of slag (TFHRC, 2000), presented in Table 5, the volume of a typical slag pile has been

estimated. The activity concentrations of the radionuclides in the uranium-238 and thorium-232 decay series in the slag have been inferred from measurements by Giles and Harvey (2000) assuming secular equilibrium. The activity concentration of uranium-235 has been inferred from the activity concentration of uranium-238 and the natural isotopic content of uranium-235 in natural uranium, 4.5% by activity. The activity concentrations of the radionuclides assumed to be present in slag are given in Table 5.

TABLE 5 Activity concentrations assumed for slag

Radionuclide	Activity concentration (Bq kg^{-1})
^{238}U	$8.8\ 10^{1}$
^{234}U	$8.8\ 10^{1}$
^{230}Th	$8.8\ 10^{1}$
^{226}Ra	$8.8\ 10^{1}$
^{232}Th	$4.9\ 10^{1}$
^{228}Ra	$4.9\ 10^{1}$
^{228}Th	$4.9\ 10^{1}$
^{235}U	$4.0\ 10^{0}$
^{231}Pa	$4.0\ 10^{0}$
^{227}Ac	$4.0\ 10^{0}$
Density of slag	1760 kg m^{-3}
Mass of slag produced	$1.6\ 10^{9}$ kg y^{-1}

The measurements of activity concentration were made on slag removed from the blast furnace; it has been assumed that the activity concentrations of radionuclides in slag from the blast furnace and BOS furnace would be the same. This assumption is conservative since most of the activity will have been removed in the blast furnace slag, hence the radionuclide concentrations in slag removed from the BOS furnace will be significantly lower. Details of the method of calculating atmospheric releases by wind driven resuspension from the slag heap are presented in Appendix B.

3.3 Disposal inventory

The dust collected from the sinter plant gas cleaning system is disposed directly to landfill. The slurry from the blast furnace and BOS furnace wet gas cleaning systems is collected, and that which is not recycled is de-watered in lagoons then sent to landfill. The majority of dust disposed to landfill originates from the blast furnace and BOS furnace gas cleaning systems.

The activity concentrations of lead-210 and polonium-210 in the collected dust have been measured. The activity concentrations, given in Table 6, are below the exemption limit given in the Phosphatic Substances, Rare Earths etc. Exemption Order (RSEO, 1962) and so no authorisation for disposal is required.

The activity concentrations of lead-210 and polonium-210 in the collected dust sent to landfill show less enhancement, with respect to those in the raw materials, than those in the dust released to atmosphere from the sinter plant and blast furnace. There are a number of reasons for this. Enhancement is most pronounced on the finest particles (these have higher surface area to volume ratios), which are those most likely to escape to atmosphere. The concentrations in the collected dust are also effectively diluted by lower activity dust from the BOS furnace wet gas cleaning system. The activity concentrations of lead-210 and polonium-210 in the waste sent to landfill are, however, still enhanced in relation to their concentrations in the raw materials.

It had been assumed that the only significant radionuclides are lead-210 and polonium-210. Therefore other radionuclides in the uranium-238 and thorium-232 natural decay chains have not been measured. This assumption may, however, be optimistic for landfill disposal because radionuclides higher up the uranium-238 decay series would result in much higher doses following disposal. For example, the dose per unit disposal from uranium-238 and all of its daughters in secular equilibrium is two orders of magnitude greater than the dose from lead-210 and all of its daughters in secular equilibrium.

In order to scope the radiological impact of landfill disposal of the wastes the assessment considered the doses and risks resulting from two inventory assumptions:

Inventory I - waste contains only lead-210 and polonium-210; and

Inventory II - waste contains radionuclides from the uranium-238, thorium-232 and uranium-235 decay chains, with all members of the uranium-238 decay chain having the same activity concentration as lead-210, and all members of the thorium-232 and uranium-235 decay chains at levels consistent with uranium-238.

The first assumption (inventory I) is expected to result in a slight underestimate of the doses that a member of the public may receive as a result of landfill disposal of waste from steel production, as some of the other radionuclides will be present, albeit perhaps at low concentrations. The other (inventory II) is extremely conservative, as it assumes enhanced levels for all the other radionuclides. It is anticipated that the concentrations of these other radionuclides will be found to be closer to those in the raw materials (ie a factor of approximately 40 lower), however, this is difficult to confirm without additional measurement data. The use of inventory II therefore effectively scopes the possible range of risks. The activity concentrations used in the study to assess both options are given in Table 6.

Individual doses to workers and members of the public have been calculated using data from the steel production site disposing of the largest quantities of material to landfill. In order to calculate collective doses to the UK population from landfill disposal of the waste from steel production, the total quantity of waste disposed from all steel production sites is required. The quantities of waste assumed in the assessment to be disposed from each of the steel production

sites are given in Table 7. Details of the assessment methodology are given in Appendix C.

TABLE 6 Activity concentrations for all landfill disposal scenarios

Radionuclide	Inventory I Measured activity concentrations (Bq kg^{-1})	Inventory II Equilibrium with measured activity concentrations (Bq kg^{-1})
U+238		$9.00\ 10^{2}$
U-234		$9.00\ 10^{2}$
Th-230		$9.00\ 10^{2}$
Ra+226		$9.00\ 10^{2}$
Pb+210	$9.00\ 10^{2}$	$9.00\ 10^{2}$
Th-232		$4.50\ 10^{2}$
Ra+228		$4.50\ 10^{2}$
Th+228		$4.50\ 10^{2}$
U+235		$4.05\ 10^{1}$
Pa-231		$4.05\ 10^{1}$
Ac+227		$4.05\ 10^{1}$

Notes

U+238 indicates ^{238}U in secular equilibrium with ^{234}Th, $(0.998)^{234m}$Pa and $(3.3\ 10^{-3})^{234}$Pa

Ra+226 indicates ^{226}Ra in secular equilibrium with ^{222}Rn, ^{218}Po, $(2\ 10^{-4})^{218}$At, ^{214}Pb, ^{214}Bi and ^{214}Po

Pb+210 indicates ^{210}Pb in secular equilibrium with ^{210}Bi and ^{210}Po

Ra+228 indicates ^{228}Ra in secular equilibrium with ^{228}Ac

Th+228 indicates ^{228}Th in secular equilibrium with ^{224}Ra, ^{220}Rn, ^{216}Po, ^{212}Pb, ^{212}Bi, $(0.641)^{212}$Po and $(0.359)^{208}$Tl

U+235 indicates ^{235}U in secular equilibrium with ^{231}Th

Ac+227 indicates ^{227}Ac in secular equilibrium with $(0.986)^{227}$Th, $(1.4\ 10^{-2})^{223}$Fr, ^{223}Ra, ^{219}Rn, ^{215}Po, ^{211}Pb, ^{211}Bi, $(2.8\ 10^{-3})$ ^{211}Po and (0.997) ^{207}Tl

TABLE 7 Landfill disposal quantities from all UK steel works

Steel production site	Quantity of dust disposed (kg y^{-1})
Scunthorpe	$5.7\ 10^{7}$
Redcar	$4.4\ 10^{7}$
Llanwern*	$4.6\ 10^{7}$
Port Talbot	$4.9\ 10^{7}$
Total	$2.0\ 10^{8}$

*Llanwern ceased iron production in 2001

3.4 Dust in the working environment

In addition to the activity concentrations in processed materials, which were obtained from Corus (Harvey, 1999; Giles and Harvey, 2000) and have been

discussed in the previous sections, the concentration of radionuclides in resuspended dust in the working environment is required to estimate the radiological impact on steel plant workers. In the absence of experimental data it has been assumed that the concentrations of lead-210 and polonium-210 in the dust in the working area around the blast furnace are the same as those in the furnace off gases. This is a conservative estimate as workers in the area will primarily be exposed to dust from the metal and slag. It was further assumed that, as these workers are also exposed to slag piles, the remaining members of the uranium-238, uranium-235 and thorium-232 decay chains are present with the activity concentrations measured in slag. The resuspended dust concentration at the blast furnace was measured by Giles and Harvey (2000), dust concentrations at the lagoons and slag heaps have been estimated based on data for industrial areas taken from Simmonds et al (1995). A summary of resuspended dust concentrations in relevant areas of the steel production plant, and the activity concentrations assumed to be in the dust are given in Table 8.

TABLE 8 Activity concentration in dust and resuspended dust concentration for each steel plant work area

Worker location	Dust concentration in air (kg m^{-3})	Activity concentration of dust (Bq kg^{-1})	
Blast furnace	$7.5\ 10^{-7}$ *	^{210}Pb	$8\ 10^{3}$ †
		^{210}Po	$2.8\ 10^{3}$ †
		^{238}U ‡	88 *
		^{232}Th §	49 *
		^{235}U §	4 *
Slurry lagoons	$1.0\ 10^{-7}$ ¶	^{210}Pb	360 *
		^{210}Po	88 *
Digging cooled slag	$4.0\ 10^{-7}$ ¶	^{238}U	88 *
		^{232}Th	49 *
		^{235}U	4 *

* Giles and Harvey (2000).

† Harvey (1999).

‡ Including all daughters, except ^{210}Pb, ^{210}Bi and ^{210}Po, in secular equilibrium.

§ Including all daughters in secular equilibrium.

¶ Simmonds et al (1995).

4 METHODOLOGY

The methodologies used to determine the radiological consequences of the exposure scenarios listed in Table 1 are described in this section. The assumptions made and data used are also defined.

4.1 Atmospheric releases

There are three scenarios that result in the release of radionuclides to atmosphere from steel production sites: releases of dust from the stacks, releases of dust from slurry lagoons and releases of slag dust from slag piles. Individual doses from each have been assessed separately as, given the size of the site, it is very unlikely that an individual would be exposed to all three sources of atmospheric emissions. The methodologies, data and assumptions used to determine doses resulting from the dispersion of dust and slag to atmosphere are described below. The assumptions made about the activity concentrations in the dust and slag were discussed in Section 3.

4.1.1 Stack releases

Radionuclides are discharged into the environment via the stack of the sinter plant and the blast furnace. The heights of the stacks at the four integrated steel production plants in the UK are between 76.2m and 133m (Mayall et al, 1997a). Doses to members of the public were assumed to result from five exposure pathways:

- inhalation of radionuclides in the plume;
- external irradiation by radionuclides in the plume;
- inhalation of radionuclides resuspended from a surface deposit;
- external irradiation by deposited radionuclides; and
- ingestion of food grown on land contaminated by a deposit of radionuclides.

Individual and collective doses from stack releases from the four UK steel plants have been determined by scaling the unit release critical group and collective doses derived in previous NRPB studies (Mayall et al, 1997a; Mayall and Bexon, 1997) by the source terms in Table 3. These earlier NRPB studies used detailed site specific information on the location of homes and farms around each of the steel plants to identify the location of the critical group and potential sources of locally produced foods, and the type of foods produced. Actual stack heights and details of mass release rates and temperatures of the release were used. Site specific meteorological data was also used for each site.

Generic calculations

There was also a need as part of this study to determine generic doses from atmospheric releases from a 'typical' steel plant, to allow direct comparison with those from other UK industries that are being considered as part of the overall study of the radiological impact of such industries in the UK. These generic calculations are undertaken using the same set of defined conservative assumptions for each industry. For this study they have been undertaken using the source term for Scunthorpe, Table 2.

For these generic calculations two exposed groups are considered: a group intended to represent the typical exposure of local people in the area, and a

group of people who are assumed to be located at a place, and have habits such that they are likely to receive high doses from the plant. For the typical group it was assumed that they live 5 km from the release point, and remain at that location for the entire year. During this time individuals of all age groups were assumed to be outside for 10% and inside for 90% of the year (Robinson, 1996). They were assumed to eat all foodstuffs at an average rate and obtain 25% from the local area except for green vegetables, where there is evidence that 50% is locally sourced (Simmonds et al, 1993). The assumptions used to characterise the high exposure group were based on those described in Robinson et al (1994). It is assumed that they live on a farm 500 m from the release point. It has further been assumed that they spend all their time at the location, adults spend 50% of the time outside and children and infants spend 20% of their time outside. They obtain all their food from local sources, eating the two most significant foodstuffs at a critical rate and others at average rates. Doses to adults, children and infants were determined. The characteristics of these two groups are described in more detail in Appendix B and summarised in Table B1. It should be stressed that the assumptions made for the two groups are not site specific but are generic assumptions applicable to the UK. The estimated doses will therefore differ from those that would be determined using site-specific information on locations of habitation and agricultural practices around a site. They are intended simply to facilitate comparison of radiological impacts with other industries.

For these generic calculations an effective release height of 100m was assumed. To assess doses from these pathways, the radionuclide concentrations in air, deposition rates and external doses from gamma irradiation from the cloud were calculated. This was done using the atmospheric dispersion model PLUME (part of the PC CREAM (Mayall et al, 1997b) suite of models). The doses arising from deposited radionuclides have been integrated over 50 years as this is equivalent to the highest annual dose that could be received from a plant that operates for 50 years. Integrated dose rates per unit deposit were calculated for inhalation of resuspended radionuclides, external irradiation and ingestion of food using parts of the PC CREAM suite of models: RESUS, GRANIS, and the NRPB terrestrial foodchain model FARMLAND (Brown and Simmonds, 1995). A full description of the generic dose calculations and the assumptions made is given in Appendix B.

4.1.2 Exposures from slurry lagoons

Dust is released from the lagoons through wind-driven resuspension. Doses to members of the public are assumed to result from five exposure pathways:

- inhalation of radionuclides in the plume;
- external irradiation by radionuclides in the plume;
- inhalation of radionuclides resuspended from a surface deposit;
- external irradiation by deposited radionuclides; and
- ingestion of food grown in land contaminated by a deposit of radionuclides.

No site specific dose calculations have been undertaken for releases from slurry lagoons. Instead the approach described above for the determination of generic doses has been used to provide some indication of the radiological consequences of these releases. The models described above were used to calculate doses from releases of dust to atmosphere. For the purposes of determining air concentrations, deposition rates and external irradiation from the cloud, the releases were assumed to be at ground level. A full description of these generic dose calculations and the assumptions made is given in Appendix B.

4.1.3 Exposures from slag storage

Slag dust is released from large storage piles of slag by wind-driven resuspension. Doses to members of the public from slag heaps were assumed to result from six exposure pathways:

- inhalation of radionuclides in the plume;
- external irradiation by radionuclides in the plume;
- inhalation of radionuclides resuspended from a surface deposit;
- external irradiation by deposited radionuclides;
- ingestion of food grown in land contaminated by a deposit of radionuclides; and
- direct external irradiation by the slag heap.

No site specific dose calculations have been undertaken for releases from slurry lagoons. Instead the approach described above for the determination of generic doses has been used to provide some indication of the radiological consequences of these releases. The models described were used to calculate doses from releases of dust to atmosphere. For the purposes of determining air concentrations, deposition rates and external irradiation from the cloud, the releases were assumed to be at ground level. In addition to doses resulting from the release of dust to atmosphere, the dose from direct irradiation from slag was also considered. The code QAD-CG (Cain, 1997) was used to calculate these doses. A full description of these generic dose calculations and the assumptions made is given in Appendix B.

4.2 Landfill disposal

4.2.1 Public

Following the disposal of radioactive material in the ground there are two main scenarios in which exposure to people could occur. The most likely occurrence (migration scenario) is the gradual migration of radionuclides with ground water from the waste through the surrounding rock and soil (geosphere) into the local environment (biosphere). From here, people could be exposed via a wide variety of routes including direct external irradiation from contaminated soil and the consumption of contaminated food. This type of exposure has a probability of occurrence close to unity, although there may be some uncertainty in the precise magnitude and time of occurrence of the doses. The second way (probabilistic

scenarios) involves events that disturb the natural evolution of the site, and has a lower probability of occurrence. One example is excavation of the site for development.

Since these exposures would arise in the future, predictive mathematical modelling is required to estimate the doses and risks to those people most likely to be exposed. Where there is no specific characterisation of the landfill site it is common practice to make generic assumptions about the site to determine parameter values for modelling. These assumptions are based on past experience of similar situations.

For this study a general methodology developed by NRPB (not yet published) for the assessment of risks from the disposal of radionuclides in landfill sites was used. This considered the following scenarios for exposure of members of the public:

- migration;
- borehole water extraction;
- excavation during development of the site; and
- residence on the site.

As part of the development of the NRPB landfill methodology a review of landfill disposal sites in the UK was carried out to determine suitable characteristics for a set of representative generic site types to be defined. An extensive review of landfills currently in use was undertaken, with emphasis on a small number of key parameters found in previous work to have the most influence on the risks from disposal. The aim was to characterise, as well as possible, the general types of landfill operational in the UK. The particular geology, dimensions and level of containment of around 50 landfills were compared, and four broad categories identified. These are described in Appendix C.

The principal output from the landfill study was the production of a set of doses and risks from the above scenarios for unit disposals of a large number of radionuclides to each of these four generic landfill types. For the study, modelling of the migration of radionuclides in the geosphere was performed using GEOS (Hill, 1989). This is a simple one-dimensional geosphere migration code, developed by NRPB, which models the transport of radionuclides with ground water, taking account of advection, dispersion/diffusion, radioactive decay and sorption. Transfer in the biosphere was modelled using BIOS (Martin et al, 1991), a compartmental model developed at NRPB, that represents the transfer of radionuclides through deep and surface soils, rivers and seas, and provides estimates of doses for the exposure pathways outlined above. Individual doses and risks per unit disposal from all the above scenarios were determined. Collective doses per unit disposal for the migration scenario were also evaluated.

The unit disposal results were used in this study to provide an estimate of the overall doses and risks associated with the disposal of steel plant gas cleaning system wastes. There were three main steps in this process. The first involved identifying the disposal inventory, and this was discussed in Section 3.3. The

second step was to identify which of the four generic landfill site types is most similar to sites where disposal of such wastes would take place. This involved a brief review of the characteristics of landfill sites currently used in the UK by Corus. Landfill sites used for the disposal of waste from the steel production process are generally purpose built and fully engineered (Giles and Harvey, 2000). When the landfill sites are full they will be capped with clay and topsoil, limited landscaping will take place and then they will be returned to heathland. On the basis of this information it was decided that the generic landfill site that most closely represented the general characteristics of the steel waste landfill disposal sites was 'Type D'. The final step was to scale the unit disposal results for the appropriate landfill site type according to the inventory and then sum over radionuclides to produce total doses and risks. This process is described in detail in Appendix C.

4.2.2 Landfill workers

The NRPB landfill study discussed above also involved the development of a methodology and identification of associated data for the assessment of doses to landfill workers. It was assumed that the landfill operator distributes waste within the site using mechanical excavators. The methodology assumed landfill workers would be exposed during a working year (2000 hours per year) via the following exposure pathways: external irradiation, inhalation of contaminated dust, contamination of the skin and inadvertent ingestion of dust. The NRPB landfill methodology study involved the determination of doses per unit disposal to landfill workers. Information obtained from Giles and Harvey (2000) indicated that workers associated with landfill disposal of wastes at a typical steel works would be exposed for a full working year. Doses to landfill workers were therefore determined by using the unit disposal results from the landfill methodology study scaled according to the inventory of radionuclide concentrations in the waste. A full description of the dose calculations and the assumptions made is given in Appendix C.

4.3 Production and use of construction materials containing recycled slag

All slag that is produced as a by-product during steel production is sold. The main areas of use for this slag are in road construction and maintenance, and housing construction. It is used, for example, in the manufacture of bricks, cement, concrete and insulating materials.

The composition and form of all road surfaces are covered by British Standards. For example, the composition of 'dense tar' road surfacing is covered by BS 5273 (BSI, 1990) and the specification of constituent materials and asphalt mixtures for 'hot rolled asphalt' for roads and other paved areas is given in BS 594 (BSI, 1992a). There are a large number of different types of road surfaces, but those mentioned above are the most widely used. The main constituent materials of road surfaces are coarse aggregate, binder (a mixture of fine aggregate and tar),

filler (also fine aggregate) and chippings for application to the surface of the wearing course (made from coarse aggregate). The proportions of these materials can be varied depending on the type of road surface required, however, slag from the steel production process could form any or all of the materials, with the exception of tar. The British Standards mentioned above (BSI, 1990, 1992a) indicate that tar comprises approximately 10% of the surfacing material. It is therefore possible that road surfaces could have a slag content of up to 90%, and this conservative assumption has been made in this study.

Portland blast furnace cements are manufactured using ordinary Portland cement and slag that meets the requirements of BS 6699 (BSI, 1992b). The proportions of ordinary Portland cement and selected granulated blast furnace slag are varied to achieve products complying with BS 146 (BSI, 1996a) or BS 4246 (BSI, 1996b). Portland blast furnace cements can be used in all normal applications for which ordinary Portland cement is suitable, from lean mix to high strength structural concretes. Portland blast furnace cement containing up to 35% slag may be used in a wide range of mortars. Concrete with a slag content in excess of 35% may have a lower early strength and greater care is required with curing and in cold weather. Portland blast furnace cements give specific benefits, in reduced heat of hydration and improved chemical resistance. The reduced heat evolution in concrete can be utilised to reduce thermal stresses and ultimately, cracking in large concrete pours. Ground granulated blast furnace slag has also been shown to reduce the rate of diffusion of chlorides into concrete. The quantity of slag used in cement and concrete is usually up to 35%, but can be as much as 55% (Castlecement, 2000). Cement containing more than 55% blast furnace slag can only be used in special applications. For this study it has been assumed that cement and concrete contain 35% blast furnace slag.

The use of slag in roads and building materials will lead to the exposure of a number of groups. The three most significant, and those considered in this study, are:

- workers manufacturing road materials and building products containing slag;
- workers constructing buildings and roads using these products; and
- members of the public exposed in car parks, playgrounds and houses constructed from materials containing slag.

The methodologies, assumptions and data used to determine doses to each of the groups are described below.

Several sets of doses were considered in the study. The first set comprises doses arising from the radionuclides present in the slag only. It is important to note, however, that building materials other than those containing slag also contain naturally occurring radionuclides, which result in exposures of workers and members of the public. Similarly, it must be remembered that slag replaces other constituents that would themselves contain naturally occurring radionuclides and thus give rise to radiation exposures. In order to build up a

picture of the overall radiological significance of the use of slag in building materials it is necessary to generate two further sets of doses. One set consisting of doses from all the radionuclides present in the building materials (i.e. not simply those originating in the slag). The other comprising doses from the use of similar building materials that do not contain slag.

4.3.1 Manufacturing workers

The doses received by individual workers at plants manufacturing road construction and building materials will clearly vary substantially depending in detail on their work activities, with the majority receiving trivial doses. The aim of this part of the study was to determine doses typical of those received by the most exposed workers. These workers would typically be those directly involved in the management of slag; in particular those involved in the distribution of slag into the manufacturing process. It was assumed that workers were exposed by the following pathways:

- inhalation of slag dust in the workplace;
- inadvertent ingestion of slag dust in the workplace;
- external irradiation from contamination of exposed skin areas; and
- external irradiation from piles of stored slag and building materials.

The methodology used for the determination of doses to these workers was developed from that used in previous studies (Smith et al, 2001; Penfold et al, 1997; Harvey et al, 1998). The methodology and assumptions are described in detail in Appendix D.

4.3.2 Construction workers

It was assumed that construction workers would be exposed to similar pathways as manufacturing workers, namely:

- inhalation of slag dust in the workplace;
- inadvertent ingestion of slag dust in the workplace;
- external irradiation from contamination of exposed skin areas; and
- external irradiation from piles of stored slag and building materials.

The methodology used for the determination of doses to these workers was also developed from that used in previous studies (Smith et al, 2001; Penfold et al, 1997; Harvey et al, 1998). The methodology and assumptions are described in detail in Appendix D.

4.3.3 Members of the public

A number of scenarios have been considered for the exposure of members of the public. These scenarios are intended to represent realistic situations where members of the public would be most exposed to slag. The first scenario considers members of the public exposed as a result of the use of a car park

surfaced using tarmac containing slag; adults using the car park for short periods of time were considered, it was assumed that there would be no parking attendants. The second scenario considers children using a play area surfaced using tarmac containing slag, a more conservative scenario of children playing on an area of dusty waste ground surfaced with slag was also considered. This is an unlikely scenario but was chosen to represent a hypothetical dusty area to explore potential doses from other exposure pathways. Finally, members of the public living in a house built using materials containing slag were also considered.

The exposure of members of the public resulting from radionuclides in building materials has been assumed to be via two main pathways; external exposure and inhalation of radon emanating from the walls. The only pathway considered to be significant for members of the public using the car park or a tarmac play area was external irradiation from the surface of the tarmac. However, children playing on waste ground surfaced using slag could be exposed via a number of different pathways, including external exposure from the ground, inhalation of dust and inadvertent ingestion of dust. A full description of the methodology and assumptions used to calculate doses from these exposure scenarios is given in Appendix D.

4.4 Steel production plant workers

The doses received by individual steel production plant workers will vary substantially depending in detail on their work activities, with the majority receiving trivial doses. The aim of this part of the study was to determine doses typical of those received by the most exposed workers. Three groups were considered: workers at the blast furnace, workers at the lagoons containing slurry from the wet gas cleaning system, and workers involved in the transfer of slag from production to storage areas. Each of these three groups has been considered separately because the size of a steel production site means that it is unlikely that workers would be exposed in more than one area of the site. It is also known (Giles and Harvey, 2000) that steel production workers have specific jobs and are unlikely to spend a significant amount of time in more than one area of the site. The three groups of workers were assumed to be exposed via four pathways:

- external exposure to dust, slurry or slag;
- inhalation of dust in the workplace;
- inadvertent ingestion of dust in the workplace; and
- external irradiation from contamination of exposed skin areas.

The total dose to a worker in each of the groups is then the sum of the doses from these four pathways. The methodology and assumptions used to determine doses to these workers are given in detail in Appendix E.

5 RESULTS

5.1 Individual doses and risks

The estimated individual doses and risks from all of the exposure scenarios discussed in the previous sections are presented in Tables 9–12. Detailed descriptions of the dominant pathways and radionuclides are given in the relevant Appendices.

The estimated annual individual doses from stack releases from each of the four steel plants are presented in Table 9. These were determined by scaling the site specific unit release calculations undertaken previously by NRPB by the source terms in Table 3. The highest dose is from Llanwern, which ceased iron production in 2001. The doses from the other plants are all less than 10 μSv y^{-1}. The age group receiving the highest doses are infants and the dominant pathway is ingestion of locally produced food.

Generic dose calculations were also undertaken for stack releases. These were undertaken using generic conservative assumptions and the estimated doses are, as expected, higher than those presented in Table 9. Detailed results from the generic calculations of doses from stack releases are presented in Appendix B.

The above results were determined on the basis of the source terms in Table 3. If the steel production sites were to release lead-210 and polonium-210 up to their maximum authorised limits, the estimated doses would increase slightly. The maximum for the currently operating sites would be 17 μSv y^{-1} for Port Talbot.

No site specific dose calculations have been undertaken for releases from slurry lagoons. Instead the results of the generic dose calculations described in Appendix B have been used to provide some indication of the radiological consequences of these releases. These are based on the activity concentrations in Table 4. The estimates of doses from releases from slurry lagoons are less than 0.01 μSv y^{-1}. See Appendix B for more details of the results. Given the conservative nature of the assumptions used in the generic calculations it is expected that in reality the doses are lower.

Similarly, no site specific dose calculations have been undertaken for releases from slag piles. Instead, as above, the results of the generic dose calculations described in Appendix B have been used to provide some indication of the radiological consequences of these releases. These are based on the activity concentrations in Table 5. The estimated doses from the generic calculations are less than 7 μSv y^{-1}. See Appendix B for more details of the results. Given the conservative nature of the assumptions used in the generic calculations it is expected that in reality the doses are lower.

TABLE 9 Annual individual doses from stack releases

	Individual effective dose ($\mu Sv\ y^{-1}$)	Dominant age group and exposure pathway
Scunthorpe	5.8	Infant, food
Redcar	3.5	Infant, food
Llanwern¶	30.6	Infant, food
Port Talbot	9.4	Infant, food

¶ Llanwern ceased iron production in 2001

The estimated doses and risks from landfill disposal of inventory I and inventory II (see Section 3.3) are presented in Table 10. The annual individual dose to the public from inventory I is 140 $\mu Sv\ y^{-1}$ from residence on a redeveloped site. With the associated probability of occurrence this gives an individual risk of $5.2\ 10^{-9}\ y^{-1}$. The estimated doses and risks from the migration and ingestion of well water scenarios are effectively zero because of the long time taken for radionuclides to migrate to the biosphere in comparison with the relatively short half lives of lead-210 and polonium-210.

The estimated annual dose to a member of the public from the disposal of inventory II is 380 $\mu Sv\ y^{-1}$, also from residence on a redeveloped site. With the associated probability of occurrence, this gives a risk from this scenario of $1.4\ 10^{-8}\ y^{-1}$. The migration scenario gives an estimated individual dose of 0.25 $\mu Sv\ y^{-1}$, and a corresponding risk of $1.5\ 10^{-8}\ y^{-1}$. For migration, the peak risk arises between 10^5 and 10^6 years following disposal. It is generally considered that calculations of individual risk beyond about 10,000 years can only provide an *indication* of the possible level of risk rather than a *prediction* of the risk (NRPB, 1992). From an examination of the results of the landfill study, it is anticipated that the migration risks to 10,000 years would be several orders of magnitude lower. Individual risks to 10,000 years are therefore dominated by those from the residence scenario. As was discussed earlier, inventory I is expected to result in a slight underestimate of doses and risks. Inventory II, however, was derived using conservative assumptions to scope the possible range of risks. The actual peak risk is therefore expected to be within the range of the two predictions, closer to that of Inventory I. The difference between the two predictions is, however, only a factor of 2 or 3.

The estimated doses to landfill workers are 11 $\mu Sv\ y^{-1}$ and 1.7 $mSv\ y^{-1}$, respectively, for Inventory I and Inventory II. As was discussed earlier, Inventory I is expected to result in a slight underestimate of doses. Inventory II, however, was derived using very conservative assumptions. The actual dose is expected to lie in the range between them, closer to that for Inventory I. If, for example, the concentrations of the radionuclides other than lead-210 and polonium-210 were similar to those in the raw materials, then the dose would be a few tens of microsieverts per year. It is recommended that measurements of these radionuclides are undertaken to refine these predictions.

TABLE 10 Annual individual doses and risks from landfill disposal of waste

	Peak individual effective dose ($\mu Sv\ y^{-1}$)	Peak individual risk (y^{-1})	Dominant exposure pathways
Inventory I			
Public	$1.4\ 10^{2}$	$5.2\ 10^{-9}$	Residence (dose and risk)
Worker	$1.1\ 10^{1}$	$6.3\ 10^{-7}$	Inadvertent ingestion
Inventory II			
Public	$3.8\ 10^{2}$	$1.5\ 10^{-8}$	Residence (dose); Migration (risk)
Worker	$1.7\ 10^{3}$	$1.0\ 10^{-4}$	External irradiation

The estimated individual doses from all situations where it was considered that exposure to slag might occur are presented in Table 11. The estimated annual dose to workers manufacturing aggregates for use in roads is 18 $\mu Sv\ y^{-1}$, and the dose to workers manufacturing building materials is 6 $\mu Sv\ y^{-1}$. These doses are the result of exposure from radionuclides in the slag only. The difference in the doses from the two scenarios arises from the quantity of slag used in the different types of materials; it was assumed that road building materials could be made almost entirely from slag.

Doses to workers manufacturing building materials, from the radionuclides present in ordinary cement and concrete (ie non slag containing) have also been estimated, these calculations are described in detail in Appendix D. The estimated excess dose to manufacturing workers from the use of slag in building materials is 2 $\mu Sv\ y^{-1}$. The excess dose from the use of slag in the manufacture of road building materials has not been assessed because there is little data available on the natural activity concentrations of road aggregates. However, it can be assumed that there will be some naturally occurring radionuclide content and the excess dose resulting from the use of slag as a replacement aggregate will therefore be less than 18 $\mu Sv\ y^{-1}$.

The estimated annual dose to workers constructing roads using slag aggregates is 19 $\mu Sv\ y^{-1}$. This dose has also been estimated assuming exposure from radionuclides in the slag only. As discussed above the excess dose resulting from the use of slag as a replacement for other road building materials will be less than 19 $\mu Sv\ y^{-1}$, because ordinary materials used to build roads will inevitably have some naturally occurring radionuclide content. Doses to workers using materials containing slag to build houses were not considered because the fraction of slag used in housing materials is much less than the fraction used in road materials, and thus the doses will be lower.

TABLE 11 Annual individual doses from slag use scenarios

	Individual effective dose* (μSv y^{-1})	Dominant exposure pathways (% contribution to total dose in brackets)
Workers		
Manufacturing aggregates for roads	1.8 10^{1}	Inhalation (88%)
Manufacturing building materials	6.3 10^{0}	Inhalation (88%)
Construction of roads	1.9 10^{1}	Inhalation (84%)
Members of the public		
Car park adult	6.3 10^{-2}	External (100%)
Tarmac play area 10 year old	1.7 10^{-1}	External (100%)
Waste ground play area 10 year old	4.5 10^{0}	Inhalation (82%)
Residents adults	8.1 10^{1}§	Radon inhalation (60%)
	1.2 10^{2}†	-

* Doses from radionuclides in the slag only, except as indicated below.

§ Additional dose from the use of slag in building materials, assuming radon emanation fraction for slag of 5%.

† Reduction in dose from the use of slag in building material, assuming radon emanation fraction for slag of 0.7%.

TABLE 12 Peak annual individual doses to workers at the steel plant

Exposure scenario	Exposed group	Peak individual effective dose (μSv y^{-1})	Dominant exposure pathways (% contribution to total dose in brackets)
Blast furnace	Worker	83.7	Inadvertent ingestion (74%)
Dust lagoons	Worker	2.8	Inadvertent ingestion (95%)
Transferral of slag	Worker	5.7	Inhalation (55%)

Doses to members of the public from the use of slag are also presented in Table 11. The estimated dose to an adult member of the public using a car park surfaced using slag based tarmac is 0.06 μSv y^{-1}, and the estimated dose to a child playing on a playground surfaced with the same material is 0.17 μSv y^{-1}. The conservative, hypothetical scenario of children playing on an area of dusty waste ground surfaced entirely with crushed slag, with no tar used to bind the surface, gives rise to doses of 4.5 μSv y^{-1}. These are the doses resulting from radionuclides in the slag only.

The effect of using building materials containing slag varied with the assumptions made, with a maximum increase in the total dose to a resident of 81 μSv y^{-1}. This increase in dose is predominantly caused by inhalation of radon. Radon is 'exhaled' from building materials into the indoor air. The radon exhalation rate depends primarily on the radionuclide content of the material and the radon emanation factor. The radionuclide content of the building materials and the emanation fractions assumed are discussed fully in Appendix D. The estimated concentration of radon from exhalation from building materials in a

typical house constructed using 'standard' (ie slag-free) building materials is 9.9 Bq m^{-3}, and the estimated concentrations in a house constructed using materials containing 35% slag are 7 Bq m^{-3} and 10.8 Bq m^{-3}, when emanation fractions for slag of 0.7% and 5%, respectively, are assumed. This results in doses from the inhalation of radon ranging from a decrease of 147 μSv y^{-1} to an increase of 49 μSv y^{-1}, compared with those in homes constructed using slag-free building materials. The dose resulting from direct external irradiation from ordinary building materials is increased from 758 μSv y^{-1} to 790 μSv y^{-1} when the building materials contain 35% slag; an increase of 32 μSv y^{-1}. Further discussion of the estimated doses to residents from radon inhalation and direct external irradiation is given in Appendix D.

It is important to note that in the investigation described above into the radiological significance of the use of slag in building materials, the assumed concentrations of radionuclides in building materials are based on those for modern composite building materials (eg concrete blocks). No attempt has been made to compare doses from the use of buildings formed using materials containing slag with those that would be received using traditional building materials such as natural stone and brick. The results of the investigation are clearly dependent upon the assumptions made regarding concentrations of activity in building materials, which can vary significantly, and the use of a 35% slag content. Nevertheless, they serve to provide a useful overall indication of the radiological impact of using slag in building materials.

Estimated doses to steel plant workers are presented in Table 12. The workers at the steel plant considered to be most exposed to naturally occurring radioactivity are those located at the blast furnace and the slurry lagoons and those involved in the transferral of cooled slag. Workers at the blast furnace are estimated to receive a dose of 84 μSv y^{-1}, estimated doses to workers at the slurry lagoons are 3 μSv y^{-1} and workers involved in digging cooled slag receive doses of 6 μSv y^{-1}. All of these doses have been estimated using conservative assumptions on working practices, it is therefore likely that typical workers will receive lower doses.

5.2 Collective doses and per caput individual doses

Collective doses truncated at 500 years* from stack releases and landfill disposal of steel industry waste were determined. The use of slag in construction materials will clearly also contribute to the collective dose to the UK population. However, there are significant uncertainties involved in the estimation of

* When using collective dose as a measure of total radiation induced health detriment, it has been suggested that the dose truncated at 500 years should be used (Barraclough et al, 1996). A similar conclusion has been reached by ICRP (1997). The greater the integration time, the greater will be the uncertainty surrounding a number of the assumptions made in the calculation of collective doses. Amongst the assumptions that may be affected at long time frames are those associated with human behaviour, population size and the environment.

collective doses from these scenarios and thus they have not been considered in this study. The estimated collective doses, truncated at 500 years, to members of the public in the UK from landfill disposal of steel industry waste are effectively zero, due to the long time taken for radionuclides to migrate from the landfill site into the biosphere.

The estimated collective doses truncated at 500 years to the UK population from stack releases in 1999 from the four integrated steel production plants in the UK are presented in Table 13. These were determined by scaling the site specific unit release calculations undertaken previously by NRPB by the source terms in Table 3. The total collective dose truncated at 500 years is 6.5 manSv. There would be no increase in collective doses from stack releases at later truncation times because the radionuclides considered in the assessment have half lives that are relatively short.

TABLE 13 Collective doses from stack releases in 1999

	Collective dose truncated at 500 years (manSv)
Scunthorpe	3.1
Redcar	1.1
Llanwern¶	1.3
Port Talbot	1.0
Total	6.5

¶ Llanwern ceased iron production in 2001

The continuation of steel production at its current level for many years into the future could give rise to concern over the exposures arising from the build up of long lived radionuclides in the environment. A useful indicator of radiological impact in this situation is the average annual dose to an individual in the exposed population in, for example, the final year after 500 years of continuous operation of the practice. An estimate of this quantity can be made from the collective dose truncated at 500 years, as the maximum annual collective dose rate is numerically equal to the value of the truncated collective dose, all other factors being equal. Thus, assuming a rounded population of 55 million, the per caput dose rate in the UK population from 500 years of continuous steel production at current levels would be 0.1 μSv y^{-1}. It can be concluded that the build up of radionuclides is unlikely to be radiologically significant.

5.3 Comparison of results with those from other studies

There are few reported studies in the literature to compare with this study. The site specific individual dose per unit release rate data produced by NRPB (Mayall et al, 1997a), and used in this study, were also used by British Steel, with site specific information on sinter plant releases, to determine doses to the critical

group for each of the four steel plants (Harvey, 1998). The estimated individual doses to the critical group for the Teesside, Scunthorpe and Port Talbot plants were less then 2 μSv y^{-1}. The highest dose, 18.5 μSv y^{-1}, was for the critical group for the Llanwern plant. These doses are lower than those presented in Table 9. However, those in Table 9 include all stack releases rather than simply those from the sinter stack, and are generally consistent with those in Harvey (1998).

5.4 Comparison of modelling data with environmental measurements

There is considerable natural variability in the concentrations of natural radionuclides in the terrestrial environment. For example, one recent study (Ham et al, 1998) examined radionuclide concentrations in a number of environmental media from two sites in the UK. One of the sites was in an area where levels of natural radiation were typical of the UK and the other was in an area of somewhat higher background levels. Concentrations of lead-210 and polonium-210 in soil were found to range from about 10 Bq kg^{-1} up to 40 Bq kg^{-1} at the typical site, and up to 100 Bq kg^{-1} at the other site. A consequence of this natural variability is that it is difficult to draw firm conclusions about the local significance of man-made sources of natural radionuclides.

TABLE 14 Radionuclide concentrations in soil near steel plants

	Radionuclide concentration in soil, Bq kg^{-1}	
Site	^{210}Pb	^{210}Po
Redcar*	-	28
Port Talbot†	48	42
Scunthorpe†	25	24
Estimated at 500 m from stack	0.01¶	0.3¶
Estimated at 5 km from stack	0.10¶	2.3¶

* Data for Redcar taken from FSA and SEPA (2000).

† Data for Port Talbot and Scunthorpe from MAFF and SEPA (1999).

¶ Values estimated in this study using the source term in Table 2. These are the additional concentrations, above the natural background levels, arising from the stack releases.

Radionuclide concentrations measured in soil around steel production sites (MAFF and SEPA, 1999; FSA and SEPA, 2000), presented in Table 14, are within the ranges expected for natural sources. The estimated soil concentrations arising from atmospheric releases from steel plant generated in this study are also presented in Table 14. It can be seen that these estimated additional concentrations are low in comparison to natural levels.

6 SUMMARY AND CONCLUSIONS

The results of a study to investigate the radiological consequences to the UK population of integrated steel production plants operating within the UK have been presented above.

IAEA has concluded (IAEA, 1988) that a level of dose of some tens of microsieverts a year could reasonably be regarded as trivial by regulatory authorities. IAEA also recommends (IAEA, 1988) the use of a 10 μSv y^{-1} dose criterion for the derivation of exemption levels. The estimated individual doses from atmospheric releases from all the currently operating integrated steel production plants in the UK are less than 10 μSv y^{-1}. These are therefore below the 'trivial' level, and also well below the dose limit to members of the public of 1000 μSv y^{-1}, and the maximum public dose constraint of 300 μSv y^{-1}.

The estimated doses to most workers at the steel production plant, landfill workers dealing with Inventory I, and workers involved in the manufacture and use of products containing slag are in the range of a few, to a few tens of μSv per year, ie at or below the IAEA 'trivial' level. Estimated doses to workers at the blast furnace are slightly higher, 84 μSv y^{-1}. However, it should be noted that this is because very conservative assumptions were made in determining the radionuclide content of the dust at the blast furnace and actual doses are likely to be lower. Estimated doses to landfill workers dealing with inventory II are 1.7 mSv y^{-1}. Again it should be stressed that this disposal inventory is very conservative. If the radionuclides other than lead-210 and polonium-210 are present at the concentrations typical for the raw materials, then the expected doses would be in the region of a few tens of microsieverts per year. It is recommended that more detailed measurements of the radionuclide concentrations of the wastes sent to landfill are undertaken to allow these estimates to be refined. In general it is expected that doses to all workers are lower than 1 mSv y^{-1}, which EC guidance indicates is the dose level below which regulation is not necessary for workplaces processing materials with enhanced levels of naturally occurring radionuclides (EC, 1999a) as is reflected in current UK regulatory guidance (HSE, 2000).

The estimated doses to individuals using car parks or play areas surfaced using slag containing materials are all well below 10 μSv y^{-1}, ie below the 'trivial' level.

The estimated radon concentration in buildings constructed from materials containing slag, originating from radionuclides within the structure, ranged from 7.0 to 10.8 Bq m^{-3}, depending upon the assumptions made. EC guidance (EC, 1999b) recommends that the amount of radium in building materials should be restricted at least to a level where it is unlikely that it would be a major cause of exceeding the design level for indoor radon introduced in the EC Recommendations (200 Bq m^{-3}). The estimated indoor radon activity concentrations are clearly below this level.

The estimated external dose to residents arising from building materials is 758 μSv y^{-1} for standard building materials and 790 μSv y^{-1} when the building materials contain slag. It should be noted that the exposure scenario used in this study is conservative, assuming concrete walls, floors and ceilings, ie bulk quantities of concrete used. Subtracting a typical value for external irradiation outdoors, as used in EC (1999b), gives doses of, respectively, 460 μSv y^{-1} and 490 μSv y^{-1}. These are within the range of 0.3 mSv y^{-1} to 1 mSv y^{-1} (excess external irradiation dose to that received outdoors) within which EC guidance indicates that controls on the use of such building materials should be instituted (EC, 1999b). This conclusion is in agreement with a general evaluation produced by the EC of the possibility of exceeding 0.3 mSv y^{-1} because of the use of certain building materials. This EC study (EC, 1999b) concluded that it was possible that the use of concrete could result in exposures above 0.3 mSv y^{-1} almost anywhere where bulk amounts are used. The study also concluded that exposures above 1 mSv y^{-1} from concrete were possible if bulk amounts are used and the concrete contained large amounts of blast furnace slag, fly ash or natural sand or rock rich in natural radionuclides.

The EC has recently produced guidance on exemption levels for materials containing naturally occurring radionuclides (EC, 2001), see Appendix A for more details. Exemption levels were determined using a set of exposure scenarios. The scenarios included the use of materials containing naturally occurring radionuclides, such as slag, in building materials. The building material scenario adopted more realistic assumptions than those used in this study. The resulting recommended exemption levels are 0.5 Bq g^{-1} for uranium-238 and thorium-232 in secular equilibrium. Exemption levels were also derived for segments of the decay chains. The activity concentrations of the radionuclides in slag are well below these recommended exemption levels; thus providing a further illustration of the low radiological impact of the use of slag in building materials.

In order to put the above doses into context it is worthwhile noting that the average annual dose in the UK from all sources is 2.6 mSv, with a wide variation depending on the location. Most of this variation is due to differences in radon concentrations in homes. An exposure review recently conducted by NRPB (Hughes, 1999) on the UK population estimated that the dose due to radon accounted for up to 50% of the total dose, and that the dose received from exposure to radon had a range of 0.3 – 100 mSv y^{-1}. The majority of this dose arises from the inhalation of radon emitted from the ground beneath homes.

In the UK the acceptability of purpose built disposal facilities for radioactive waste would be judged against a risk target of 10^{-6} y^{-1} (Cmd2919, 1995; EA, 1996) which is equal to the design target recommended by NRPB (1992) for such facilities. This criterion relates to the risk to an average member of the critical group of fatal cancer or serious hereditary defects, for which ICRP recommends a risk factor of 0.06 Sv^{-1} (ICRP, 1991). These criteria all relate to purpose built repositories for radioactive waste, and therefore do not necessarily apply to burial at landfill sites. However, risks below 10^{-6} y^{-1} are considered to be 'broadly acceptable' (Royal Society, 1983; HSE, 1982), and this therefore seems a

reasonable choice of criterion for judging such disposals. The estimated individual risk from the disposal to landfill of inventory I is 5 10^{-9} y^{-1}. This is the risk arising from redevelopment of the site. The doses from migration are effectively zero because, as a result of the level of containment in the landfill and the relatively short half-lives of lead-210 and polonium-210, the radionuclides will have decayed before they migrate from the landfill to the biosphere. The peak risk from disposal of the conservative inventory (inventory II) is 1.5 10^{-8} y^{-1}. The risks from both inventories are both clearly below the 10^{-6} y^{-1} risk criteria.

Currently, radiological controls on the operation of steel production sites are confined to atmospheric releases from the sinter plant stacks. There are no restrictions on the disposal of solid wastes or the use of by-products, which relate to their radionuclide content. This position seems entirely consistent with the low radiological impact of the industry as presented above.

This work forms part of a broad study to assess the radiological impact on the UK population of non-nuclear industries within the UK involved in the production or processing of materials containing enhanced levels of naturally occurring radionuclides. Currently there is much interest in developing regulatory approaches for these industries. It is hoped that the results of this study will also provide useful background information to inform discussions in this area.

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APPENDIX A

LEGISLATION

A1 UK legislation

The primary UK legislation relating to the use of radioactive materials in the UK are the Radioactive Substances Act 1993 (RSA, 1993) and the Ionising Radiations Regulations 1999 (IRR, 1999). These implement the European Union's Basic Safety Standards Directive (BSS)[•] (EC, 1996). The Radioactive Substances Act 1993 (RSA93) regulates the accumulation, storage and disposal of radioactive waste, principally to control potential doses to members of the public. The Ionising Radiations Regulations 1999 (IRR, 1999) deal primarily with regulating the doses that people receive at work. Both of these pieces of legislation apply to the use of materials containing naturally occurring radionuclides. The application of both of these regulatory instruments to the UK steel industry is discussed in detail below.

A1.1 Application of RSA93 to steel plant

Under the provisions of RSA93 all work activities that use radioactive materials need to be registered, and the accumulation and disposal of waste authorised, unless the material is specifically excluded from RSA93. RSA93 covers work involving materials containing naturally occurring radionuclides; however, under Section 1 of RSA93, materials that have concentrations of naturally occurring radionuclides (other than those involved in the nuclear fuel cycle) below those values presented in Schedule 1 of RSA93 are not considered to be radioactive and are therefore excluded from the provisions of the Act. Schedule 1 contains exclusion levels for solids, liquids and gases or vapours. In addition to exclusion, there are a number of Exemption Orders (EOs) made under RSA93 that exempt specific materials from certain provisions of RSA93. Of particular relevance to industries that use or produce materials containing enhanced levels of naturally occurring radionuclides is the Radioactive Substances (Phosphatic Substances, Rare Earths etc.) Exemption Order 1962 (RSEO, 1962). This EO exempts certain materials containing naturally occurring radionuclides from the requirement for registration and authorisation of accumulation and disposal under RSA93.

In Table A1 the RSA93 exclusion levels for solid materials and the relevant exempt concentrations from the Radioactive Substances (Phosphatic Substances,

[•] The BSS has been implemented within the UK through the production of the Ionising Radiations Regulations 1999 (IRR, 1999) and minor regulatory amendments (BSS, 2000a; 2000b) to the Radioactive Substances Act 1993 (RSA, 1993) and by Directions (BSS, 2000c; 2000d) to the applicable Environment Agencies.

Rare Earths etc.) Exemption Order 1962 (RSEO, 1962) are compared with the concentrations of radionuclides in various steel plant waste streams considered in this study. The Schedule 1 and EO values are defined in terms of the elements present. It is important to note, therefore, that in determining the concentrations for comparison with these values judgements have to be made about which radionuclides to include. In this analysis all radionuclides have been included. Other interpretations are possible and the final judgement is that of the regulatory authority.

TABLE A1 Comparison of RSA93 Schedule 1 and Radioactive Substances (Phosphatic Substances, Rare Earths etc.) Exemption Order (RSEO, 1962) activity concentrations for solid materials with those assumed in this study for various steel plant waste streams

	Activity concentration (Bq g^{-1})					
Element*	RSA93	EO	Waste from gas cleaning system of sinter plant†	Blast furnace and BOS slurry†	Total waste to landfill‡	Slag†
Actinium	0.37	14.8				0.05
Lead	0.74	14.8	0.91	0.90§ (0.36)¶	0.90	0.14
Polonium	0.37	14.8	3.94	0.22§ (0.09)¶	0.29	0.26
Protactinium	0.37	14.8				0.09
Radium	0.37	14.8				0.19
Thorium	2.59	14.8				0.28
Uranium	11.1	14.8				0.18

* Activity concentrations for each element were determined by summing the activity concentrations of each radioactive isotope of the element present.

† From Giles and Harvey (2000).

‡ Derived from concentrations of first two waste streams and total quantities of waste produced from Giles and Harvey (2000).

§ Concentrations following storage in lagoons (ie following reduction in water content).

¶ Initial concentrations in wet waste from gas cleaning system.

From Table A1 it is clear that the concentrations of both lead and polonium in the waste from the sinter plant gas cleaning system are above the exclusion levels. The concentrations of lead in the waste from the Blast furnace and BOS furnace gas cleaning systems (following intermediate storage in lagoons to reduce water content) and the wastes sent to landfill are also above the exclusion level. The concentrations in all the waste streams are, however, below the EO values and there is therefore no regulatory requirement for authorisation of the accumulation or disposal of these materials.

In Table A2 the RSA93 exclusion levels for gases and vapours are compared with the concentrations of radionuclides in atmospheric releases from the sinter plant. It is clear that the concentrations of radionuclides in the atmospheric releases are above the defined exclusion levels. Therefore, under the terms of RSA93, authorisations for atmospheric releases from each of the four UK steel production plants are required. Corus applied for authorisations for each plant and the

Environment Agency granted these (EA, 1998; 2000a; 2000b). In general, the authorisations all state similar conditions; only gaseous emissions from the sinter stacks are covered, there are both daily and annual discharge limits (specified by the activity released) and the authorisations specify the need for monitoring and keeping of records, with a copy sent to the Environment Agency each year.

TABLE A2 Comparison of RSA93 Schedule 1 activity concentrations for gases/vapours with those for atmospheric releases used in this study

	Activity concentration (Bq per gramme of gaseous release)	
Element	RSA93	Atmospheric release from sinter plant*
Lead	$1.11\ 10^{-4}$	$1.0\ 10^{-3}$
Polonium	$2.22\ 10^{-4}$	$2.8\ 10^{-3}$

* Activity concentrations for each element were determined by summing the activity concentrations of each radioactive isotope of the element present. Activity concentrations from Harvey (1998).

A1.2 Application of IRR99 to steel plant

Regulation 3 (Application) of IRR99 makes it clear that the scope of the regulations includes work with radioactive substances containing naturally occurring radionuclides. The associated approved code of practice (HSE, 2000) provides more detailed guidance in this area. In the case of substances containing naturally occurring radionuclides used in work other than a practice*, eg steel plants, the regulations only apply if 'their use is likely to lead to employees or other people receiving an effective dose of ionising radiation in excess of 1 millisievert in a year'. The results of this study indicate that doses to workers and members of the public are significantly less than 1 mSv y^{-1} and thus the regulations would not be applied at steel plants.

A2 Relevant EU legislation and guidance

The European Union has its own set of regulations relating to radioactive materials, the Basic Safety Standards Directive (EC, 1996), which the UK had to bring its regulations into line with by May 2000. The primary scope of the BSS is 'all practices which involve a risk from ionizing radiation from an artificial source or from a natural radiation source in cases where natural radionuclides are or

* In IRR99 a practice is defined as work involving the production, processing, handling, use, holding, storage, transport or disposal of radioactive substances; or the operation of any electrical equipment emitting ionising radiation and containing components operating at a potential difference of more than 5kV, which can increase the exposure of individuals to radiation from an artificial source, or from a radioactive substance containing naturally occurring radionuclides which are processed for their radioactive, fissile or fertile properties.

have been processed in view of their radioactive, fissile or fertile properties'. Therefore the general scope does not cover exposures due to natural radiation sources other than when these are part of the nuclear fuel cycle. However, paragraph 2 of Article 2 states that the Directive also applies to 'work activities which involve the presence of natural radiation sources and lead to a significant increase in the exposure of workers or members of the public which cannot be disregarded from the radiation protection point of view'. Title VII of the BSS expands on this by requiring each member state to identify work activities involving exposure to natural radiation sources which may be of concern from a radiological protection point of view and to apply the requirements of the Directive to the identified activities. The BSS therefore allows each member state a degree of discretion in this area.

The European Commission (EC) has, however, produced a document to assist in the implementation of Title VII, which gives reference levels for workplaces processing materials with enhanced levels of naturally occurring radionuclides (EC, 1999a). This guidance is not binding on member states but it offers a simple technique for screening and categorising the relevant industries based on radiation dose criteria. The guidance is limited to consideration of occupational exposures. The guide proposes four control bands, as follows:

Band 1 no need to consider regulation,

Band 2 lower level of regulation should be applied,

Band 3 higher level of regulation should be applied,

Band 4 process should not be permitted without a full individual assessment.

The four band system has three marker points to separate the bands related to the radiation doses workers receive. The doses chosen are 1 mSv y^{-1}, 6 mSv y^{-1} and 20 mSv y^{-1} under normal conditions, and 6 mSv y^{-1}, 20 mSv y^{-1} and 50 mSv $^{-1}$ under unlikely conditions. The guide therefore recommends that, for the relevant industries, if worker exposures are under 1 mSv y^{-1} under normal conditions no regulation is required.

The BSS does not apply to exposure to radon in dwellings. The EC recommendation on radon in dwellings (EC, 1990) introduces a design level for radon exposure. The design level corresponds to an average radon gas concentration of 200 Bq m^{-3}. The design level is to be used to aid relevant national authorities in establishing regulations, standards or codes of practice for circumstances under which the design level might otherwise be exceeded. It also acts as an exclusion level, since concentrations below 200 Bq m^{-3} do not need to be taken into account when assessing doses.

The EC has also produced guidance for setting controls on the radioactivity of building materials (EC, 1999b). The purpose of setting controls on the radioactivity of building materials is to limit the radiation exposure due to materials with enhanced or elevated levels of natural radionuclides. The general

principle is that the doses to members of the public should be kept as low as reasonably achievable. However, since small exposures from building materials are ubiquitous, controls should be based on exposure levels that are above typical levels of exposures and their normal variation. The guidance states that the amount of radium in building materials should be restricted at least to a level where it is unlikely that it could be a major cause for exceeding the design level for indoor radon of 200 Bq m^{-3}. The guidance also states that controls on building materials should be required for estimated doses in the range 0.3 – 1 mSv y^{-1} (this is the excess gamma dose to that received outdoors), although it is acknowledged that higher doses could be accepted in some very exceptional cases where materials are used locally. It is further stated that building materials should be exempted from all restrictions concerning their radioactivity if the excess gamma radiation originating from them increases the annual effective dose of a member of the public by, at most, 0.3 mSv (this is the excess gamma dose to that received outdoors).

The EC has also, more recently, produced guidance on the application of the concepts of exemption and clearance to natural radiation sources (EC, 2001). These concepts relate to different ways of avoiding wastage of regulatory resources in cases where there would be no or nothing but a trivial benefit. The guidance concludes that, because of the large volumes of material processed and released by industries processing or producing materials containing enhanced levels of naturally occurring radionuclides, it is appropriate to lay down a single set of levels for both exemption and clearance. For the purpose of deriving general clearance and exemption levels for materials containing naturally occurring radionuclides a set of enveloping scenarios and parameters were developed on the basis of expert opinion. These scenarios include the use of by products such as slag in construction materials. Exemption-clearance levels were calculated using these enveloping scenarios and parameter values. A dose criterion of a dose increment, in addition to background exposure from natural radiation sources, of 300 μSvy^{-1} was used. For both uranium-238 and thorium-232 in secular equilibrium the applicable level is 0.5 Bqg^{-1}. Clearance-exemption levels were also derived for segments of the decay chains. The activity concentrations of the steel plant waste streams and slag, presented in Table A3, are all below the recommended clearance-exemption levels.

The BSS requires practices to be reported unless one of a number of conditions is met. One condition is that the concentrations of radionuclides should be less than the exemption levels contained within the BSS. Although, as has been mentioned above, these requirements are not strictly applicable to work activities such as steel production, a member state could, if it wished, decide to apply them if it felt that the radiological impact could be of concern. However, in the UK, the BSS reporting requirement has been implemented through the IRR99 notification levels. These are the same as the BSS exemption levels. As shown in Table A3, the radionuclide concentrations in the relevant materials produced at steel plants are below the notification levels and hence would be exempt from the requirement for reporting even if it was considered applicable.

TABLE A3 Comparison of IRR99 Schedule 8 activity concentrations for solid materials with those assumed in this study for various steel plant waste streams

	Activity concentration (Bq g^{-1})				
Radionuclide*	IRR99 column 2 part I Schedule 8	Waste from gas cleaning system of sinter plant†	Blast furnace and BOS slurry†	Total landfill‡	Slag†
^{238}U +	1 10^{1}				8.8 10^{-2}
^{234}U	1 10^{1}				8.8 10^{-2}
^{230}Th	1 10^{0}				8.8 10^{-2}
^{226}Ra+	1 10^{1}				8.8 10^{-2}
^{210}Pb	1 10^{1}	9.1 10^{-1}	9.0 10^{-1} (3.6 10^{-1})	9.0 10^{-1}	
^{210}Po	1 10^{1}	3.9 10^{0}	2.2 10^{-1} (9.0 10^{-2})	2.9 10^{-1}	
^{232}Th sec	1 10^{0}				4.9 10^{-2}
^{235}U +	1 10^{1}				4.0 10^{-3}
^{231} Pa	1 10^{0}				4.0 10^{-3}
^{227}Ac +	1 10^{-1}				4.0 10^{-3}
^{227}Th	1 10^{1}				4.0 10^{-3}
^{223}Ra +	1 10^{2}				4.0 10^{-3}

* The terms 'sec' and '+' indicate that in deriving the IRR99 Schedule 8 values for these radionuclides some or all of their radioactive daughters have been assumed to be in equilibrium with the parent. See reference IRR99 (IRR, 1999) for more details.

† From Giles and Harvey (2000). For blast furnace dust and BOS slurry concentrations in brackets are for (wet) waste direct from gas cleaning system. Main concentrations are for waste prior to disposal following interim storage in lagoons, during which time water content is reduced.

‡ Derived from concentrations of first two waste streams and total quantities of waste produced from Giles and Harvey(2000).

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APPENDIX B

GENERIC CALCULATIONS OF DOSES FROM ATMOSPHERIC RELEASES FROM THE STACK, LAGOONS AND STORED SLAG

B1 Introduction

As was identified in the main text, there was a need as part of this study to determine generic doses from atmospheric releases from a 'typical' steel plant to allow direct comparison with those from other UK industries that are being considered as part of the overall study of the radiological impact of such industries in the UK. These generic calculations are undertaken using the same set of defined conservative assumptions for each industry.

Releases from the stack, slurry lagoons and slag piles have been considered. The stack releases in Table 2 of the main text have been used. The activity concentrations of slurry dust and slag used are those in Tables 4 and 5 of the main text.

B2 Exposure pathways

Radionuclides released to the atmosphere either from the stack or by re-suspension from the lagoons or slag heaps may expose people by six pathways:

a inhalation of radionuclides in the plume;
b external irradiation by radionuclides in the plume;
c inhalation of radionuclides resuspended following deposition;
d external irradiation by deposited radionuclides;
e ingestion of food grown in land contaminated by a deposit of radionuclides; and
f inhalation of radon.

Direct external irradiation from slag piles is discussed in Section B8.

B3 Habit data

For these generic calculations two exposed groups are considered: a group intended to represent the typical exposure of local people in the area, and a group of people who are assumed to be located at a place, and have habits such

that they are likely to receive high doses from the plant. The habits of a typically exposed person have been based on assumptions described elsewhere (Lawson et al, 1990; Robinson, 1996; Simmonds et al, 1993). It was assumed they live 5 km from the release point, and remain at that location for the entire year. During this time individuals of all age groups were assumed to be outside for 10% and inside for 90% of the year (Robinson, 1996). They were assumed to eat all foodstuffs at an average rate and obtain 25% from the local area except for green vegetables, where there is evidence that 50% is locally sourced (Simmonds et al, 1993). The assumptions used to characterise individuals receiving a high dose from the release were based on those described in Robinson et al (1994). It is assumed that they live on a farm 500 m from the release point. It has further been assumed that they spend all their time at the location, adults spend 50% of the time outside and children and infants spend 20% of their time outside. They obtain all their food from local sources, eating the two most significant foodstuffs at a critical rate and others at average rates. Doses to adults, children and infants were determined. The characteristics of these two groups are summarised in Table B1. It should be stressed that the assumptions made for the two groups are not site specific but are generic assumptions applicable to the UK. The estimated doses will therefore differ from those that would be determined using site-specific information on locations of habitation and agricultural practices around a site. They are intended simply to facilitate comparison of radiological impacts with other industries.

B4 Models used to calculate doses

The atmospheric dispersion of radionuclides was modelled using PLUME (part of the PC CREAM suite of models (Mayall et al, 1997)). Assuming a continuous release of radionuclides for one year, this model was used to estimate the radionuclide concentrations in air (Bq m^{-3}), deposition rates (Bq m^{-2} s^{-1}) and external doses from gamma irradiation from the cloud (Sv y^{-1}) at the points of interest.

A release height of 100 m was used for the determination of individual doses from stack releases. This corresponds, approximately, to the actual stack height at the steel plant considered of 107 m (Giles and Harvey, 2000). PLUME was

TABLE B1 Characteristics of generic exposed groups for atmospheric releases

	Typical exposure	High exposure	Units
Distance from release	5000	500	m
Time spent at location	8760	8760	h y^{-1}
Fraction of time indoors	0.9 all age groups	0.5 adults 0.8 children (1 and 10 years)	-
Fraction of time outdoors	0.1 all age groups	0.5 adults 0.2 children (1 and 10 years)	-
Inhalation rate	0.22 1 year old 0.65 10 year old 0.83 adult	0.22 1 year old 0.65 10 year old 0.83 adult	m^3 h^{-1}
Food integration time	50	50	y
Ingestion rate: green veg	2.5 – 1 year old 5 – 10 year old 15 – adult	5 – 1 year old 10 – 10 year old 30 – adult	kg y^{-1}
Ingestion rate: root veg	3.75 – 1 year old 12.5 – 10 year old 15 – adult	15 – 1 year old 50 – 10 year old 60 – adult	kg y^{-1}
Ingestion rate: grain	3.75 – 1 year old 11.25 – 10 year old 12.5 – adult	30 – 1 year old 75 – 10 year old 100 – adult	kg y^{-1}
Ingestion rate: fruit	1.875 – 1 year old 3.75 – 10 year old 3.75 – adult	7.5 – 1 year old 15 – 10 year old 15 – adult	kg y^{-1}
Ingestion rate: cow meat	0.75 – 1 year old 2.5 – 10 year old 3.75 – adult	3 – 1 year old 10 – 10 year old 15 – adult	kg y^{-1}
Ingestion rate: cow liver	0.05 – 1 year old 0.125 – 10 year old 0.25 – adult	0.2 – 1 year old 0.5 – 10 year old 1 – adult	kg y^{-1}
Ingestion rate: milk	30 – 1 year old 27.5 – 10 year old 23.75 – adult	120 – 1 year old 110 – 10 year old 95 – adult	kg y^{-1}
Ingestion rate: sheep meat	0.15 – 1 year old 0.375 – 10 year old 0.75 – adult	0.6 – 1 year old 1.5 – 10 year old 3 – adult	kg y^{-1}
Ingestion rate: sheep liver	0.05 – 1 year old 0.125 – 10 year old 0.25 – adult	2.75 – 1 year old 5 – 10 year old 10 – adult	kg y^{-1}
Indoors reduction factor: Inhalation dose	1	1	-
Indoors reduction factor: Cloud gamma dose	0.2	0.2	-
Indoors reduction factor: Dose from deposited radionuclides	0.1	0.1	-

also used to calculate the required results for releases from lagoons and slag heaps assuming a ground level release. Typical UK meteorological conditions corresponding to a 60% frequency of Pasquill stability category D were assumed (Clarke, 1979). PLUME default washout coefficients and deposition velocities were used, respectively, 10^{-4} and 0.001 m s^{-1}. A uniform windrose was assumed.

Radionuclides which deposit from the plume remain at the same location for a significant time, and therefore doses arising from deposited radionuclides (external exposure from radionuclides deposited on the ground, inhalation of resuspended radionuclides and the ingestion of foods grown on land on which radionuclides have been deposited) were integrated over time. An integration period of 50 years was considered appropriate. The estimated doses integrated over 50 years, for a continuous release rate for 1 year, are numerically equal to the highest annual dose that would be received assuming the plant operates for 50 years. The integrated doses from inhalation of resuspended radionuclides, external irradiation from deposited radionuclides and ingestion of food were calculated for a unit deposition rate (Bq m^{-2} s^{-1}) using parts of the PC CREAM suite (RESUS and GRANIS) and the NRPB terrestrial foodchain model, FARMLAND (Brown and Simmonds, 1995).

A spreadsheet was set up to scale the above data according to release rates, and to determine doses by incorporating the habit data and summing over radionuclides. The calculations performed in the spreadsheet are described below.

B5 Release rate from the stack

If constant operation is assumed, the average rate of discharge of radionuclides in dust and ash from the stack (in Bq s^{-1}) is

$$A_{R,N} = [(A_{SP,N} \times Q_{SP}) + (A_{BF,N} \times Q_{BF})]/spy$$

where $A_{R,N}$ = Activity released of radionuclide N (Bq s^{-1})

$A_{SP,N}$ = Activity in sinter plant dust of radionuclide N (Bq kg^{-1})

Q_{SP} = Quantity of sinter plant dust released to atmosphere per year (kg y^{-1})

$A_{BF,N}$ = Activity in blast furnace dust of radionuclide N (Bq kg^{-1})

Q_{BF} = Quantity of blast furnace dust released to atmosphere per year (kg y^{-1})

spy = 3.15 10^7 seconds per year

The values of $A_{SP,N}$, $A_{BF,N}$, Q_{SP} and Q_{BF} used in this study are discussed in the main text and are presented in Table B2. The values of $A_{R,N}$ used for the determination of individual doses are also presented (in terms of Bq y^{-1}) in the main text of the report, see Table 2.

TABLE B2 Radionuclide concentrations and total quantity of stack releases

Location	Quantity of dust (kg y^{-1})	Activity concentration ^{210}Pb (Bq kg^{-1})	Activity concentration ^{210}Po (Bq kg^{-1})	Activity released ^{210}Pb (Bq y^{-1})	Activity released ^{210}Po (Bq y^{-1})
Sinter plant:					
Main stack	$3.9\ 10^5$	$1.13\ 10^4$	$9.98\ 10^4$	$4.41\ 10^9$	$3.89\ 10^{10}$
De-dust stack	$1.6\ 10^5$	-	-	-	-
Blast furnace:					
Hot blast stoves	$1.3\ 10^5$	$8.00\ 10^3$	$2.80\ 10^3$	$1.04\ 10^9$	$3.64\ 10^8$
Furnace bleeders	$2.3\ 10^4$	$8.00\ 10^3$	$2.80\ 10^3$	$1.84\ 10^8$	$6.44\ 10^7$
Semi-cleaned gas bleeders	$2.3\ 10^3$	$8.00\ 10^3$	$2.80\ 10^3$	$1.84\ 10^7$	$6.44\ 10^6$
Cast house ventilation	$7.1\ 10^4$	$8.00\ 10^3$	$2.80\ 10^3$	$5.68\ 10^8$	$1.99\ 10^8$
Total				$6.22\ 10^9$	$3.96\ 10^{10}$

B6 Release rates from slurry lagoons and slag storage areas

At all steel production sites there are lagoons to store dust collected from the wet gas cleaning system. The material stored is a wet slurry, the purpose of the lagoons is to allow much of the water to drain away before the remaining slurry is disposed to landfill. There are two lagoons on the site chosen for study, each with a volume of 50 000 m^3 (Giles and Harvey, 2000). It has been assumed that they are 10 m deep with a length and width of 71m. There are also heaps of slag on all sites. The slag is removed from the blast furnace and the BOS furnace and stored on site before sale for recycling. At the site chosen for study, the company that the slag is sold to is situated next to the steel plant. The slag may be stored outside for up to 18 months before recycling, see main text. It has been assumed that the public are exposed to atmospheric releases from the total amount of slag produced in 1 year. Since the production of slag and transfer for recycling is a continuous process, in any year this quantity of slag will be stored outside. It is further assumed that the total amount of slag is stored in two separate piles, one at the steel plant and one on the neighbouring site. These piles will each have volumes of $4.55\ 10^5$ m^3. This is based on the total mass of slag produced in one year, $1.6\ 10^9$ kg y^{-1}, and the density of slag, which has been assumed to be 1760 kg m^{-3}, see main text for discussion of these assumptions.

The effects of atmospheric releases from slag storage areas and slurry lagoons were assessed separately as it is considered that an individual is unlikely to be exposed to both slag heaps and lagoons at the same time. The site is large and the slag heaps would be at least 500m from the lagoons, on the other side of the site, with buildings in between them.

The consequences of releases of radon-220 were not considered because its half-life is very short, it is therefore unlikely to travel any distance from the lagoons or slag heaps before decay and the probability of exposure of the public is very small. Table B3 summarises the radionuclide independent scenario data used in the calculation of doses resulting from releases from lagoons and slag heaps.

TABLE B3 Radionuclide independent scenario data for releases from lagoons and slag heaps

Parameter		Units
Ambient dust concentration:		
Dust lagoons	$1\ 10^{-4}$	g m^{-3}
Slag heaps	$1\ 10^{-4}$	g m^{-3}
Windspeed	4.4	m s^{-1}
Emanation fraction Rn-222	0.23	-
Radon diffusion coefficient	$1\ 10^{-8}$	m^2 s^{-1}
Density:		
Slurry	2.1	t m^{-3}
Slag	1.8	t m^{-3}
Effective width†:		
Slurry lagoons	142	m
Slag piles	$3.64\ 10^4$	m
Surface area of slag heaps‡	$2.6\ 10^5$	m^2
Rn-222 lambda	$2.1\ 10^{-6}$	s^{-1}
Diffusion length for Rn-222	0.18	m

† Width used in determination of atmospheric releases from wind-driven resuspension

‡ Area from which Radon is emanating

B6.1 Release rate from slurry lagoons

The release rate of radionuclide N in dust from the lagoons (by resuspension), $A_{R,N}$ Bq s^{-1}, was estimated by assuming that at any instant the concentration of dust in the air up to 1m above the lagoon was $1\ 10^{-4}$ g m^{-3} (Simmonds et al, 1995). Typical dust loadings in large industrial areas can range from 100 – 800 μg m^3 (Simmonds et al, 1995). As the material is damp a dust loading at the lower end of this range was considered appropriate. The concentrations of radionuclides assumed to be in the dust are given in Table 4 of the main report. It is assumed that dust is continuously swept off the top of the lagoon by the wind. The release rate is thus

$$A_{R,N} = A_{A,N} \times c_A \times d_A \times hc_A \times v_A$$

where $A_{A,N}$ = Activity in dust of radionuclide N, see main text Table 4 (Bq g^{-1})

c_A = Concentration of resuspended dust above the lagoon, $1\ 10^{-4}$ g m^{-3}

d_A = Width of the lagoon (71m), two lagoons give an equivalent length of 142 m

hc_A = Height above the pile that the concentration c_A persists, 1m

v_A = Average windspeed; for atmospheric stability with average 60% neutral conditions, as appropriate for central England, the average windspeed is 4.4 m s^{-1} (Clarke, 1979).

B6.2 Release rate from slag storage area

The release rate of radionuclide N in dust from stored slag (by resuspension), $A_{R,N}$ Bq s^{-1}, was estimated by assuming that at any instant the concentration of dust in the air up to 1 m above the slag heaps was 1 10^{-4} g m^{-3} (Simmonds et al, 1995). Typical dust loadings in large industrial areas can range from 100 – 800 μg m^3 (Simmonds et al, 1995). The slag consists of large lumps of material, a dust loading at the lower end of this range was therefore considered appropriate. The concentrations of radionuclides assumed to be in the slag dust are given in Table 5 of the main report. It is assumed that dust is continuously swept off the side of the pile by the wind. The release rate is thus

$$A_{R,N} = A_{A,N} \times c_A \times d_A \times hc_A \times v_A$$

where $A_{A,N}$ = Activity in slag dust of nuclide N, see main text Table 5 (Bq g^{-1})

c_A = Concentration of resuspended dust above the stored slag, 1 10^{-4} g m^{-3}

d_A = Width of the pile acting as a source, 1.82 10^4 m, two piles give an effective width of 3.64 10^4 m

hc_A = Height above the pile that the concentration c_A persists, 1m

v_A = Average windspeed; for atmospheric stability with average 60% neutral conditions, as appropriate for central England, the average windspeed is 4.4 m s^{-1} (Clarke, 1979).

It is also assumed that radon is released from the stored slag. The rate of exhalation of radon-222, $E_{R,Rn}$ in Bq m^{-2} s^{-1}, from the pile of slag by diffusion was calculated using a method described in UNSCEAR (1993). The exhalation rate depends on two important parameters, emanation fraction and diffusion coefficient. Although these parameters have been studied extensively for soils and building materials, few data are available for slag. UNSCEAR (1993) gives an emanation fraction of 0.23 for radon-222 from soil and a diffusion coefficient of 1 10^{-8} m^2 s^{-1} for radon through building materials, where the building materials have a porosity of 0.15. These values were used as a conservative representation of the slag heap under consideration. The diffusion coefficient and porosity of building materials have been assumed because slag is used as an aggregate in building materials so is expected to have similar properties. It was assumed that the radon was in secular equilibrium with its precursor. This is a reasonable assumption as it takes about 1 month for equilibrium to be re-established (Penfold et al, 1997) and the slag is stored outside for up to 18 months.

$$E_{R,Rn} = A_{A,Ra} \times K_{Rn} \times \lambda_{Rn} \times \rho \times L_D$$

where $A_{A,Ra}$ = Activity concentration of the radon precursor in the slag, ^{226}Ra for ^{222}Rn, in Bq g^{-1}

K_{Rn} = Emanation fraction for the material, assumed to be 0.23

λ_{Rn} = Decay constant for the ^{222}Rn, 2.099 10^{-6} s^{-1}

ρ = Density of the slag, 1.8 10^{6} g m^{-3} has been assumed (TFHRC, 2000)

L_D = Diffusion length (m), given by $\sqrt{(D_{Rn} / \varepsilon \lambda_{Rn})}$, where D_{Rn} is the effective diffusion coefficient of radon, assumed to be 1 10^{-8} m^2 s^{-1} and ε is the porosity of building materials, assumed to be 0.15 (UNSCEAR, 1993)

The release rate in Bq s^{-1}, is therefore,

$$A_{R,\,Rn} = E_{R,Rn} \times d_A$$

where $A_{R,\,Rn}$ = Activity of radon released (Bq s^{-1})

d_A = Area of slag pile from which radon is emanating, ie the surface area of one face of both slag piles, 2.6 10^{5} m^2

B7 Method of calculating individual doses from atmospheric releases

The following equations were used to estimate doses from the various pathways considered. These equations incorporate the results of the models and the habit data assumed. They apply equally to the calculation of doses from the release of dust and radon from the stack and the release of dust and radon from slag heaps and lagoons.

B7.1 Inhalation of radionuclides in the plume

The dose from inhalation of radionuclides in the plume, $D_{PI,N}$, in Sv y^{-1}, was calculated using the equation given below. This was used for both the inhalation of dust and radon. However, for radon no inhalation rate was used, as the radon dose coefficient is in the form of Sv h^{-1} per Bq m^{-3}.

$$D_{PI,N} = A_{R,N} \times AC \times DC_{I,N} \times T_{PI} \times I_{PI} \times (f_O + [f_I \times r_I])$$

where $A_{R,N}$ = Activity release rate (Bq s^{-1}), of nuclide N.

AC = Air concentration (Bq m^{-3} per Bq s^{-1}) for a continuous release from either stack or at ground level, at a distance of 5000 m for the typical exposed group or at 500 m for the high exposure group, these values were calculated using the PLUME (Mayall et al, 1997) model

$DC_{I,N}$ = Effective dose coefficient for inhalation of radionuclide N by members of the public (Sv Bq^{-1}) (ICRP, 1996)

T_{PI} = Exposure duration, assumed to be 8760 h y^{-1}

I_{PI} = Average inhalation rate of the individual exposed (m^3 h^{-1})

f_O = Fraction of time that a person is outside

f_I = Fraction of time that a person is inside

r_I = Fractional reduction in activity concentration inside. This value was conservatively assumed to be 1, i.e. no reduction, for both typical and critical group exposures.

B7.2 External irradiation by radionuclides in the plume

The dose rate from external irradiation from the plume of radioactivity, $D_{PE,N}$ in Sv y^{-1}, was calculated by the PLUME model using a finite cloud model. These data, in Sv y^{-1} per Bq s^{-1}, were scaled to the appropriate release, taking into account indoor occupancy and shielding, using the following equation.

$$D_{PE,N} = A_{R,N} \times DE_N \times (f_O + [f_I \times r_{PE}])$$

where DE_N = Dose rate (in Sv y^{-1} per continuous release rate of Bq s^{-1} of radionuclide N) to a person immersed in a plume of radionuclide N, at 5000 m (typical exposure) or 500 m (high exposure) from the source, these values were calculated by PLUME for each radionuclide and each type of release

r_{PE} = Fractional reduction in external dose from radionuclides in the plume when indoors, 0.2

B7.3 Inhalation of resuspended radionuclides

The integrated resuspended air concentration was calculated using the resuspension model developed by Garland and implemented in the RESUS code (Mayall et al, 1997). This empirical model, based on experimental observations, assumes that the resuspended concentrations vary over time as the radionuclides become associated with the soil surface, and predicts integrated resuspended concentrations (Bq s m^{-3}) per unit deposition rate (Bq m^{-2} s^{-1}). The deposition rate was estimated by the PLUME model. The maximum annual individual dose, $D_{DI,\ N}$ in Sv y^{-1}, assuming that the plant operates for 50 years, was calculated using

$$\Delta D_{DI,\ N} = A_{R,N} \times DR \times {}^{t}\int A_{DI,N} \times DC_{I,N} \times I_{RI} \times (f_O + [f_I \times r_I]) \ / \ sph$$

where DR = Deposition rate (Bq m^{-2} s^{-1} per Bq s^{-1}, for radionuclide N) for a release from either stack or at ground level, at a distance of 5000 m for the typical exposed group or 500 m for the high exposure group

$^{t}\int A_{R,N}$ = Resuspended activity concentration integrated to 50 years (Bq s m^{-3}), as calculated by RESUS for nuclide N, from a deposition rate of 1 Bq m^{-2} s^{-1} for 1 year

I_{RI} = Inhalation rate (m^3 h^{-1})

sph = Number of seconds in an hour (3600 s h^{-1})

B7.4 External iradiation from deposited radionuclides

The integrated external dose from deposited radionuclides was calculated using the GRANIS model (Mayall et al, 1997). This model incorporates a soil model to account for the downward migration of radionuclides in soil, and estimates the integrated dose in Sv per unit deposition rate in Bq m^{-2} s^{-1}, ingrowth of daughters in included explicitly in the model. The maximum annual individual dose, $D_{DE,N}$ in Sv y^{-1}, assuming the plant operates for 50 years, was calculated using

$$D_{DE,N} = A_{R,N} \times DR \times {}^{t}\int A_{DE,N} \times (f_O + [f_I \times r_{DE}])$$

where $^{t}\int A_{DE,N}$ = Effective dose to 50 years (Sv), as calculated by GRANIS for radionuclide N, from 1 year's deposition at a rate of 1Bq m^{-2} s^{-1}

r_{DE} = Fractional reduction in external dose from deposited radionuclides when indoors, 0.1

B7.5 Ingestion of foodstuffs grown in contaminated soil

The dose from the ingestion of a variety of foodstuffs was calculated by summing the integrated dose from the ingestion of each foodstuff, P. The integrated activity concentration in a food P (Bq y kg^{-1} for 50 y) was calculated using the FARMLAND model (Brown and Simmonds, 1995). These values were calculated for a unit deposition rate (Bq m^{-2} s^{-1}) and were then scaled to estimate the activity in food grown at a nearby farm. The dose rate from ingestion of food, P, $D_{Ing,\,P,\,N}$, was calculated using the following equation

$$D_{Ing,\,P,\,N} = DR \times I_{F,\,P} \times {}^{t}\int A_{DF,N,\,P} \times A_{R,\,N} \times DC_{Ing,\,N}$$

where DR = Deposition rate (Bq m^{-2} s^{-1} per Bq s^{-1}) for a release from either stack or at ground level, at a distance of 5000 m for the typical exposed group or 500 m for the high exposure group

$^{t}\int A_{DF,N,P}$ = Integrated activity concentration of radionuclide N, in food P (Bq y kg^{-1}), calculated using FARMLAND for a 50 year integration time for a continuous deposition of 1 Bq m^{-2} s^{-1} for 1 year

$I_{F.P}$ = Annual intake of food, P, in kg

$DC_{Ing,N}$ = Effective dose coefficient for ingestion of radionuclide N and its short-lived daughters by members of the public (Sv Bq^{-1}) (ICRP, 1996)

For the calculation of doses to the typical exposure group $I_{F,\,P}$ is equal to 0.25 × average ingestion rate for all foods, except green vegetables where 50% of the average person's consumption is locally sourced. For the calculation of doses to the high exposure group the two most significant food groups were identified using the above calculation, with critical consumption rates for all foods. $I_{F,\,P}$ is then equal to the critical consumption rate for the two most significant food groups and the average consumption rate for all other food groups (see Table B1).

In general, the FARMLAND calculations do not consider in-growth of daughters, or build up of daughters in the soil. These assumptions are reasonable for the uranium-235 decay series (uranium-235, protactinium-231 and actinium-227), for the upper members of the uranium-238 decay series (uranium-238, uranium-234, thorium-230 and radium-226) and for thorium-232 because all of the radionuclides being considered have very long half-lives. For radium-228, the integrated activity was multiplied by the sum of the dose coefficients for radium-228, actinium-228, thorium-228 and all daughters to the stable isotope. The dose from thorium-228 was also calculated using the integrated activity from FARMLAND and the sum of the dose coefficients of all daughters to the stable isotope. This was done to account for the ingrowth that will occur because of the shorter half-lives of radium-228 and thorium-228. For lead-210 and polonium-210 additional non-standard FARMLAND calculations were undertaken to obtain results for the ingrowth of lead-210 from polonium-210. These integrated activities were therefore multiplied by their respective dose coefficients and no summing of dose coefficients was required.

The maximum annual individual dose, $D_{DF,N}$ in Sv y^{-1}, from ingestion of all foods containing radionuclide N, assuming the plant operates for 50 years, is then

$$D_{DF,N} = \Sigma_P D_{Ing,\,P,\,N}$$

B8 Method of calculating doses from direct irradiation (slag only)

Direct irradiation of the public by material in the lagoons is considered to be zero because the lagoons, when full, are level with the ground.

The dose from direct radiation from slag piles was calculated assuming that the source presents a face of 2.6 10^5 m^2, and has a density of 1.8 t m^{-3}. The dose rates at 500 m and 5 km were calculated using the code QAD-CG (Cain, 1977),

assuming that the slag has build-up and attenuation characteristics approximately the same as concrete. The effective dose from this source was calculated using the following equation.

$D_{DD,\,N} = A_{A,N} \times DE_{DD,N} \times T_{DD} \times (f_O + [f_I \times r_{DE}])$

where $DE_{DD,N}$ = Dose rate from a pile of slag at a distance from the exposed individual, Sv h^{-1} per Bq g^{-1} for radionuclide N

T_{DD} = Time exposed to external irradiation from the ash piles, 8760 h y^{-1}

B9 Total individual doses from stack releases, slurry lagoons and stored slag

For stack discharges the total dose for each pathway was calculated by summing the doses arising from the radionuclides.

$D_{PI} = \sum^N D_{PI,N};\ D_{PE} = \sum^N D_{PE,N};\ D_{DI} = \sum^N D_{DI,N};\ D_{DE} = \sum^N D_{DE,N};\ D_{DF} = \sum^N D_{DF,N}$

The total dose from all exposure pathways for stack discharges is

$D_{STACK} = D_{PI} + D_{PE} + D_{DI} + D_{DE} + D_{DF}$

for lagoons the total dose is

$D_{LAGOON} = D_{PI} + D_{PE} + D_{DI} + D_{DE} + D_{DF}$

and for slag heaps the total dose is

$D_{SLAG} = D_{PI} + D_{PE} + D_{DI} + D_{DE} + D_{DF} + D_{DD}$

B10 Results

The generic estimated individual doses to members of the public from all atmospheric exposure scenarios are presented in Tables B4 to B21. Typical doses from stack releases are presented, for adults, 10 year old children and infants, respectively, in Tables B4, B5 and B6. The doses range from 1.2 μSv y^{-1} for adults up to 1.8 μSv y^{-1} for infants. For each age group, the majority of the dose comes from the ingestion of food grown in land contaminated by deposited radionuclides (61%, 70% and 78% respectively, for adults, children and infants). The majority of the remainder of the dose for each age group is from inhalation of radionuclides in the plume. Ingestion of nine different food groups was considered; the most significant of these was grain, followed by green vegetables and then root vegetables. The radionuclide responsible for the majority of the dose from ingestion of contaminated food is polonium-210. The doses to the generic high exposure group are also dominated by ingestion of food grown in

soil contaminated by deposited radionuclides. The doses to adults, children and infants are presented in Tables B7, B8 and B9, respectively. As for the typical individual doses, the highest doses are to infants, 88 μSv y^{-1}. Doses to children are 60 μSv y^{-1} and doses to adults are 43 μSv y^{-1}. In each case 99% of the total dose is due to ingestion of food. The most important foods are grain and sheep liver, and the radionuclide contributing most of the dose is polonium-210. Doses from the inhalation of radionuclides in the plume are similar for both the typical and high exposure groups, but the dose to the high exposure group from ingestion of foods is more than 50 times greater than the dose to typical individuals.

TABLE B4 Typical individual doses to adults from stack releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: - plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	2.4 10^{-8}	1.6 10^{-15}	6.2 10^{-11}	6.0 10^{-12}	2.4 10^{-8}	4.8 10^{-8}
^{210}Po	4.2 10^{-7}	6.7 10^{-17}	5.9 10^{-10}	2.7 10^{-14}	6.8 10^{-7}	1.1 10^{-6}
Total	4.5 10^{-7}	1.7 10^{-15}	6.5 10^{-10}	6.1 10^{-12}	7.1 10^{-7}	1.2 10^{-6}
% of total	39%	0%	0%	0%	61%	

TABLE B5 Typical individual doses to 10 year old children from stack releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External:- plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	2.6 10^{-8}	1.6 10^{-15}	6.6 10^{-11}	6.0 10^{-12}	4.3 10^{-8}	6.9 10^{-8}
^{210}Po	4.6 10^{-7}	6.7 10^{-17}	6.4 10^{-10}	2.7 10^{-14}	1.1 10^{-6}	1.6 10^{-6}
Total	4.9 10^{-7}	1.7 10^{-15}	7.1 10^{-10}	6.1 10^{-12}	1.1 10^{-6}	1.6 10^{-6}
% of total	30%	0%	0%	0%	70%	

TABLE B6 Typical individual doses to 1 year old children from stack releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	2.1 10^{-8}	1.6 10^{-15}	5.5 10^{-11}	6.0 10^{-12}	4.1 10^{-8}	6.2 10^{-8}
^{210}Po	3.7 10^{-7}	6.7 10^{-17}	5.2 10^{-10}	2.7 10^{-14}	1.3 10^{-6}	1.7 10^{-6}
Total	3.9 10^{-7}	1.7 10^{-15}	5.8 10^{-10}	6.1 10^{-12}	1.4 10^{-6}	1.8 10^{-6}
% of total	22%	0%	0%	0%	78%	

TABLE B7 Doses to high exposure group adults from stack releases (generic caluclations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$3.4\ 10^{-8}$	$2.1\ 10^{-14}$	$4.5\ 10^{-10}$	$1.3\ 10^{-10}$	$7.3\ 10^{-7}$	$7.6\ 10^{-7}$
^{210}Po	$6.0\ 10^{-7}$	$1.5\ 10^{-15}$	$4.3\ 10^{-9}$	$5.6\ 10^{-13}$	$4.1\ 10^{-5}$	$4.2\ 10^{-5}$
Total	$6.4\ 10^{-7}$	$2.3\ 10^{-14}$	$4.7\ 10^{-9}$	$1.3\ 10^{-10}$	$4.2\ 10^{-5}$	$4.3\ 10^{-5}$
% of total	1.5%	0%	0%	0%	98.5%	

TABLE B8 Doses to high exposure group 10 year old children from stack releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$3.7\ 10^{-8}$	$1.3\ 10^{-14}$	$4.8\ 10^{-10}$	$6.5\ 10^{-11}$	$1.4\ 10^{-6}$	$1.5\ 10^{-6}$
^{210}Po	$6.6\ 10^{-7}$	$9.0\ 10^{-16}$	$4.7\ 10^{-9}$	$2.9\ 10^{-13}$	$5.8\ 10^{-5}$	$5.8\ 10^{-5}$
Total	$6.9\ 10^{-7}$	$1.4\ 10^{-14}$	$5.1\ 10^{-9}$	$6.5\ 10^{-11}$	$5.9\ 10^{-5}$	$6.0\ 10^{-5}$
% of total	1%	0%	0%	0%	99%	

TABLE B9 Doses to high exposure group 1 year old children from stack releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$3.0\ 10^{-8}$	$1.3\ 10^{-14}$	$4.0\ 10^{-10}$	$6.5\ 10^{-11}$	$1.3\ 10^{-6}$	$1.4\ 10^{-6}$
^{210}Po	$5.3\ 10^{-7}$	$9.0\ 10^{-16}$	$3.8\ 10^{-9}$	$2.9\ 10^{-13}$	$8.6\ 10^{-5}$	$8.7\ 10^{-5}$
Total	$5.6\ 10^{-7}$	$1.4\ 10^{-14}$	$4.2\ 10^{-9}$	$6.5\ 10^{-11}$	$8.7\ 10^{-5}$	$8.8\ 10^{-5}$
% of total	1%	0%	0%	0%	99%	

TABLE B10 Typical individual doses to adults from lagoon releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$1.4\ 10^{-11}$	$5.9\ 10^{-19}$	$1.9\ 10^{-14}$	$1.9\ 10^{-15}$	$7.4\ 10^{-12}$	$2.1\ 10^{-11}$
^{210}Po	$9.1\ 10^{-12}$	$7.5\ 10^{-22}$	$7.1\ 10^{-15}$	$3.2\ 10^{-19}$	$8.2\ 10^{-12}$	$1.7\ 10^{-11}$
Total	$2.3\ 10^{-11}$	$5.9\ 10^{-19}$	$2.7\ 10^{-14}$	$1.9\ 10^{-15}$	$1.6\ 10^{-11}$	$3.8\ 10^{-11}$
% of total	59%	0%	0%	0%	41%	

TABLE B11 Typical individual doses to 10 year old children from lagoon releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$1.4\ 10^{-11}$	$5.9\ 10^{-19}$	$2.1\ 10^{-14}$	$1.9\ 10^{-15}$	$1.4\ 10^{-11}$	$2.8\ 10^{-11}$
^{210}Po	$9.9\ 10^{-12}$	$7.5\ 10^{-22}$	$7.7\ 10^{-15}$	$3.2\ 10^{-19}$	$1.3\ 10^{-11}$	$2.3\ 10^{-11}$
Total	$2.4\ 10^{-11}$	$5.9\ 10^{-19}$	$2.9\ 10^{-14}$	$1.9\ 10^{-15}$	$2.7\ 10^{-11}$	$5.1\ 10^{-11}$
% of total	48%	0%	0%	0%	52%	

TABLE B12 Typical individual doses to 1 year old children from lagoon releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$1.2\ 10^{-11}$	$5.9\ 10^{-19}$	$1.7\ 10^{-14}$	$1.9\ 10^{-15}$	$1.3\ 10^{-11}$	$2.5\ 10^{-11}$
^{210}Po	$8.0\ 10^{-12}$	$7.5\ 10^{-22}$	$6.3\ 10^{-15}$	$3.2\ 10^{-19}$	$1.6\ 10^{-11}$	$2.4\ 10^{-11}$
Total	$2.0\ 10^{-11}$	$5.9\ 10^{-19}$	$2.4\ 10^{-14}$	$1.9\ 10^{-15}$	$2.9\ 10^{-11}$	$4.9\ 10^{-11}$
% of total	41%	0%	0%	0%	59%	

TABLE B13 Doses to high exposure group adults from lagoon releases(generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$7.8\ 10^{-10}$	$3.9\ 10^{-17}$	$7.9\ 10^{-13}$	$2.2\ 10^{-13}$	$1.3\ 10^{-9}$	$2.1\ 10^{-9}$
^{210}Po	$5.3\ 10^{-10}$	$3.6\ 10^{-20}$	$2.9\ 10^{-13}$	$3.8\ 10^{-17}$	$2.8\ 10^{-9}$	$3.3\ 10^{-9}$
Total	$1.3\ 10^{-9}$	$3.9\ 10^{-17}$	$1.1\ 10^{-12}$	$2.2\ 10^{-13}$	$4.1\ 10^{-9}$	$5.4\ 10^{-9}$
% of total	24%	0%	0%	0%	76%	

TABLE B14 Doses to high exposure group 10 year old children from lagoon releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External:- plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	$8.4\ 10^{-10}$	$2.4\ 10^{-17}$	$8.5\ 10^{-13}$	$1.1\ 10^{-13}$	$2.5\ 10^{-9}$	$3.4\ 10^{-9}$
^{210}Po	$5.8\ 10^{-10}$	$2.2\ 10^{-20}$	$3.2\ 10^{-13}$	$1.9\ 10^{-17}$	$3.9\ 10^{-9}$	$4.5\ 10^{-9}$
Total	$1.4\ 10^{-9}$	$2.4\ 10^{-17}$	$1.2\ 10^{-12}$	$1.1\ 10^{-13}$	$6.4\ 10^{-9}$	$7.9\ 10^{-9}$
% of total	18%	0%	0%	0%	82%	

TABLE B15 Doses to high exposure group 1 year old children from lagoon releases (generic calculations)

	Dose (Sv y^{-1})					
Radionuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Total
^{210}Pb	6.9 10^{-10}	2.4 10^{-17}	7.1 10^{-13}	1.1 10^{-13}	2.4 10^{-9}	3.1 10^{-9}
^{210}Po	4.7 10^{-10}	2.2 10^{-20}	2.6 10^{-13}	1.9 10^{-17}	5.8 10^{-9}	6.3 10^{-9}
Total	1.2 10^{-9}	2.4 10^{-17}	9.6 10^{-13}	1.1 10^{-13}	8.2 10^{-9}	9.4 10^{-9}
% of total	12%	0%	0%	0%	88%	

TABLE B16 Typical individual doses to adults from stored slag (generic calculations)

	Dose (Sv y^{-1})							
Radio-nuclide	Inhalation: plume	External: plume	Inhalation: resuspension	External: deposition	Ingestion of food	Direct external	Inhalation radon	Total
^{238}U	2.1 10^{-9}	1.3 10^{-16}	3.3 10^{-12}	6.1 10^{-12}	1.9 10^{-11}	4.1 10^{-17*}		2.1 10^{-9}
^{234}U	2.5 10^{-9}	4.3 10^{-18}	4.0 10^{-12}	2.2 10^{-14}	1.9 10^{-11}			2.5 10^{-9}
^{230}Th	9.9 10^{-9}	9.9 10^{-18}	1.6 10^{-11}	3.7 10^{-12}	5.9 10^{-11}			1.0 10^{-8}
^{226}Ra	2.5 10^{-9}	1.6 10^{-16}	4.0 10^{-12}	5.0 10^{-10}	2.1 10^{-}10		1.3 10^{-8}	1.7 10^{-8}
^{232}Th	9.8 10^{-9}	2.9 10^{-18}	1.6 10^{-11}	6.6 10^{-10}	3.6 10^{-11}	2.5 $10^{-16\dagger}$		1.1 10^{-8}
^{228}Ra	1.0 10^{-9}	5.9 10^{-16}	1.3 10^{-12}	6.3 10^{-11}	1.9 10^{-10}			1.3 10^{-9}
^{228}Th	1.7 10^{-8}	2.9 10^{-17}	1.8 10^{-11}	7.5 10^{-12}	2.1 10^{-11}			1.7 10^{-8}
^{235}U	1.0 10^{-10}	1.7 10^{-16}	1.6 10^{-13}	1.7 10^{-12}	8.4 10^{-13}	1.3 $10^{-25\ddagger}$		1.0 10^{-10}
^{231}Pa	4.5 10^{-9}	3.8 10^{-17}	7.3 10^{-12}	1.9 10^{-12}	6.2 10^{-11}			4.6 10^{-9}
^{227}Ac	1.8 10^{-8}	2.2 10^{-19}	2.6 10^{-11}	3.0 10^{-12}	1.6 10^{-11}			1.8 10^{-8}
Total	6.8 10^{-8}	1.1 10^{-15}	9.6 10^{-11}	1.3 10^{-9}	6.4 10^{-10}	2.9 10^{-16}	1.3 10^{-8}	8.3 10^{-8}
% of Total	81%	0%	0%	2%	1%	0%	16%	

* Includes contributions from all members of the ^{238}U decay chain assumed to be in secular equilibrium.

† Includes contributions from all members of the ^{232}Th decay chain assumed to be in secular equilibrium.

‡ Includes contributions from all members of the ^{235}U decay chain assumed to be in secular equilibrium.

TABLE B17 Typical individual doses to 10 year old children from stored slag (generic calculations)

	Dose (Sv y^{-1})							
Radio-nuclide	Inhalation plume	External plume	Inhalation resuspension	External deposition	Ingestion of food	Direct external	Inhalation radon	Total
^{238}U	$2.2\ 10^{-9}$	$1.3\ 10^{-16}$	$3.6\ 10^{-12}$	$6.1\ 10^{-12}$	$1.8\ 10^{-11}$	$4.1\ 10^{-17*}$		$2.3\ 10^{-9}$
^{234}U	$2.7\ 10^{-9}$	$4.3\ 10^{-18}$	$4.3\ 10^{-12}$	$2.2\ 10^{-14}$	$1.7\ 10^{-11}$			$2.7\ 10^{-9}$
^{230Th}	$8.9\ 10^{-9}$	$9.9\ 10^{-18}$	$1.4\ 10^{-11}$	$3.7\ 10^{-12}$	$2.8\ 10^{-11}$			$8.9\ 10^{-9}$
^{226}Ra	$2.7\ 10^{-9}$	$1.6\ 10^{-16}$	$4.4\ 10^{-12}$	$5.0\ 10^{-10}$	$4.3\ 10^{-10}$		$1.3\ 10^{-8}$	$1.7\ 10^{-8}$
^{232}Th	$8.0\ 10^{-9}$	$2.9\ 10^{-18}$	$1.3\ 10^{-11}$	$6.6\ 10^{-10}$	$1.9\ 10^{-11}$	$2.5\ 10^{-16\dagger}$		$8.7\ 10^{-9}$
^{228}Ra	$1.4\ 10^{-9}$	$5.9\ 10^{-16}$	$1.8\ 10^{-12}$	$6.3\ 10^{-11}$	$6.6\ 10^{-10}$			$2.2\ 10^{-9}$
^{228Th}	$1.8\ 10^{-8}$	$2.9\ 10^{-17}$	$1.9\ 10^{-11}$	$7.5\ 10^{-12}$	$2.3\ 10^{-11}$			$1.8\ 10^{-8}$
^{235}U	$1.1\ 10^{-10}$	$1.7\ 10^{-16}$	$1.8\ 10^{-13}$	$1.7\ 10^{-12}$	$7.6\ 10^{-13}$	$1.3\ 10^{-25\ddagger}$		$1.1\ 10^{-10}$
^{231}Pa	$3.8\ 10^{-9}$	$3.8\ 10^{-17}$	$6.1\ 10^{-12}$	$1.9\ 10^{-12}$	$6.2\ 10^{-11}$			$3.9\ 10^{-9}$
^{227}Ac	$1.9\ 10^{-8}$	$2.2\ 10^{-19}$	$2.7\ 10^{-11}$	$3.0\ 10^{-12}$	$1.1\ 10^{-11}$			$1.9\ 10^{-8}$
Total	$6.7\ 10^{-8}$	$1.1\ 10^{-15}$	$9.4\ 10^{-11}$	$1.3\ 10^{-9}$	$1.3\ 10^{-9}$	$2.9\ 10^{-16}$	$1.3\ 10^{-8}$	$8.3\ 10^{-8}$
% of Total	80%	0%	0%	2%	2%	0%	16%	

* Includes contributions from all members of the ^{238}U decay chain assumed to be in secular equilibrium.

† Includes contributions from all members of the ^{232}Th decay chain assumed to be in secular equilibrium.

‡ Includes contributions from all members of the ^{235}U decay chain assumed to be in secular equilibrium.

TABLE B18 Typical individual doses to 1 year old children from stored slag (generic calculations)

	Dose (Sv y^{-1})							
Radio-nuclide	Inhalation plume	External plume	Inhalation resuspension	External deposition	Ingestion of food	Direct external	Inhalation radon	Total
^{238}U	$1.8\ 10^{-9}$	$1.3\ 10^{-16}$	$2.9\ 10^{-12}$	$6.1\ 10^{-12}$	$2.5\ 10^{-11}$	$4.1\ 10^{-17*}$		$1.8\ 10^{-9}$
^{234}U	$2.1\ 10^{-9}$	$4.3\ 10^{-18}$	$3.4\ 10^{-12}$	$2.2\ 10^{-14}$	$2.2\ 10^{-11}$			$2.1\ 10^{-9}$
^{230Th}	$6.6\ 10^{-9}$	$9.9\ 10^{-18}$	$1.1\ 10^{-11}$	$3.7\ 10^{-12}$	$2.3\ 10^{-11}$			$6.6\ 10^{-9}$
^{226}Ra	$2.1\ 10^{-9}$	$1.6\ 10^{-16}$	$3.4\ 10^{-12}$	$5.0\ 10^{-10}$	$2.7\ 10^{-10}$		$1.3\ 10^{-8}$	$1.6\ 10^{-8}$
^{232}Th	$5.2\ 10^{-9}$	$2.9\ 10^{-18}$	$8.5\ 10^{-12}$	$6.6\ 10^{-10}$	$1.4\ 10^{-11}$	$2.5\ 10^{-16\dagger}$		$5.9\ 10^{-9}$
^{228}Ra	$1.1\ 10^{-9}$	$5.9\ 10^{-16}$	$1.3\ 10^{-12}$	$6.3\ 10^{-11}$	$5.3\ 10^{-10}$			$1.7\ 10^{-9}$
^{228Th}	$1.5\ 10^{-8}$	$2.9\ 10^{-17}$	$1.5\ 10^{-11}$	$7.5\ 10^{-12}$	$2.9\ 10^{-11}$			$1.5\ 10^{-8}$
^{235}U	$8.5\ 10^{-11}$	$1.7\ 10^{-16}$	$1.4\ 10^{-13}$	$1.7\ 10^{-12}$	$1.0\ 10^{-12}$	$1.3\ 10^{-25\ddagger}$		$8.8\ 10^{-11}$
^{231}Pa	$2.0\ 10^{-9}$	$3.8\ 10^{-17}$	$3.2\ 10^{-12}$	$1.9\ 10^{-12}$	$3.8\ 10^{-11}$			$2.0\ 10^{-9}$
^{227}Ac	$1.4\ 10^{-8}$	$2.2\ 10^{-19}$	$2.0\ 10^{-11}$	$3.0\ 10^{-12}$	$1.1\ 10^{-11}$			$1.4\ 10^{-8}$
Total	$4.9\ 10^{-8}$	$1.1\ 10^{-15}$	$6.9\ 10^{-11}$	$1.3\ 10^{-9}$	$9.6\ 10^{-10}$	$2.9\ 10^{-16}$	$1.3\ 10^{-8}$	$6.5\ 10^{-8}$
% of Total	76%	0%	0%	2%	1%	0%	21%	

* Includes contributions from all members of the ^{238}U decay chain assumed to be in secular equilibrium.

† Includes contributions from all members of the ^{232}Th decay chain assumed to be in secular equilibrium.

‡ Includes contributions from all members of the ^{235}U decay chain assumed to be in secular equilibrium.

TABLE B19 Doses to high exposure group adults from stored slag (generic calculations)

	Dose (Sv y^{-1})							
Radio-nuclide	Inhalation plume	External plume	Inhalation resuspension	External deposition	Ingestion of food	Direct external	Inhalation radon	Total
^{238}U	$1.2\ 10^{-7}$	$6.5\ 10^{-15}$	$1.4\ 10^{-10}$	$7.2\ 10^{-10}$	$2.3\ 10^{-9}$	$7.3\ 10^{-7*}$		$8.6\ 10^{-7}$
^{234}U	$1.4\ 10^{-7}$	$3.2\ 10^{-16}$	$1.6\ 10^{-10}$	$2.6\ 10^{-12}$	$2.3\ 10^{-9}$			$1.5\ 10^{-7}$
^{230Th}	$5.7\ 10^{-7}$	$6.2\ 10^{-16}$	$6.6\ 10^{-10}$	$4.4\ 10^{-10}$	$6.2\ 10^{-9}$			$5.8\ 10^{-7}$
^{226}Ra	$1.5\ 10^{-7}$	$8.5\ 10^{-15}$	$1.7\ 10^{-10}$	$5.9\ 10^{-8}$	$3.4\ 10^{-8}$		$7.8\ 10^{-7}$	$1.0\ 10^{-6}$
^{232}Th	$5.7\ 10^{-7}$	$1.9\ 10^{-16}$	$6.5\ 10^{-10}$	$7.9\ 10^{-8}$	$3.8\ 10^{-9}$	$6.5\ 10^{-7\dagger}$		$1.3\ 10^{-6}$
^{228}Ra	$6.0\ 10^{-8}$	$2.9\ 10^{-15}$	$5.2\ 10^{-11}$	$7.4\ 10^{-9}$	$3.2\ 10^{-8}$			$9.9\ 10^{-8}$
^{228Th}	$9.9\ 10^{-7}$	$1.6\ 10^{-15}$	$7.3\ 10^{-10}$	$8.9\ 10^{-10}$	$2.0\ 10^{-9}$			$9.9\ 10^{-7}$
^{235}U	$5.8\ 10^{-9}$	$9.2\ 10^{-15}$	$6.6\ 10^{-12}$	$2.1\ 10^{-10}$	$1.0\ 10^{-10}$	$2.2\ 10^{-9\ddagger}$		$8.3\ 10^{-9}$
^{231}Pa	$2.6\ 10^{-7}$	$2.1\ 10^{-15}$	$3.0\ 10^{-10}$	$2.3\ 10^{-10}$	$1.0\ 10^{-8}$			$2.7\ 10^{-7}$
^{227}Ac	$1.1\ 10^{-6}$	$8.8\ 10^{-18}$	$1.1\ 10^{-9}$	$3.5\ 10^{-10}$	$1.9\ 10^{-9}$			$1.1\ 10^{-6}$
Total	$3.9\ 10^{-6}$	$3.2\ 10^{-14}$	$3.9\ 10^{-9}$	$1.5\ 10^{-7}$	$9.5\ 10^{-8}$	$1.4\ 10^{-6}$	$7.8\ 10^{-7}$	$6.3\ 10^{-6}$
% of Total	62%	0%	0%	2%	2%	22%	12%	

* Includes contributions from all members of the ^{238}U decay chain assumed to be in secular equilibrium.

† Includes contributions from all members of the ^{232}Th decay chain assumed to be in secular equilibrium.

‡ Includes contributions from all members of the ^{235}U decay chain assumed to be in secular equilibrium.

TABLE B20 Doses to high exposure group 10 year old children from stored slag (generic calculations)

	Dose (Sv y^{-1})							
Radio-nuclide	Inhalation plume	External plume	Inhalation resuspension	External deposition	Ingestion of food	Direct external	Inhalation radon	Total
^{238}U	$1.3\ 10^{-7}$	$3.9\ 10^{-15}$	$1.5\ 10^{-10}$	$3.7\ 10^{-10}$	$2.5\ 10^{-9}$	$3.7\ 10^{-7*}$		$5.1\ 10^{-7}$
^{234}U	$1.5\ 10^{-7}$	$1.9\ 10^{-16}$	$1.8\ 10^{-10}$	$1.3\ 10^{-12}$	$2.5\ 10^{-9}$			$1.6\ 10^{-7}$
^{230Th}	$5.1\ 10^{-7}$	$3.7\ 10^{-16}$	$5.9\ 10^{-10}$	$2.2\ 10^{-10}$	$3.4\ 10^{-9}$			$5.2\ 10^{-7}$
^{226}Ra	$1.6\ 10^{-7}$	$5.1\ 10^{-15}$	$1.8\ 10^{-10}$	$3.0\ 10^{-8}$	$7.4\ 10^{-8}$		$7.8\ 10^{-7}$	$1.0\ 10^{-6}$
^{232}Th	$4.7\ 10^{-7}$	$1.1\ 10^{-16}$	$5.3\ 10^{-10}$	$4.0\ 10^{-8}$	$2.3\ 10^{-9}$	$3.3\ 10^{-7\dagger}$		$8.4\ 10^{-7}$
^{228}Ra	$8.3\ 10^{-8}$	$1.8\ 10^{-15}$	$7.2\ 10^{-11}$	$3.8\ 10^{-9}$	$1.2\ 10^{-7}$			$2.1\ 10^{-7}$
^{228Th}	$1.1\ 10^{-6}$	$9.6\ 10^{-16}$	$7.8\ 10^{-10}$	$4.5\ 10^{-10}$	$2.6\ 10^{-9}$			$1.1\ 10^{-6}$
^{235}U	$6.3\ 10^{-9}$	$5.5\ 10^{-15}$	$7.2\ 10^{-12}$	$1.0\ 10^{-10}$	$1.1\ 10^{-10}$	$1.1\ 10^{-9\ddagger}$		$7.6\ 10^{-9}$
^{231}Pa	$2.2\ 10^{-7}$	$1.3\ 10^{-15}$	$2.5\ 10^{-10}$	$1.2\ 10^{-10}$	$1.1\ 10^{-8}$			$2.3\ 10^{-7}$
^{227}Ac	$1.1\ 10^{-6}$	$5.3\ 10^{-18}$	$1.1\ 10^{-9}$	$1.8\ 10^{-10}$	$1.5\ 10^{-9}$			$1.1\ 10^{-6}$
Total	$3.9\ 10^{-6}$	$1.9\ 10^{-14}$	$3.8\ 10^{-9}$	$7.5\ 10^{-8}$	$2.2\ 10^{-7}$	$7.0\ 10^{-7}$	$7.8\ 10^{-7}$	$5.7\ 10^{-6}$
% of Total	68%	0%	0%	1%	4%	13%	14%	

* Includes contributions from all members of the ^{238}U decay chain assumed to be in secular equilibrium.

† Includes contributions from all members of the ^{232}Th decay chain assumed to be in secular equilibrium.

‡ Includes contributions from all members of the ^{235}U decay chain assumed to be in secular equilibrium.

TABLE B21 Doses to high exposure group 1 year old children from stored slag (generic calculations)

	Dose (Sv y^{-1})							
Radio-nuclide	Inhalation plume	External plume	Inhalation resuspension	External deposition	Ingestion of food	Direct external	Inhalation radon	Total
^{238}U	$1.0\ 10^{-7}$	$3.9\ 10^{-15}$	$1.2\ 10^{-10}$	$3.7\ 10^{-10}$	$3.7\ 10^{-9}$	$3.7\ 10^{-7*}$		$4.8\ 10^{-7}$
^{234}U	$1.2\ 10^{-7}$	$1.9\ 10^{-16}$	$1.4\ 10^{-10}$	$1.3\ 10^{-12}$	$3.3\ 10^{-9}$			$1.2\ 10^{-7}$
^{230}Th	$3.8\ 10^{-7}$	$3.7\ 10^{-16}$	$4.4\ 10^{-10}$	$2.2\ 10^{-10}$	$2.7\ 10^{-9}$			$3.8\ 10^{-7}$
^{226}Ra	$1.2\ 10^{-7}$	$5.1\ 10^{-15}$	$1.4\ 10^{-10}$	$3.0\ 10^{-8}$	$4.8\ 10^{-8}$		$7.8\ 10^{-7}$	$9.8\ 10^{-7}$
^{232}Th	$3.0\ 10^{-7}$	$1.1\ 10^{-16}$	$3.5\ 10^{-10}$	$4.0\ 10^{-8}$	$1.7\ 10^{-9}$	$3.3\ 10^{-7\dagger}$		$6.8\ 10^{-7}$
^{228}Ra	$6.2\ 10^{-8}$	$1.8\ 10^{-15}$	$5.3\ 10^{-11}$	$3.8\ 10^{-9}$	$9.7\ 10^{-8}$			$1.6\ 10^{-7}$
^{228}Th	$8.4\ 10^{-7}$	$9.6\ 10^{-16}$	$6.2\ 10^{-10}$	$4.5\ 10^{-10}$	$3.1\ 10^{-9}$			$8.4\ 10^{-7}$
^{235}U	$4.9\ 10^{-9}$	$5.5\ 10^{-15}$	$5.7\ 10^{-12}$	$1.0\ 10^{-10}$	$1.5\ 10^{-10}$	$1.1\ 10^{-9\ddagger}$		$6.3\ 10^{-9}$
^{231}Pa	$1.1\ 10^{-7}$	$1.3\ 10^{-15}$	$1.3\ 10^{-10}$	$1.2\ 10^{-10}$	$6.6\ 10^{-9}$			$1.2\ 10^{-7}$
^{227}Ac	$8.2\ 10^{-7}$	$5.3\ 10^{-18}$	$8.3\ 10^{-10}$	$1.8\ 10^{-10}$	$1.6\ 10^{-9}$			$8.2\ 10^{-7}$
Total	$2.9\ 10^{-6}$	$1.9\ 10^{-14}$	$2.8\ 10^{-9}$	$7.5\ 10^{-8}$	$1.7\ 10^{-7}$	$7.0\ 10^{-7}$	$7.8\ 10^{-7}$	$4.6\ 10^{-6}$
% of Total	62%	0%	0%	2%	4%	15%	17%	

* Includes contributions from all members of the ^{238}U decay chain assumed to be in secular equilibrium.

† Includes contributions from all members of the ^{232}Th decay chain assumed to be in secular equilibrium.

‡ Includes contributions from all members of the ^{235}U decay chain assumed to be in secular equilibrium.

The individual doses resulting from releases from lagoons are presented in Tables B10 to B15. The doses to typical individuals are 5 orders of magnitude lower than the corresponding doses from stack releases. Typical individual doses to adults are $3.8\ 10^{-5}$ μSv y^{-1}, the doses to children are $5.1\ 10^{-5}$ μSv y^{-1} and the doses to infants are $4.9\ 10^{-5}$ μSv y^{-1}. The two most significant pathways are the ingestion of food grown in land contaminated by deposited radionuclides and inhalation of radionuclides in the plume. Polonium-210 and lead-210 contribute roughly equally to the total dose from each pathway. The doses to members of the high exposure' group are 4 orders of magnitude lower than the corresponding doses from stack releases; doses to adults are $5.4\ 10^{-3}$ μSv y^{-1}, doses to children are $7.9\ 10^{-3}$ μSv y^{-1} and doses to infants are $9.4\ 10^{-3}$ μSv y^{-1}. The majority of the dose to members of the high exposure group is from ingestion of food grown in land contaminated by deposited radionuclides. The doses from atmospheric releases from lagoons are many orders of magnitude lower than those from stack releases because the releases of radioactivity from the lagoons are so much less.

The individual doses resulting from releases from slag heaps are presented in Tables B16 to B21. A larger number of radionuclides are released from the slag heaps compared to those released from the lagoons and the stack. The physical dimensions of the slag heaps also made it necessary to consider two extra exposure pathways; direct external irradiation from the slag heap and the inhalation of radon. The doses are lower than those from stack releases but higher than those resulting from releases from lagoons. This follows, since the activity released from the slag heaps is greater than that released from lagoons but less than releases from the stack. The doses to typical individuals are

8.3 10^{-2} μSv y^{-1} to adults and children, and 6.5 10^{-2} μSv y^{-1} to infants. Around 80% of the dose to each of the three age groups is from the inhalation of radionuclides in the plume, with the majority of the remainder of the dose from inhalation of radon. The doses to members of the high exposure group are 6.3 μSv y^{-1} to adults, 5.7 μSv y^{-1} to children and 4.6 μSv y^{-1} to infants. Again the dominant pathway is the inhalation of radionuclides in the plume; this contributes around 60% of the total dose. For the high exposure group, direct external irradiation from the slag heap is also a significant pathway, contributing 22% to the adult dose, 12.5% to the dose to children and 15% to the dose to infants. Inhalation of radon is the only other significant pathway for all three age groups.

It should be stressed that the characteristics of the two generic exposure groups are not site specific but are generic assumptions applicable to the UK. The estimated doses will therefore differ from those that would be determined using site-specific information on locations of habitation and agricultural practices around a site. They are intended simply to facilitate comparison of radiological impacts with other industries.

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APPENDIX C

DOSES AND RISKS FROM THE LANDFILL DISPOSAL OF STEEL WASTES

C1 Introduction

Doses and risks arising from the disposal of wastes from the steel manufacturing industry have been estimated in this study by scaling the results from a study previously conducted by NRPB to develop a methodology for the assessment of the consequences of landfill disposal of radioactive material. The NRPB "landfill study" is described briefly below.

Following the disposal of radioactive material in the ground there are two scenarios which could lead to exposure of the population. The most likely occurrence is the gradual migration of radionuclides with ground water from the waste, through the surrounding rock and soil (geosphere), into the local environment (biosphere). From here, people could be exposed by a wide variety of routes including direct external irradiation from contaminated soil and the consumption of contaminated food. This type of exposure has a probability of occurrence close to unity, although there may be some uncertainty in the precise magnitude and time of occurrence of the doses. The second scenario includes those routes of a more probabilistic nature and involves events that disturb the natural evolution of the site. This has a probability of occurrence less than unity as not all landfill sites are used for the same purpose once the site has ceased to be used for landfill. One example is excavation of the site for development.

Since these exposures would arise in the future, predictive mathematical modelling is required to estimate the doses and risks to those people most likely to be exposed. Where there is no specific characterisation of the landfill site it is common practice to make generic assumptions about the site to determine parameter values for modelling. These assumptions are based on past experience of similar situations. In developing a general methodology for the assessment of individual risks from the disposal of radionuclides in landfill sites the following scenarios for the exposure of members of the public were considered:

a migration;
b borehole water extraction;
c excavation during development of the site; and
d residence on the site.

The exposure pathways considered for each scenario are presented in Table C1. For the determination of collective doses from terrestrial pathways consumption of the following were considered:

a drinking water;
b freshwater fish;
c green vegetables;
d grain;
e root vegetables; and
f animal foodstuffs.

For the calculation of collective doses from marine dispersion the following pathways were considered:

a fish consumption;
b crustacea consumption;
c mollusc consumption;
d seaweed consumption;
e external γ-irradiation (beach occupancy);
f inhalation of airborne sediment; and
g inhalation of seaspray.

TABLE C1 Exposure pathways for each scenario

Scenario	Exposure pathways
Migration	External irradiation from contaminated soil or river sediments
	Inhalation of wind blown (or mechanically disturbed) dust from contaminated soils or sediments
	Ingestion of contaminated water and aquatic animals, plants grown on contaminated soil, products from animals feeding on plants grown in contaminated soil or (inadvertently) contaminated soil itself
Borehole extraction of water	Drinking contaminated water
Excavation of site	External irradiation by waste
	Inhalation of resuspended radionuclides
	Inadvertent ingestion of waste
Residence on site following redevelopment	External irradiation from waste
	Inhalation of resuspended radionuclides
	Inadvertent ingestion of waste
	Consumption of vegetables grown in contaminated soil

As part of the development of the NRPB landfill methodology a review of landfill disposal sites in the UK was carried out to determine suitable characteristics for a set of representative generic site types to be defined. An extensive review of landfills currently in use was undertaken, with emphasis on a small number of key parameters found in previous work to have the most influence on the risks from disposal. The aim was to characterise, as well as possible, the general types of landfill operational in the UK. The particular geology, dimensions and level of containment of around 50 landfills were compared, and four broad categories identified.

The principal output from the landfill study was a set of doses and risks from the above scenarios for unit disposals of a large number of radionuclides to each of

these four generic landfill types. For the study the modelling of the migration of radionuclides in the geosphere was performed using GEOS (Hill, 1989), a simple one-dimensional geosphere migration code developed by NRPB which models the transport of radionuclides with ground water, taking account of advection, dispersion/diffusion, radioactive decay and sorption. Transfer in the biosphere was modelled using BIOS (Martin et al, 1991), a compartmental model developed at NRPB that represents the transfer of radionuclides through deep and surface soils, rivers and seas, and provides estimates of doses for the exposure pathways outlined above. Individual doses and risks per unit disposal from all the above scenarios were determined for each of the four landfill categories. Collective doses per unit disposal for the migration scenario were also evaluated.

The unit disposal results were used in this study to provide an estimate of the overall doses and risks associated with the disposal of steel industry waste. There were three main steps in this process. The first involved identifying the disposal inventory. This is discussed in Section C2. The second step was to identify which of the four generic landfill site types is most similar to sites where steel industry waste disposal would take place. This is described in Section C3. The final step was to scale the unit disposal results for the appropriate landfill site type according to the inventory and then sum over radionuclides to produce total doses and risks. This is described in Sections C4 and C5.

C2 Source term definition

Various waste streams from the steel production plant are sent to landfill. Dust from the sinter plant gas cleaning system is disposed directly to landfill. The waste from the blast furnace and BOS furnace wet gas cleaning systems that is not recycled is first collected and de-watered in lagoons before disposal to landfill. For landfill disposal, both of these sources of wastes are considered together.

The activity concentrations of lead-210 and polonium-210 in the collected dust have been measured (Giles and Harvey, 2000) and it was found that the activity concentrations are below the exemption limit given in the Phosphatic Substances, Rare Earths etc. Exemption Order (RSEO, 1962) so no authorisation for disposal is required.

The activity concentrations of lead-210 and polonium-210 in the collected dust sent to landfill show less enhancement, with respect to those in the raw materials, than those in the dust released to atmosphere from the sinter plant and blast furnace. There are a number of reasons for this. Enhancement is most pronounced on the finest particles (these have higher surface area to volume ratios), which are those most likely to escape to atmosphere. The concentrations in the collected dust are also effectively diluted by lower activity dust from the BOS furnace wet gas cleaning system. The activity concentrations of lead-210

and polonium-210 in the waste sent to landfill are, however, still enhanced in relation to their concentrations in the raw materials.

It had been assumed that the only significant radionuclides are lead-210 and polonium-210. Therefore other radionuclides in the uranium-238 and thorium-232 natural decay chains have not been measured. This assumption may, however, be optimistic for landfill disposal because radionuclides higher up the uranium-238 decay series would result in much higher doses following disposal. For example, the dose per unit disposal from uranium-238 and all of its daughters in secular equilibrium is two orders of magnitude greater than the dose from lead-210 and all of its daughters in secular equilibrium.

In order to scope the radiological impact of landfill disposal of the wastes the assessment considered the doses and risks resulting from two inventory assumptions:

Inventory I - waste contains only lead-210 and polonium-210; and

Inventory II - waste contains radionuclides from the uranium-238, thorium-232 and uranium-235 decay chains. With all members of the uranium-238 decay chain having the same activity concentration as lead-210. It was also assumed that radionuclides in the uranium-235 decay chain are present at the natural activity ratio of 0.045 of uranium-238. Slag from the blast furnace has activity concentrations of uranium-238 and thorium-232 of 88 Bq kg^{-1} and 49 Bq kg^{-1}, respectively. As an approximation, it was therefore assumed that radionuclides in the thorium-232 decay chain were present in the material sent to landfill with half the activity concentration of uranium-238.

The first assumption (inventory I) is expected to result in a slight underestimate of the doses that a member of the public may receive as a result of landfill disposal of waste from steel production, as some of the other radionuclides will be present, albeit perhaps at low concentrations. The other (inventory II) is extremely conservative as it assumes enhanced levels for all the other radionuclides. It is anticipated that the concentrations of these other radionuclides will be found to be closer to those in the raw materials (ie a factor of approximately 40 lower), however, this is difficult to confirm without additional measurement data. The use of inventory II therefore effectively scopes the possible range of risks. The activity concentrations used in the study to assess both options are given in Table C2.

TABLE C2 Summary of radionculide concentrations and inventories for steel waste disposed to landfill

	Radionuclide concentration (Bq kg^{-1})*		Radionuclide inventory (GBq y^{-1})*	
Radionuclide	Inventory I	Inventory II	Inventory I	Inventory II
^{210}Pb and ^{210}Po	9.00 10^2	9.00 10^2	51.3	51.3
^{238}U series		9.00 10^2		51.3
^{232}Th series		4.50 10^2		25.7
^{235}U series		4.05 10^1		2.31

*Radionuclide concentration and inventory apply to each radionuclide in the decay series

C3 Choice of generic landfill type

Four broad categories of landfill site were identified and these are summarised below.

Type A ('dilute and disperse')

This type reflects the older style of landfill, typically built without containment measures in a fairly permeable sub-soil, and often with very large capacity. Examples of this type of landfill are still in existence, although newer phases to these landfills will be built to more rigorous standards. Engineering controls, such as lining or capping, were assumed not to be present at this type of site, and the landfill was assumed to be quite close to local aquifers and water courses.

Type B ('deep quarry')

The second class encompasses the group of deeper landfills, typically disused quarries. Although some level of containment may be in place, a number of confounding factors affect such landfills. The quarry will usually have steep sides, making thorough lining difficult. As such quarries are deep the poor lining on the sides of the landfill may also mean that leachate and water can leak out laterally. The type of subsoil or rock is obviously very variable, and may range from very permeable and/or fractured rocks to relatively impermeable clay. For the purposes of assessment, this category was assumed to be moderately permeable.

Type C ('natural containment')

The nature of the sub-soil or rock (the geosphere) can be of more importance than the level of engineered containment, as the path of contaminants through, say, a natural layer of clay several metres thick is clearly longer than any plastic or clay lining. This importance has been recognised by the inclusion of a class of landfill which is in a clayey subsoil, providing a 'natural' level of containment. Most examples of such landfills tend to be more recent, and are often smaller than Type A or B.

Type D ('engineered containment')

The final type of landfill considered is typical of that with a high level of engineered containment and thoughtful siting (in a low permeability subsoil, away from aquifers). This type was intended to be representative of the best modern practice used in the UK.

The landfill sites used by Corus are generally purpose built and fully engineered (Giles and Harvey, 2000). At Scunthorpe, for example, two landfills are in use: one with a lifetime of ~25-30 years and one with a remaining lifetime of 50 years. When these landfills are full they will be capped with clay and topsoil and limited landscaping will take place before they are returned to heathland. On the basis of this information the landfills used to dispose of wastes from steel production are best characterised as Type D. Characteristics of the Type D landfill and other important model parameters are presented in Tables C3-C6.

TABLE C3 Characteristics of generic landfill Type D

Parameter	Type D*
Number of cases reviewed	16
Volume (m^3)	$2.00\ 10^6$ ($1.00\ 10^5$ to $4.90\ 10^6$)
Area (ha)	$2.00\ 10^1$ ($2.00\ 10^0$ to $7.00\ 10^1$)
Average depth (m)	$1.00\ 10^1$ ($4.30\ 10^0$ to $1.00\ 10^1$)
Waste density (t m^{-3})†	$7.50\ 10^{-1}$ ($5.80\ 10^{-1}$ to $8.00\ 10^{-1}$)
Lifetime (y)‡	15§

* The numbers in brackets show the maximum and minimum reviewed values.

† Density at the time of disposal is presented.

‡ The lifetime estimate is based on an assumed waste input of $1\ 10^5$ t y^{-1} independent of facility

§ Assumed value.

TABLE C4 Characteristics of the geosphere for generic landfill Type D

Layer	Type D*
1	Waste
2	Reworked clay lining (1 m)
3	Clayey subsoil (5 m)
4	Sandy aquifer† (1000 m)

*The numbers in brackets give the distance that it has been assumed that contaminated water travels in each layer. The contaminated ground water always flows to a higher layer number (ie. 1 to 2, 2 to 3 etc).

†The aquifer layer discharges into the biosphere, either to soil or river.

TABLE C5 Parameter values for migration scenario

Parameter	Value
Aquifer velocity (m y^{-1})	100
Aquifer dispersive length (m)	50
Clay layer groundwater velocity (m y^{-1})	0.1
Clay layer dispersive length (m)	1.0
K_D multiplier for sorbed nuclides	1.0
K_D for unsorbed nuclides	0.0
Fraction to river water	0.5
River flow rate (m^3 y^{-1})	1.0 10^7
River water velocity (m y^{-1})	5.0 10^6
River length (m)	1.0 10^3
River depth (m)	2.0
Suspended sediment load (t m^{-3})	1.0 10^{-5}
Annual depth of irrigation (m y^{-1})	0.1
Area affected (km^2)	0.5

TABLE C6 Summary of data for site excavation and residence scenarios

Parameter	Excavation of site	Residence on site
Time spent indoors (h y^{-1})	0	7884
Total time spend outdoors (h y^{-1})	2000	876
Time for low activity work outdoors (h y^{-1})	1800	866
Time for high activity work outdoors (h y^{-1})	200	10
Shielding factor indoors	-	0.1
Airborne dust loading for low activity work (mg m^{-3})	0.1	0.1
Airborne dust loading for high activity work (mg m^{-3})	10	10
Breathing rate for low activity work (m^3 h^{-1})	1.18	0.83
Breathing rate for high activity work (m^3 h^{-1})	1.69	1.69
Rate of inadvertent ingestion of soil (whilst outside) (mg h^{-1})	5	5
Rate of ingestion of root vegetables (kg y^{-1})	-	60
Rate of ingestion of green vegetables (kg y^{-1})		35

C4 Determination of doses and risks to members of the public

Individual doses and risks to members of the public were estimated for four different scenarios. These are described below.

C4.1 Individual doses from excavation and residence

These scenarios assume that once the landfill site has been completed it is available for the construction of residential buildings. Two exposed groups are considered: excavation workers and residents living on the site. Exposure during

construction will be mainly due to exposure to any dust created during the excavation of foundations, when disturbance of the underlying waste may occur, even though the site is capped with soil. In order to determine doses to these groups using the unit disposal doses generated for the NRPB landfill methodology study it was necessary to scale the doses by the activity concentration of the radionuclides in the steel industry waste. Individual doses to workers excavating the site and residents living on the site were estimated using the following equation:

$$D_{scen} = \sum_R \frac{DUD_{scen,R}}{CONC_{RER}} \times AC_R$$

where D_{scen} = Annual dose to excavation workers or members of the public resident on the site in the future (Sv y^{-1})

$DUD_{scen,R}$ = Annual dose to excavation workers or residents per unit disposal of radionuclide R (Sv y^{-1}) from the landfill methodology for landfill Type D

AC_R = Activity concentration of the material disposed to landfill, (Bq kg^{-1}), for radionuclide R, from Table C2

$CONC_{RER}$ = Concentration of radionuclide R in the waste assumed in the landfill methodology (Bq kg^{-1})

$CONC_{RER}$ is calculated using the following:

$$CONC_{RER} = \frac{Act_P}{VOL \times Rho}$$

where Act_P = Activity disposed to landfill assumed in the landfill methodology, 10^{10} Bq (10^9 Bq y^{-1} for 10 years)

VOL = Volume of generic landfill Type D, 2 10^6 m^3 (see Table C3)

Rho = Density of the material disposed, assumed in the landfill methodology, 750 kg m^{-3}, (see Table C3)

C4.2 Individual doses from migration and consumption of well water

The following equation was used to determine doses to members of the public resulting from the migration of radionuclides from the landfill to the biosphere and the consumption of water from a well on the site. The doses and risks per unit activity disposed, determined for the landfill methodology, are scaled by the total activity disposed per year to the steel industry landfill site.

$$D_{scen} = \sum_R \frac{DUD_{scen,R}}{Act_p} \times A_R$$

where D_{scen} = Annual dose to the public from the well or migration scenarios (Sv y^{-1})

$DUD_{scen,R}$ = Annual dose to the public per unit disposal of radionuclide R for the well or migration scenarios for landfill Type D (Sv y^{-1})

A_R = Activity of radionuclide R disposed to landfill (Bq)

Act_P = Activity disposed to landfill assumed in the landfill methodology, 10^{10} Bq

A_R is calculated using the following:

$$A_R = AC_R \times MASS$$

where AC_R = Activity concentration of the material disposed to landfill (Bq kg^{-1}), for radionuclide R, from Table C2

MASS = Mass of material sent to landfill each year by a single steel works, 5.7 10^7 kg (Giles and Harvey, 2000)

C4.3 Determination of individual risks

The risks to members of the public from the above scenarios were calculated using the following equation:

$$R_T = D_T \times P_T \times RC$$

where T = Selected scenario (excavation, residence, well or migration)

R_T = Individual risk from scenario T (y^{-1})

D_T = Annual individual dose from scenario T (Sv y^{-1})

P_T = Probability that scenario T will occur, 1.0 y^{-1} for migration, 6 10^{-4} y^{-1} for redevelopment, 4 10^{-6} y^{-1} for a well

RC = Risk coefficient for fatal cancer and serious hereditary effects, 0.06 Sv^{-1} (ICRP, 1990)

C4.4 Collective doses to members of the public

Collective doses from the disposal of steel industry waste were determined by scaling the doses determined using the landfill methodology, using the following equation:

$$D_{Coll} = \sum_R \frac{DUD_{Coll,R}}{Act_P} \times AC_R \times Mass$$

where D_{Coll} = Collective dose to the UK population (manSv)

$DUD_{Coll,R}$ = Collective dose per unit disposal of radionuclide R to landfill Type D (manSv)

AC_R = Activity concentration of radionuclide R in material sent to landfill (Bq kg^{-1}), from Table C2

Mass = Mass of waste sent to landfill per year from all UK steel plants, 2.0 10^8 kg (Giles and Harvey, 2000)

Act_P = Activity assumed to be disposed in the landfill methodology, 10^{10} Bq

C5 Determination of doses and risks to landfill workers

The NRPB landfill study also involved the development of a methodology and the identification of associated data for the assessment of doses to landfill workers. It was assumed that the landfill operator distributes waste within the site using mechanical excavators. The methodology assumed that landfill workers would be exposed during a working year (2000 hours per year) via the following exposure pathways: external irradiation, inhalation of contaminated dust, contamination of the skin, and inadvertent ingestion of dust. It was assumed that the radioactive waste disposed to the site in any one year was mixed evenly within the total mass of waste disposed per year. Doses from all the exposure pathways considered are proportional to the concentration of activity in the waste. The NRPB landfill methodology study determined doses to landfill workers per unit disposal. The doses to landfill workers in this study were determined by scaling these doses by the activity concentrations in the steel industry wastes. From discussions with Corus (Giles and Harvey, 2000), it has been established that the assumptions about landfill work practices are reasonable for workers at steel waste landfill sites. The doses and risks to landfill workers at sites where steel production waste is disposed were therefore determined using the following equations:

$$D_{LW} = \sum_R \frac{DUD_{LW,R}}{CONC_{RW}} \times AC_R$$

where D_{LW} = Annual dose to landfill workers (Sv y^{-1})

$DUD_{LW,R}$ = Annual dose to landfill worker per unit disposal of radionuclide R from landfill methodology study (Sv y^{-1})

AC_R = Activity concentration of the waste disposed to landfill (Bq kg^{-1}), for radionuclide R, from Table C2

$CONC_{RW}$ = Concentration of radionuclide R in the waste (Bq kg^{-1}) assumed in the landfill methodology.

$CONC_{RW}$ was determined using the following:

$$CONC_{RW} = \frac{Act_W}{MASS}$$

where Act_W = Activity disposed per year assumed in the landfill methodology, 10^9 Bq

MASS = Mass of waste sent to landfill per year assumed in the landfill methodology, 10^8 kg.

C6 Results

C6.1 Members of the public

Estimated doses and risks to members of the public from the disposal of steel industry waste to landfill, for the two disposal inventories described in Section C2, are presented in Tables C7 and C8. For disposal inventory I (measured activity concentrations of lead-210 and polonium-210) the estimated peak dose is 140 μSv y^{-1}, from the residence scenario. The estimated doses from the migration and ingestion of well water scenarios are effectively zero because of the long time taken for radionuclides to migrate to the biosphere in comparison with the relatively short half lives of lead-210 and polonium-210. The peak individual risk is 5.2 10^{-9} y^{-1}, again from the residence scenario.

TABLE C7 Doses to the public from landfill disposal of steel production waste

	Dose (Sv y^{-1})				
Radionuclide inventory	Excavation	Residence*	Ingestion of well water	Migration	Peak migration time (y)
Inventory I	3.0 10^{-6}	1.4 10^{-4}	-	-	-
Inventory II	2.5 10^{-4}	3.8 10^{-4}	3.8 10^{-7}	2.5 10^{-7}	5.0 10^{5}

* Dose without contribution from radon

TABLE C8 Risks to the public from landfill disposal of steel production waste

	Risk (y^{-1})			
Radionuclide inventory	Excavation	Residence*	Ingestion of well water	Migration
Inventory I	1.0 10^{-10}	5.2 10^{-9}	-	-
Inventory II	9.0 10^{-9}	1.4 10^{-8}	9.1 10^{-14}	1.5 10^{-8}

*Risk without contribution from radon

Disposal inventory II includes radionuclides with much longer half-lives than of lead-210 and polonium-210. In this case, the time required for all radionuclides to decay to an insignificant level is greater than the time taken for radionuclides to leach from the landfill site with groundwater. Hence, doses and risks from migration and ingestion of well water are non zero. For disposal inventory II the estimated peak dose is 380 μSv y^{-1}, from residence on the redeveloped landfill site. With the associated probability of occurrence this gives a risk of 1.4 10^{-8} y^{-1}. The maximum individual risk from disposal inventory II is 1.5 10^{-8} y^{-1}, from the migration scenario. The estimated dose from this scenario is 0.25 μSv y^{-1}. For migration, the peak risk arises between 10^{5} and 10^{6} years following disposal. It is generally considered that calculations of individual risk beyond about 10,000 years can only provide an *indication* of the possible level of risk rather than a *prediction* of the risk (NRPB, 1992). From an examination of the results of the

landfill study, it is anticipated that the migration risks to 10,000 years would be several orders of magnitude lower. Individual risks to 10,000 years are therefore dominated by those from the residence scenario ($1.4\ 10^{-8}\ y^{-1}$). As was discussed earlier, inventory I is expected to result in a slight underestimate of doses and risks. Inventory II, however, was derived using conservative assumptions to scope the possible range of risks. The actual peak risk is therefore expected to be within the range of the two predictions, closer to that of Inventory I. The difference between the two predictions is, however, only a factor of 2 or 3.

One exposure pathway that has not been considered in the above analysis is the inhalation of radon, which may escape from the steel waste and diffuse through the surrounding waste and soil. For the majority of the exposure scenarios considered above, which involve exposure outdoors, the radon would be rapidly dispersed and therefore inclusion of this exposure pathway would have a minor impact upon the estimated doses and risks. Doses to members of the public in homes constructed on steel waste disposal sites may, however, be non-trivial due to the build up of concentrations of the gas in the home. One study (Green, 1986) estimated that individual doses to residents in homes built on ash disposal sites covered with 50 cm of topsoil would be approximately 100 $\mu Sv\ y^{-1}$ greater than the doses to residents living in homes built on normal ground. The concentration of radium-226 in ash assumed in (Green, 1986) was 100 $Bq\ kg^{-1}$. Inventory I assumes zero concentration of radium-226, and so for this inventory estimated doses from seepage of radon from the waste are also clearly zero. Inventory II assumes a radium-226 concentration of 900 $Bq\ kg^{-1}$. Assuming that radon would be released from and transported through the steel wastes in a similar manner to the ash would lead to a estimated dose from radon in the region of 0.9 mSv. This is significant in comparison with the other pathways considered and also with general background exposures. It should, however, be recognised that inventory II is pessimistic and intended to represent a bounding case. The actual doses received from this pathway are expected to be significantly lower. It must also be remembered that the concentrations of radon in dwellings (and hence doses) can vary substantially even if the concentration of radium-226 in the underlying soil and rock is the same. The following factors influence the amount of radon entering buildings from the ground: entry routes into homes and under pressure of homes. These factors vary greatly from one dwelling type to another and could lead to large differences in radon concentrations.

The estimated collective doses truncated at 500* years and 10,000 years to members of the public in the UK from the disposal of steel industry waste are

* When using collective dose as a measure of total radiation-induced health detriment, it has been suggested that the dose truncated at 500 years should be used (Barraclough at al, 1996). A similar conclusion has been reached by ICRP (1997). The greater the integration time, the greater will be the uncertainty surrounding a number of the assumptions made in the calculation of collective doses. Amongst the assumptions that may be affected at long time frames are those associated with human behaviour, population size and the environment.

effectively zero, due to the long time taken for radionuclides to migrate from the landfill site to the biosphere.

C6.2 Landfill workers

Doses to landfill workers are presented in Table C9 for the two disposal inventories described in Section C2. The maximum estimated dose resulting from the disposal of inventory I (measured activity concentrations of lead-210 and polonium-210) is 11 μSv y^{-1}. The dominant pathway for exposure to the landfill operator is inadvertent ingestion of contaminated dust from the landfill site. The maximum estimated dose from the disposal of inventory II is 1.7 mSvy^{-1}. For this disposal inventory, the dominant pathway for exposure of the landfill operator is external irradiation from the waste. As was discussed earlier, Inventory I is expected to result in a slight underestimate of doses. Inventory II, however, was derived using very conservative assumptions. The actual dose is expected to lie in the range between them, closer to that for Inventory I. If, for example, the concentrations of the radionuclides other than lead-210 and polonium-210 were similar to those in the raw materials then the dose would by a few tens of microsieverts per year. It is recommended that measurements of these radionuclides are undertaken to refine these predictions.

TABLE C9 Doses to landfill workers

Radionuclide Inventory	Dose (Sv y^{-1})
Inventory I	1.1 10^{-5}
Inventory II	1.7 10^{-3}

C6.3 Comparison with other work

There are few studies on the radiological impact of the disposal of wastes from the steel industry reported in the literature. A previous assessment of the radiological impact of the steel production industry has, however, been undertaken at NRPB (Mayall et al, 1997). This study determined doses from the landfill disposal of unit activities of the radionuclides lead-210 and polonium-210, for scenarios similar to those studied here. In order to compare the results with the current work, the unit disposal doses were scaled by the measured activity concentrations of lead-210 and polonium-210 used for this assessment. The comparison of the results is shown in Table C10.

TABLE C10 Comparison of results for inventory I with previous NRPB study (Mayall at al, 1997)

	Previous study (Mayall et al, 1997)	Present work
Scenario	Dose (μSv y^{-1})	Dose (μSv y^{-1})
Excavation	2.1 10^{0}	3.0 10^{0}
Residence	9.3 10^{-1}	1.4 10^{2}
Migration	-	-

It can be seen that there is good agreement between the results for the excavation and migration scenarios. In both studies the migration scenario produces a negligible risk. There are however, differences in the results for the residence scenario. In the present study the estimated dose arising from residence on a redeveloped landfill site is 2 orders of magnitude greater than the dose estimated by Mayall et al (1997). The differences in the results are due to differences in modelling between the two studies. The study by Mayall et al (1997) calculated doses to residents only when they were outside, whilst the current study also considers the dose resulting from indoor occupancy; the total site occupancy is increased but allowance is made for shielding from external irradiation by the building. The present study also takes into account different activities with varying dust loading and inhalation rates, thus producing a more realistic set of doses. However, it is evident that both studies indicate that there should be little radiological concern over the current practice of landfill disposal of waste material from the steel production industry.

C7 References

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APPENDIX D

DOSES FROM THE USE OF SLAG

D1 Introduction

The recycling of slag from the steel production industry is a common practice in the UK. Slag from integrated steel production plants run by Corus (formerly British Steel) is processed for use as a structural material by specialist companies (Gray, 2000). It has therefore been assumed in this study that all slag is recycled. The main areas of use are in road construction and maintenance, and housing construction, i.e. manufacture of cement, concrete and insulating materials. Slag produced during steel manufacture may be cooled (quenched) in air or sprayed with water, to solidify it, and then granulated. The areas of use of air cooled and granulated slag overlap so it has been assumed that both types of slag are used in each scenario considered.

The quantity of slag used in different applications varies according to demand and availability. The scenarios considered in this Appendix cover the major uses of slag and as a result some minor uses (e.g. glass making, agricultural applications and rail ballast) have been ignored.

The use of slag in road and building materials will lead to the exposure of a number of groups of people. The most significant groups, and those considered in this study, are workers manufacturing road and building products containing slag, workers constructing roads using these products and members of the public living in houses and using car parks and playgrounds constructed using materials containing slag. The methodologies, assumptions, and data used to determine doses to these groups are described below. The resulting doses are also presented and discussed.

Several sets of doses have been determined. The first set comprises doses arising from the radionuclides present in the slag. It is important to note, however, that the other components of construction materials will also contain naturally occurring radionuclides, which will result in exposure of workers and members of the public. Similarly, it should also be remembered that the slag replaces other constituents that would themselves contain naturally occurring radionuclides and thus give rise to radiation exposures. In order to build up a picture of the overall radiological significance of the use of slag in construction materials it is therefore necessary to consider two further sets of doses, as appropriate. One set comprises doses from all the radionuclides present in the construction materials that contain slag (ie not simply those originating in the slag). The other consists of doses from the manufacture and use of construction materials that do not contain slag.

D1.1 Quantities of slag used in road building materials

The composition and form of all road surfaces are covered by British Standards. The British Standard (BS) covering the composition of dense tar road surfacing is BS5273 (BSI, 1990) and the BS covering asphalts, rolled for roads and paved areas is BS 594 (BSI, 1992a). There are a large number of different types of road surfaces but these are the most widely used. The main constituent materials of road surfaces are coarse aggregate, binder (a mixture of fine aggregate and tar), filler (also fine aggregate) and chippings for application to the surface of the wearing course (made from coarse aggregate). The proportions of these materials are varied depending on the type of road surface required, however, all of the materials, with the exception of tar, could be slag from the steel production process. The British Standards mentioned above (BSI, 1990; 1992a) indicate that tar comprises approximately 10% of the surface material. It has therefore been conservatively assumed for this study that 90% of the road surface material is slag and the remaining 10% is tar, used to bind the surface.

D1.2 Quantities of slag used in house building materials

Castle Portland blast furnace cements are manufactured using ordinary Portland cement and selected granulated blast furnace slag which complies with the requirements of BS 6699 (BSI, 1992b). The proportions are varied to achieve products complying with BS 146 (BSI, 1996a) or BS 4246 (BSI, 1996b). The quantity of slag used in the cement, and therefore concrete, is usually up to 35% (Castlecement, 2000), but can be as much as 55% (Castlecement, 2000). Cement containing more than 55% blast furnace slag can only be used in special applications (Castlecement, 2000). For this study it has been assumed that the building materials considered contain 35% slag.

D2 Doses to workers

Two types of worker are considered, those manufacturing materials used in housing and road construction and those using the materials to build roads. Doses to workers using materials containing slag to build houses were not considered because the fraction of slag used in housing materials is much less than the fraction used in road materials, as discussed in the previous sections. It was therefore considered that doses to workers resulting from the use of slag in construction materials would be most restrictive for workers constructing roads. The exposure pathways considered for all workers are external irradiation, skin contamination and inhalation and ingestion of dust. The total dose to the worker is the sum from each of these pathways. The methods and data used to determine doses from each pathway are presented below. Clearly the doses received by individual workers will vary substantially depending in detail on their work activities, with the majority receiving trivial doses. The aim of this part of

the study was to determine doses typical of those received by the most exposed workers.

D2.1 Doses to workers manufacturing building and road materials

The following exposure scenarios apply to workers manufacturing both road building materials and house building materials.

D2.1.1 Methodology

External irradiation

External irradiation is assumed to occur from a 100 t slag pile that would be stored and used in the production of construction materials. The geometry assumed for the source is a half cylinder. The worker is assumed to be 1.5 m from the source. It is assumed that the worker is exposed to undiluted slag (ie that has just been delivered), this scenario then applies equally to manufacturers of road building materials and house building materials.

$$D_{ext} = \Sigma_R A_R \times DC_{ext,R} \times T$$

where D_{ext} = Dose from external exposure to slag, 1.5 m from the pile (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1}), see Table D1

$DC_{ext,R}$ = External dose rate at 1.5 m from slag pile, from radionuclide R (Sv h^{-1} per Bq g^{-1})

T = Duration of exposure, 100 h y^{-1}

Values of $DC_{ext,R}$ were determined using energy specific dose equivalent rates (IAEA, 1992).

TABLE D1 Activity concentrations in concrete and blast furnace slag (Bq kg^{-1})

	Typical UK			Typical EU		
Material	^{238}U	^{232}Th	^{40}K	^{238}U	^{232}Th	^{40}K
Concrete	70	30	500	40	30	400
Blast furnace slag*,†	88‡	49‡	§	270	70	240

* The activity concentration of ^{235}U (4 Bq kg^{-1}) has been inferred from the measured activity concentration of ^{238}U and the natural isotopic content of uranium (4.5% by activity).

† All daughters of the decay series have been assumed to be present in secular equilibrium.

‡ Measured values (Giles and Harvey, 2000). All other values from EC (1997) and EC (1999).

§ No data available.

Inhalation of dust

An individual is assumed to inhale dust during the manufacturing process. Part of the dust will be from the slag used in production. The exposure duration is one

working year, as a constant level of dust is assumed to be in the air, either from the production process itself or from resuspension of settled material. It was assumed that no respiratory protection would be used.

$$D_{inh} = \Sigma_R A_R \times Dil \times DC_{inh,w,R} \times C_d \times I_{r,w} \times T$$

where D_{inh} = Dose from the inhalation of radionuclides in slag dust (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag dust (Bq g^{-1}) it is assumed that this will be the same as the activity concentration in slag, see Table D1

Dil = Dilution factor for slag with other materials, 0.35 for housing materials and 1 for road materials (as tar will not contribute to dust level)

$DC_{inh,w,R}$ = Dose coefficient for inhalation in the workplace for radionuclide R (Sv Bq^{-1}) (ICRP, 1994a).

C_d = Concentration of dust in the air, 10^{-3} g m^{-3} (Penfold et al, 1997)

$I_{r,w}$ = Inhalation rate for workers, 1.18 m^3 h^{-1} (Robinson, 1996)

T = Exposure duration, 2000 h y^{-1}

Inadvertent ingestion

It is assumed that a worker inadvertently ingests dust at a rate of 5 mg h^{-1} for 2000 hours per year (Robinson, 1996). The dose from inadvertent ingestion has been estimated using:

$$D_{ing} = \Sigma_R A_R \times Dil \times DC_{ing,w,R} \times Ing_{r,w} \times T$$

where D_{ing} = Dose from the inadvertent ingestion of radionuclides in slag dust (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag dust (Bq g^{-1}) it is assumed that this will be the same as the activity concentration in slag, see Table D1

Dil = Dilution factor for slag with other materials, 0.35 for housing materials and 1 for road materials (as tar will not contribute to dust level).

$DC_{ing,w,R}$ = Dose coefficient for ingestion for radionuclide R in the workplace (Sv Bq^{-1}) (ICRP, 1994a)

$Ing_{r,w}$ = Inadvertent ingestion rate for workers, 5 10^{-3} g h^{-1} (Robinson, 1996)

T = Exposure duration, 2000 h y^{-1}

Skin contamination

Estimates of the dose due to skin contamination from radionuclides in slag dust were determined using the methodology in NRPB (1997). Workers are assumed to be exposed to contaminated dust on their skin and clothing for an entire working year, 2000 h. The thickness of the deposit on the skin is assumed to be 0.1 mm (Penfold et al, 1997). Beta irradiation of the skin and gamma irradiation of the skin are treated separately because of the greater penetration of gamma rays. For beta irradiation it is assumed that the hands, face and some of the arms and neck are covered with dust (2000 cm^2 in total of UVR exposed skin). For gamma irradiation it is assumed that all of the UVR exposed skin will be irradiated as a result of dust on skin and clothing.

The activity concentration of radionuclide R per unit area on the skin (Bq cm^{-2}) is given by

$$A_{skin,R} = \rho \times d \times A_R \times Dil$$

where A_R = Activity concentration of radionuclide R in slag dust (Bq g^{-1})

Dil = Dilution factor for slag with other materials, 0.35 for housing materials and 1 for road materials

ρ = Density of dust (g cm^{-3})(see below)

d = Thickness of dust on the skin, 0.01 cm

British Standards (BSI, 1990; 1992a) describe the density of fine aggregates to be between 1.44 and 1.12 g cm^{-3}; and therefore 1.3 g cm^{-3} has been chosen to represent slag dust.

The equivalent dose, from radionuclide R, to the skin area that is exposed to beta and gamma radiation, $H_{skin,\beta,R}$ and $H_{skin,\gamma,R}$, respectively, can then be calculated as follows

$$H_{skin,\beta,R} = A_{skin,R} \times T \times \beta_{skin,R}$$

$$H_{skin,\gamma,R} = A_{skin,R} \times T \times \gamma_{skin,R}$$

where T = Exposure duration, 2000 h y^{-1}.

$\beta_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for β irradiation for radionuclide R (Sv h^{-1} per Bq cm^{-2}) (Harvey et al, 1993)

$\gamma_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for γ irradiation for radionuclide R (Sv h^{-1} per Bq cm^{-2}) (Harvey et al, 1993)

Effective doses to skin from beta and gamma irradiation, $D_{skin,\beta,R}$ and $D_{skin,\gamma,R}$, are calculated separately, summed together and then summed over radionuclides to give the total effective dose from skin contamination, D_{skin} (Sv y^{-1}).

$$D_{skin,\beta,R} = H_{skin,\beta,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994a)

EXP_{area} = Area of skin exposed to β irradiation, 2000 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2

$$D_{skin,\gamma,R} = H_{skin,\gamma,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994a)

EXP_{area} = Area of skin exposed to γ irradiation, 3000cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000cm^2

D_{skin} = Σ_R ($D_{skin,\beta,R}$ + $D_{skin,\gamma,R}$)

Total dose to manufacturing worker

The total dose to a worker manufacturing building or road construction materials is the sum from all the exposure pathways.

$D_{man.work}$ = D_{ext} + D_{inh} + D_{ing} + D_{skin}

D2.1.2 Results

The estimated doses to workers manufacturing products for road and building construction are presented in Tables D2 and D3. Table D2 presents the doses from radionuclides originating in the slag. The total dose to workers manufacturing materials used in house building is 6 $\mu Sv\ y^{-1}$. All three radionuclide decay chains contribute a similar amount to the total dose. The most significant exposure pathway is inhalation of dust (87% of the total dose). The total dose from radionuclides in the slag to workers manufacturing road building materials is 18 $\mu Sv\ y^{-1}$. All three radionuclide decay chains contribute a similar amount to the total dose and the most significant exposure pathway is the inhalation of dust (88% of the total dose). The difference in the doses from the two scenarios arises from the quantity of slag used in the different types of materials; it was assumed that road building materials could be made almost entirely from slag.

TABLE D2 Doses to workers manufacturing building and road materials containing slag from radionuclides in the slag

	Dose (Sv y^{-1})				
Radionuclide	External	Inhalation	Ingestion	Skin contamination	Total
House building					
^{238}U *	$2.0\ 10^{-8}$	$1.1\ 10^{-6}$	$4.3\ 10^{-7}$	$9.6\ 10^{-8}$	$1.6\ 10^{-6}$
^{232}Th *	$1.6\ 10^{-8}$	$2.0\ 10^{-6}$	$1.5\ 10^{-7}$	$5.4\ 10^{-8}$	$2.2\ 10^{-6}$
^{235}U *	$2.5\ 10^{-10}$	$2.4\ 10^{-6}$	$2.8\ 10^{-8}$	$1.6\ 10^{-9}$	$2.5\ 10^{-6}$
Total	$3.7\ 10^{-8}$	$5.5\ 10^{-6}$	$6.0\ 10^{-7}$	$1.5\ 10^{-7}$	$6.3\ 10^{-6}$
Road building					
^{238}U *	$2.0\ 10^{-8}$	$3.1\ 10^{-6}$	$1.2\ 10^{-6}$	$2.7\ 10^{-7}$	$4.6\ 10^{-6}$
^{232}Th *	$1.6\ 10^{-8}$	$5.6\ 10^{-6}$	$4.3\ 10^{-7}$	$1.6\ 10^{-7}$	$6.0\ 10^{-6}$
^{235}U *	$2.5\ 10^{-10}$	$6.9\ 10^{-6}$	$7.9\ 10^{-8}$	$4.6\ 10^{-9}$	$7.0\ 10^{-6}$
Total	$3.7\ 10^{-8}$	$1.6\ 10^{-5}$	$1.7\ 10^{-6}$	$4.3\ 10^{-7}$	$1.8\ 10^{-5}$

* Includes all members of the decay chain.

TABLE D3 Doses to workers manufacturing building materials

	Dose (Sv y^{-1})				
Radionuclide	External	Inhalation	Ingestion	Skin contamination	Total
Dose from slag	$3.\ 7\ 10^{-8}$	$5.5\ 10^{-6}$	$6.0\ 10^{-7}$	$1.5\ 10^{-7}$	$6.3\ 10^{-6}$
Total dose from material	$3.7\ 10^{-8}$	$1.3\ 10^{-5}$	$1.4\ 10^{-6}$	$3.6\ 10^{-7}$	$1.5\ 10^{-5}$
Dose from material without slag	$2.6\ 10^{-8}$	$1.1\ 10^{-5}$	$1.3\ 10^{-6}$	$3.2\ 10^{-7}$	$1.3\ 10^{-5}$
Excess dose due to slag	$1.0\ 10^{-8}$	$1.5\ 10^{-6}$	$1.5\ 10^{-7}$	$4.1\ 10^{-8}$	$1.7\ 10^{-6}$

Table D3 presents estimated doses received by workers manufacturing materials used in house building, from all radionuclides in the material (ie not just those originating in the slag), and from the manufacture of similar materials that do not contain slag. In each case the other components of the building materials were assumed to have activity concentrations identical to those in ordinary cement and concrete from Table D1. The total dose to a worker manufacturing building materials containing 35% slag is estimated to be 15 $\mu Sv\ y^{-1}$. The dose from ordinary building materials is estimated to be 13 $\mu Sv\ y^{-1}$, therefore the excess dose from using slag is 2 $\mu Sv\ y^{-1}$.

There is very little data available on activity concentrations in road building materials; it is therefore difficult to establish whether there is any excess dose arising from the use of slag. However, it is clear that ordinary road building materials will have some naturally occurring radionuclide content and therefore if there is any excess dose from using slag it will be less than 18 $\mu Sv\ y^{-1}$.

D2.2 Doses to workers constructing roads

D2.2.1 Methodology

External irradiation

Workers are assumed to be externally exposed from the road for the full working year.

$D_{ext} = \Sigma_R \; A_R \times DC_{ext,R} \times Dil \times T$

where D_{ext} = Dose from external exposure to slag (Sv y^{-1})(1m from source)

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1}), see Table D1

$DC_{ext,R}$ = Dose coefficient for external exposure, for radionuclide R (Sv h^{-1} per Bq g^{-1})

Dil = Dilution of slag in road building materials, 0.9, see Section D1.1

T = Duration of exposure, 2000 h y^{-1} is assumed.

Values of $DC_{ext,R}$ were determined using energy specific dose equivalent rates from IAEA (1992), values appropriate for a plane surface were used.

Inhalation of dust

Workers are assumed to inhale dust during the road construction process. Part of the dust will be from the slag. A constant level of dust is assumed for the working year, created either by the construction process or from resuspension of settled material. Assumptions for the calculation below were taken from Penfold et al (1997). It was assumed that no respiratory protection would be used.

$D_{inh} = \Sigma_R \; A_R \times Dil \times DC_{inh,w,R} \times C_d \times I_{r,w} \times T$

where D_{inh} =Dose from the inhalation of radionuclides in slag dust (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag dust (Bq g^{-1}), it is assumed that this will be the same as the activity concentration in slag, see Table D1

Dil = Dilution factor for slag with other materials, 1, see Section D1.1, it is assumed that tar will not contribute to dust levels

$DC_{inh,w,R}$ = Dose coefficient for inhalation for workers, for radionuclide R (Sv Bq^{-1})(ICRP, 1994a)

C_d = Concentration of dust in the air, 10^{-3} g m^{-3} (Penfold et al, 1997).

$I_{r,w}$ = Inhalation rate for workers, 1.18 m^3 h^{-1} (Robinson, 1996)

T = Time spent by workers inhaling dust, 2000 h y^{-1}

Inadvertent ingestion of dust

It is assumed that a worker ingests dust at a rate of 5mg h^{-1} for 2000 hours per year (Robinson, 1996).

$$D_{ing} = \Sigma_R A_R \times Dil \times DC_{ing,w,R} \times Ing_{r,w} \times T$$

Where D_{ing} = Dose from the ingestion of radionuclides in slag dust (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag dust (Bq g^{-1}) it is assumed that this will be the same as the activity concentration in slag, see Table D1

Dil = Dilution factor for slag with other materials, 1, see Section D1.1, it is assumed that tar will not contribute to dust levels

$DC_{ing,w,R}$ = Dose coefficient for ingestion for workers, for radionuclide R (Sv Bq^{-1})(ICRP, 1994a)

$Ing_{r,w}$ = Inadvertent ingestion rate for workers, 5.0 10^{-3} g h^{-1} (Robinson, 1996)

T = Exposure duration, 2000 h y^{-1}

Skin contamination

As in Section D2.1, workers are assumed to be exposed to contaminated dust on their face and hands for an entire working year, 2000 h. The thickness of the deposit on the skin is assumed to be 0.1 mm (Penfold et al, 1997). Beta irradiation of the skin and gamma irradiation of the skin are treated separately because of the greater penetration of gamma rays. For beta irradiation it is assumed that the hands, face and most of the arms and neck are covered with dust (2000 cm^2 in total of UVR exposed skin). For gamma irradiation it is assumed that all of the UVR exposed skin will be irradiated as a result of dust on skin and clothing.

The activity concentration of radionuclide R per unit area on the skin (Bq cm^{-2}) is given by

$$A_{skin,R} = \rho \times d \times A_R \times Dil$$

where A_R = Activity concentration of radionuclide R in slag dust (Bq g^{-1})

Dil = Dilution factor for slag with other materials, 1, it is assumed that tar will not contribute to dust levels

ρ = Density of dust (g cm^{-3})(see below)

d = Thickness of dust on the skin, 0.01 cm

British Standards (BSI, 1990; 1992a) describe the density of the fine aggregates to be between 1.44 and 1.12 g cm^{-3}; 1.3 g cm^{-3} has been chosen to represent slag dust.

The equivalent dose from radionuclide R to the skin area that is exposed to beta and gamma radiation, $H_{skin,\beta,R}$ and $H_{skin,\gamma,R}$ respectively, can then be calculated as follows

$H_{skin,\beta,R} = A_{skin,R} \times T \times \beta_{skin,R}$

$H_{skin,\gamma,R} = A_{skin,R} \times T \times \gamma_{skin,R}$

where T = Exposure duration, 2000 h y^{-1}

$\beta_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for β irradiation for radionuclide R (Sv h^{-1} per Bq cm^{-2})(Harvey et al, 1993)

γ_{skin} = The skin equivalent dose rate to the basal layer of the skin epidermis for γ irradiation for radionuclide R (Sv h^{-1} per Bq cm^{-2})(Harvey et al, 1993).

Effective doses to skin from beta and gamma irradiation, $D_{skin,\beta,R}$ and $D_{skin,\gamma,R}$, are calculated separately, summed together and then summed over radionuclides to give the total effective dose from skin contamination, D_{skin} (Sv y^{-1}).

$$D_{skin,\beta,R} = H_{skin,\beta,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994a)

EXP_{area} = Area of skin exposed to β irradiation, 2000cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000cm^2

$$D_{skin,\gamma,R} = H_{skin,\gamma,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994a)

EXP_{area} = Area of skin exposed to γ irradiation, 3000cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000cm^2

$D_{skin} = \Sigma_R (D_{skin,\beta,R} + D_{skin,\gamma,R})$

Total dose to road construction worker

The total dose to a worker constructing roads using materials containing slag is the sum from all the exposure pathways.

$D_{con.work} = D_{ext} + D_{inh} + D_{ing} + D_{skin}$

D2.2.2 Results

The doses to workers constructing roads, from radionuclides present in the slag are presented in Table D4. The estimated total dose to a worker is 19 μSv y^{-1}. The most significant exposure pathway is inhalation of dust; this accounts for 84% of the total dose. Each of the radionuclide decay chains contribute similar amounts to the total dose.

TABLE D4 Doses to workers constructing roads using materials containing slag

	Dose (Sv y^{-1})				
Radionuclide	External	Inhalation	Ingestion	Skin contamination	Total
Road building					
^{238}U *	$7.1\ 10^{-7}$	$3.1\ 10^{-6}$	$1.2\ 10^{-6}$	$2.7\ 10^{-7}$	$5.3\ 10^{-6}$
^{232}Th *	$5.5\ 10^{-7}$	$5.6\ 10^{-6}$	$4.3\ 10^{-7}$	$1.6\ 10^{-7}$	$6.7\ 10^{-6}$
^{235}U *	$9.2\ 10^{-9}$	$6.9\ 10^{-6}$	$7.9\ 10^{-8}$	$4.6\ 10^{-9}$	$7.0\ 10^{-6}$
Total	$1.3\ 10^{-6}$	$1.6\ 10^{-5}$	$1.7\ 10^{-6}$	$4.3\ 10^{-7}$	$1.9\ 10^{-5}$

* Includes all members of the decay chain.

There is very little data available on activity concentrations in road building materials; it is therefore difficult to establish whether there is any excess dose arising from the use of slag. However, it is clear that ordinary road building materials will have some naturally occurring radionuclide content and, therefore, if there is any excess dose from using slag it will be less than 19 μSv y^{-1}.

D3 Doses to members of the public

The use of slag in roads and building materials will lead to the exposure of a number of groups of members of the public. Those considered in this study are members of the public exposed in car parks, playgrounds and houses constructed from materials containing slag. The methodologies, assumptions and data used to determine doses to each of the groups are described below.

D3.1 Doses to members of the public in a car park surfaced using tarmac containing slag

The only exposure pathway likely to be significant in this scenario is external exposure from the tarmac; it is thought that this scenario is most applicable to adults.

D3.1.1 Methodology

External irradiation

$$D_{ext} = \Sigma_R\ A_R \times F_{slag} \times DC_{ext,R} \times T$$

Where D_{ext} = Dose from external irradiation in a car park (Sv y^{-1})(1m above ground).

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1})(see Table D1)

F_{slag} = Fraction of slag used in tarmac surface materials, 0.9 (see Section D1.1)

$DC_{ext,R}$ = Dose coefficient for external exposure for radionuclide R (Sv h^{-1} per Bq g^{-1})

T = Duration of exposure, 100 h y^{-1} is assumed, this is based on approximately 15 mins per day

Values for $DC_{ext,R}$ were determined using energy specific dose equivalent rates from IAEA (1992), rates appropriate for a plane surface were used.

D3.1.2 Results

Doses to members of the public using a car park surfaced using tarmac containing slag are given in Table D5. The dose to an adult from external exposure from the ground is 6 10^{-2} μSv y^{-1}. The uranium-238 and thorium-232 decay chains both contribute to this dose, with the contribution from the uranium-235 decay chain being 2 orders of magnitude lower.

The excess dose, if any, from the use of slag as a replacement for other road building materials will be less than 6 10^{-2} μSv y^{-1} because of the naturally occurring radionuclide content of other road surfacing materials, as discussed in Section D2.2.

TABLE D5 Doses to adult members of the public from car parks surfaced using tarmac containing slag

	Dose (Sv y^{-1})	
Radionuclide	External	Total
^{238}U *	3.5 10^{-8}	3.5 10^{-8}
^{232}Th *	2.7 10^{-8}	2.7 10^{-8}
^{235}U *	4.6 10^{-10}	4.6 10^{-10}
Total	6.3 10^{-8}	6.3 10^{-8}

* Includes all members of the decay chain.

D3.2 Doses to members of the public from playing on a tarmac area containing slag

For this scenario the most significant pathway will also be external exposure, it is thought that this pathway is most applicable to children (10 year olds).

D3.2.1 Methodology

External irradiation

$D_{ext} = \Sigma_R A_R \times F_{slag} \times DC_{ext,R} \times T$

where D_{ext} = Dose from external irradiation from tarmac (Sv y^{-1})(1 m above ground).

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1})(see Table D1).

F_{slag} = Fraction of slag used in tarmac surface materials, 0.9 (see Section D1.1)

$DC_{ext,R}$ = Dose coefficient for external exposure for radionuclide R (Sv h^{-1} per Bq g^{-1})

T = Duration of exposure, 264 h y^{-1} (EC, 2000).

Values for $DC_{ext,R}$ were determined using energy specific dose equivalent rates from IAEA (1992), rates appropriate for a plane surface were used.

D3.2.2 Results

The doses to 10 year old children from playing on a tarmac area containing slag are presented in Table D6. The dose from external irradiation from the tarmac is 0.2 μSv y^{-1}. As discussed in Section D2.2, the excess dose, if any, from using slag as a replacement for other surfacing materials will be less than 0.2 μSv y^{-1}.

TABLE D6 Doses to 10 year old children from a play area surfaced using tarmac containing slag

	Dose (Sv y^{-1})	
Radionuclide	External	Total
^{238}U *	9.3 10^{-8}	9.3 10^{-8}
^{232}Th *	7.2 10^{-8}	7.2 10^{-8}
^{235}U *	1.2 10^{-9}	1.2 10^{-9}
Total	1.7 10^{-7}	1.7 10^{-7}

* Includes all members of the decay chain.

D3.3 Doses to members of the public from playing on an area of waste ground surfaced using slag

A playing area on waste ground surfaced with slag is an unlikely scenario but was chosen to represent a hypothetical dusty area to explore potential doses from other exposure pathways. This scenario would be most applicable to children (10 year olds). Children were assumed to play football on the same piece of ground for 4 hours per week, 50 weeks of the year (200h y^{-1}). It has been conservatively assumed that the ground surface is made up of 100% slag.

The exposure pathways considered were external irradiation from the ground, inhalation of dust containing radionuclides and inadvertent ingestion of dust.

D3.3.1 Methodology

External irradiation

$D_{ext} = \Sigma_R A_R \times DC_{ext,R} \times T$

where D_{ext} = Dose from external irradiation from slag (Sv y^{-1})(1m above ground).

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1})(see Table D1).

$DC_{ext,R}$ = Dose coefficient for external exposure for radionuclide R (Sv h^{-1} per Bq g^{-1})

T = Duration of exposure, 200 h y^{-1}.

Values for $DC_{ext,R}$ were determined using energy specific dose equivalent rates from IAEA (1992), rates appropriate for a plane surface were used.

Inhalation of dust

$D_{inhal} = \Sigma_R A_R \times DC_{inhal,R} \times C_{aH} \times V_H \times T_H$

where D_{inhal} = Dose from inhalation of radionuclides in re-suspended slag dust (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1})(see Table D1).

$DC_{inhal,R}$ = Dose coefficient for inhalation for radionuclide R (Sv Bq^{-1})

C_{aH} = Inhalable dust concentration in the air, 2 10^{-3} g m^{-3} (EC, 2000)

V_H = Breathing rate for 10 year old children for high physical activity, 0.87 m^3 h^{-1} (Robinson, 1996)

T_H = annual time of exposure, 200 h y^{-1}.

It has been assumed that the activity concentrations in slag dust are the same as those in the slag.

Inadvertent ingestion of dust

$D_{ing} = \Sigma_R A_R \times DC_{ing,R} \times Ing \times T_H$

where D_{ing} = Dose from ingestion of radionuclides in slag dust (Sv y^{-1})

A_R = Activity concentration of radionuclide R in slag (Bq g^{-1})(see Table D1).

$DC_{ing,R}$ = Dose coefficient for ingestion, for radionuclide R (Sv Bq^{-1})

Ing = Inadvertent ingestion rate for a 10 year old, 10 mg h^{-1} (Robinson, 1996).

T_H = Exposure duration, 200 h y^{-1}

D3.3.2 Results

The estimated doses to 10 year old children from playing on an area of waste ground surfaced using 100% slag are presented in Table D7. The total dose is 4.5 μSv y^{-1}, the pathway that contributes most significantly to this dose is inhalation of dust, this is responsible for 82% of the total dose. Scoping calculations were undertaken to estimate the dose from inhalation of radon emanating from the slag. These indicated that this pathway does not contribute significantly to the total dose.

TABLE D7 Doses to 10 year old children from a waste ground play area made with recycled slag

	Dose (Sv y^{-1})			
Radionuclide	External	Inhalation	Ingestion	Total
^{238}U	$7.9\ 10^{-8}$ *	$1.2\ 10^{-7}$	$1.3\ 10^{-8}$	$2.2\ 10^{-7}$
^{234}U		$1.5\ 10^{-7}$	$1.3\ 10^{-8}$	$1.6\ 10^{-7}$
^{230}Th		$4.9\ 10^{-7}$	$4.2\ 10^{-8}$	$5.3\ 10^{-7}$
^{226}Ra		$1.5\ 10^{-7}$	$1.4\ 10^{-7}$	$3.0\ 10^{-7}$
^{232}Th	$6.1\ 10^{-8}$ *	$4.4\ 10^{-7}$	$2.8\ 10^{-8}$	$5.3\ 10^{-7}$
^{228}Ra		$7.9\ 10^{-8}$	$3.8\ 10^{-7}$	$4.6\ 10^{-7}$
^{228}Th		$1.0\ 10^{-6}$	$4.2\ 10^{-8}$	$1.1\ 10^{-6}$
^{235}U	$1.0\ 10^{-9}$ *	$6.0\ 10^{-9}$	$5.7\ 10^{-10}$	$7.6\ 10^{-9}$
^{231}Pa		$2.1\ 10^{-7}$	$7.4\ 10^{-9}$	$2.2\ 10^{-7}$
^{227}Ac		$1.0\ 10^{-6}$	$1.6\ 10^{-8}$	$1.1\ 10^{-6}$
Total	$1.4\ 10^{-7}$	$3.7\ 10^{-6}$	$6.9\ 10^{-7}$	$4.5\ 10^{-6}$

* Includes all members of the decay chain.

D3.4 Doses to members of the public living in houses built using materials containing slag

The exposure of members of the public has been assumed to be via two main pathways; external exposure from gamma emitting radionuclides and inhalation of radon emanating from the building materials into indoor air. The methods and data used to determine doses from these pathways are described below.

D3.4.1 Doses from inhalation of radon

Doses from the inhalation of radon were estimated using the methodology described in Cliff et al (1984) as defined in the following:

Dose from inhalation of radon-220

$D_{Rn220} = A_{Pb212} \times DC_{Pb212} \times T \times f_I \times IR$

where A_{Pb212} = Concentration of ^{212}Pb (Bq m^{-3})

DC_{Pb212} = Dose coefficient for inhalation for an adult for lead-212 (Sv Bq^{-1})

T = Time spent at location, 8760 h y^{-1}

F_I = Fraction of time spent indoors, 0.9 (Robinson, 1996)

IR = Adult inhalation rate, 0.83 m^3 h^{-1} (Robinson, 1996)

The concentration of lead-212 was used in the above equation because lead-212 is the daughter of radon-220 that dominates exposures from radon-220 and its daughters (Cliff et al, 1984; Fayers and Bexon, 1998). The concentration of lead-212 in the room was obtained using:

$$A_{Pb212} = E_{R,Rn} \times A \times \frac{1}{V \times (\lambda_{Rn220} + VR)} \times \frac{(\lambda_{Pb212})}{(\lambda_{Rn220})}$$

where $E_{R,Rn}$ = Exhalation rate of radon-220 from walls of room (Bq m^{-2} s^{-1}) ,see below

A = Surface area of walls in room (m^2)

V = Volume of room (m^3)

VR = Ventilation rate of air in room, 2.8 10^{-4} s^{-1} (equivalent to one air change per hour) (Cliff et al, 1984)

λ_{Rn220} = Decay constant for radon-220, 1.25 10^{-2} s^{-1}

λ_{Pb212} = Decay constant for lead-212, 1.81 10^{-5} s^{-1}

To determine A and V it was assumed that the room had dimensions 5 m × 4 m × 3 m. The exhalation rate of radon-220 is calculated using the approach described in UNSCEAR (1993), EC (1997) and Cliff et al (1984):

$E_{R,Rn} = F_{slag} \times A_{Ra228} \times K_{Rn} \times \lambda_{Rn220} \times \rho \times L_D \times \tanh (L_A/L_D)$

where F_{slag} = Fraction of slag used in building materials, 35% (Castlecement, 2000)

A_{Ra228} = Activity concentration of the radon precursor in the slag, i.e. radium-228 for radon-220, (Bq g^{-1}) see Table D1 for values

K_{Rn} = Emanation fraction for radon from slag, 0.7% (Bruzzi et al, 1992), or 5% (Siotis and Wrixon, 1984), see below

ρ = Density of the wall, 2.5 10^6 g m^{-3} (Siotis and Wrixon, 1984)

L_D = Diffusion length of radon in concrete (m)

L_A = Half thickness of wall, 0.15m (Siotis and Wrixon, 1984)

Note: $\tanh x = \frac{e^x - e^{-x}}{e^x + e^{-x}}$

and $L_D = \sqrt{\frac{D_{Rn}}{\varepsilon \times \lambda_{Rn220}}}$

where D_{Rn} = The radon diffusion coefficient for building materials, 1 10^{-8} m^2 s^{-1} (UNSCEAR, 1993)

ε = The porosity of building materials, 0.15 (UNSCEAR, 1993)

The emanation fraction for radon from blast furnace slag has been measured to be 0.7% (Bruzzi et al, 1992). This is considerably less than emanation fractions assumed for concrete of 5% (Siotis and Wrixon, 1984; Smith et al, 2001). Using an emanation fraction of 0.7% for the fraction of slag used in concrete may not be appropriate, however, as the slag is part of a concrete mixture. Because of the lack of data available for blast furnace slag aggregate mixtures, two sets of doses have been assessed. The first set assumed an emanation fraction of 0.7% for radon in the slag and 5% for radon emanating from the concrete, the second set assumed an emanation fraction of 5% for both components of the building materials.

Dose from inhalation of radon-222

The dose from inhalation of radon-222 was estimated using the same method as that used above modified to allow use of a dose coefficient expressed in terms of Sv y^{-1} per Bq m^{-3}, as follows:

$D_{Rn222} = A_{Rn222} \times DC_{Rn222} \times f_I$

where A_{Rn222} = The activity concentration in room air of radon-222 (Bq m^{-3})

DC_{Rn222} = Dose coefficient for inhalation for an adult of radon-222, 5.5 10^{-5} Sv y^{-1} per Bq m^{-3} (derived from ICRP 65) (ICRP 1994b).

f_I = Fraction of time spent indoors, 0.9 (Robinson, 1996)

The activity concentration of radon-222 in the room is calculated using:

$$A_{Rn222} = E_{R,Rn} \text{ x } A \frac{1}{V \text{ x } (\lambda_{Rn220} + VR)}$$

where $E_{R,Rn}$ = Exhalation rate of radon-222 from walls of room (Bq m^{-2} s^{-1}) see below

A = Surface area of walls in room (m^2)

V = Volume of room (m^3)

VR = Ventilation rate of air in room, 2.8 10^{-4} s^{-1} (equivalent to one air change per hour) (Cliff et al, 1984)

λ_{Rn222} = Decay constant for radon-222, 2.099 10^{-6} s^{-1}

$E_{R,Rn} = F_{slag} \times A_{Ra226} \times K_{Rn} \times \lambda_{Rn222} \times \rho \times L_D \times \tanh (L_A/L_D)$

where F_{slag} = Fraction of slag used in building materials, 35% (Castlecement, 2000)

A_{Ra228} = Activity concentration of the radon precursor in the slag, i.e. radium-226 for radon-222, (Bq g^{-1}) see Table D1 for values

K_{Rn} = Emanating fraction for radon from slag, 0.7% (Bruzzi et al, 1992), or 5% (Siotis and Wrixon, 1984), see above

ρ = Density of the wall, 2.5 10^6 g m^{-3} (Siotis and Wrixon, 1984)

L_D = Diffusion length of radon in concrete (m)

L_A = Half thickness of wall, 0.15m (Siotis and Wrixon, 1984)

Note: $\tanh x = \frac{e^x - e^{-x}}{e^x + e^{-x}}$

and $L_D = \sqrt{\frac{D_{Rn}}{\varepsilon \times \lambda_{Rn}}}$

where D_{Rn} = The radon diffusion coefficient for building materials, 1 10^{-8} m^2 s^{-1} (UNSCEAR, 1993)

ε = The porosity of building materials, 0.15 (UNSCEAR, 1993).

Radon exhalation rates

In Table D8 the radon exhalation rates assumed in this study are compared with some measured exhalation rates. Factors affecting the amount of radon that exhales from a material are: the internal structure of the material, its radium concentration and moisture content, the surface treatment of the material and the outer pressure. Typical results of measured exhalation rates from EU studies are presented in Table D8. Normalised exhalation rates, i.e. exhalation rate per unit radium concentration within the material are also presented. Reported results vary considerably depending on the materials measured and the methods used. It can be concluded that radon exhalation strongly depends on the type of material. It has also been noted that there are great technical difficulties in obtaining reliable results (EC, 1997). There is no standard method for measurement of exhalation rates. It is worth noting that the normalised exhalation rates measured in Finland for slag aggregate concrete are lower than those for ordinary concrete, which implies that the emanation factor for slag is lower than that of concrete. This is only one isolated measurement, but it is

consistent with the findings of Bruzzi et al (1992), that the exhalation rates of radon from slag are low.

TABLE D8 Typical exhalation rates measured in EU countries

Member state	Exhalation rate Bq m^{-2} h^{-1}	Normalised exhalation rate Bq m^{-2} h^{-1} per Bq kg^{-1}
Finland		
Concrete 10 – 20 cm thick*	15.9 – 31.5	0.27 – 0.52
Slag aggregate concrete 15cm thick*	10.0 – 12.0	0.13 – 0.16
Germany		
Concrete†	1.1	0.022
Aerated concrete†	1.0	0.05
Netherlands		
Concrete‡	2.2 – 6.2	0.009 – 0.234
Aerated concrete‡	0.85 – 1.2	0.039 – 0.066
Estimated for this study		
Concrete	8.1	0.12
Concrete with 35% slag§	5.8	0.08
Concrete with 35% slag¶	8.8	0.12

* See Mustonen (1984)
† See Keller (1991)
‡ See Van Dijk and de Jong (1991)
§ Using emanation factor of 0.007 for slag and 0.05 for concrete
¶ Using emanation factor of 0.05 for slag and concrete

It can be seen from Table D8 that the calculated values used in this study are within the range of the measured values presented. However, it should be noted that the range of measured values is highly variable and the full range of concrete that could be used in building materials is not shown in the table.

D3.4.2 Doses from external irradiation

External doses from living in a building constructed using materials containing slag were estimated using the approach described by the EC (1999). This approach was developed from a model comparison exercise reported by the EC (1997). The external dose was calculated using the following formula,

$$D_{ext} = \Sigma_R \; A_R \times F_{slag} \times DC_{ext,R} \times T \times f_I$$

where A_R = Activity concentration of radionuclide R in slag (Bq g^{-1}), see Table D1

F_{slag} = Fraction of slag used in building materials, 35% (Castlecement, 2000)

$DC_{ext,R}$ = External dose rate for exposure in a room, for radionuclide R (Sv h^{-1} per Bq g^{-1}), see below.

T = Time at location, 8760 h y^{-1}

f_I = Fraction of time indoors, 0.9 (Robinson, 1996)

The values for $DC_{ext,R}$ for the uranium-238 decay chain, the thorium-232 decay chain and potassium-40 were obtained from Radiation Protection 112 (EC, 1999) (it should be noted that the radium-226 dose rate is taken to be representative of the dose rate for the whole of the uranium-238 decay chain as the value is similar to that reported for uranium-238 by the EC (1997)). The contribution of members of the uranium-235 decay chain was ignored on the basis of the low activity concentrations. The values of $DC_{ext,R}$ given by the EC (1999) originated from its earlier work (EC, 1997), in which models that had been developed over the last 20 years were compared. All the models examined in the report gave similar values for the dose rates despite different approaches to the problem. A comparison of dose rates predicted by some of these models with measured indoor dose rates found agreement within 15% (Brown, 1982). The dose rates used are given below:

^{238}U 6.44 10^{-7} Sv h^{-1} per Bq g^{-1}

^{232}Th 7.70 10^{-7} Sv h^{-1} per Bq g^{-1}

^{40}K 5.60 10^{-8} Sv h^{-1} per Bq g^{-1}

The contribution from daughters in the decay chains are included in these values. These dose rates are for a person in a room that has been constructed using concrete for the walls, floor and ceiling.

External doses to members of the public living in homes constructed from materials containing slag, from radionuclides in all constituents of the materials, and to members of the public living in homes constructed from similar building materials that do not contain slag were determined using the same approach, using the activity concentrations presented in Table D1.

D3.4.3 Results

The estimated radon concentrations in buildings are presented in Table D9. When an emanation fraction of 0.7% for slag is assumed, the concentration of radon arising from slag in the building materials is 0.6 Bq m^{-3}, and the total radon concentration arising inside buildings constructed using materials containing slag is 7.0 Bq m^{-3}. The estimated radon concentration inside buildings constructed using 'standard' (ie non slag containing) materials is 9.9 Bq m^{-3}, this implies that the use of slag as a component of building materials causes a reduction in the radon concentration inside the building. However, when an emanation fraction of 5% is assumed for both slag and concrete, the estimated concentration of radon arising from slag in the building materials is 4.4 Bq m^{-3}, and the total radon concentration inside buildings constructed using materials containing slag is estimated to be 10.8 Bq m^{-3}. Thus for these assumptions the use of slag increases radon concentrations with respect to the use of 'standard' building materials.

TABLE D9 Estimated radon concentrations in buildings

Scenario	Concentration (Bq m^{-3})		
	^{222}Rn	^{220}Rn	Total
House constructed from 'standard' materials			
Total radon concentration from building materials	9.9	0.015	9.9
House constructed from materials containing slag (35%) *Slag emanation fraction 0.7%*			
Radon concentration from slag component	0.6	0.001	0.6
Total radon concentration from building materials	7.0	0.011	7.0
Total additional radon concentration compared to 'standard' materials	-2.9	-0.004	-2.9
House constructed from materials containing slag (35%) *Slag emanation fraction 5%*			
Radon concentration from slag component	4.4	0.009	4.4
Total radon concentration from building materials	10.8	0.019	10.8
Total additional radon concentration compared to 'standard' materials	0.9	0.004	0.9

Clearly the impact on radon concentrations in homes of using slag in building materials depends on the radon emanation rates. The two values of radon emanation fraction considered here, 0.7% and 5%, resulted in, respectively, a decrease or increase in radon concentration compared to 'standard' (ie non slag containing) materials. The lower value, 0.7%, was chosen on the basis of experimental measurements of radon emanation from slag (Bruzzi et al, 1992). However, it was considered that this might not be appropriate for slag within a concrete matrix, and thus a higher value, 5%, appropriate for concrete, was also used. The available experimental data on radon emanation from concrete containing slag is not sufficient to allow this factor to be more accurately defined for this study.

It can be seen that the slag emanation fraction has a significant impact on the estimated radon concentration in the room. It should also be noted, following the discussion of radon exhalation rates in section 3.4.3, that these radon concentrations are likely to be overestimates of the actual levels.

Doses to adults from both radon inhalation and external irradiation are presented in Table D10. The estimated dose to a resident from external irradiation is 0.79 mSv y^{-1}, from building materials containing slag. The estimated external dose from 'standard' building materials is 0.76 mSv y^{-1}. The excess external dose resulting from the use of slag in building materials is therefore 0.03 mSv y^{-1}. Table D11 gives a more detailed breakdown of the contribution of the different radionuclides to the total external dose. It can be seen that the radionuclides in the uranium-238 chain contribute most significantly to the total dose. The dose from potassium-40 is reduced because slag contains a lower concentration of this radionuclide than concrete, however the increase in dose resulting from greater concentrations of uranium-238 and thorium-232 is greater than the reduction in dose caused by lower concentrations of potassium-40.

TABLE D10 Doses to adults from residence in a house built using slag constituents

Scenario	Individual dose (Sv y^{-1})		
	Radon inhalation	External	Total
House constructed from 'standard materials'			
Total dose from building materials	5.10E-04	7.58E-04	1.27E-03
House constructed from materials containing slag (35%)			
Slag emanation fraction 0.7%			
Dose from slag component	3.18E-05	2.98E-04	3.29E-04
Total dose from building materials	3.63E-04	7.90E-04	1.15E-03
Total additional dose compared to 'standard' materials	-1.47E-04	3.22E-05	-1.15E-04
House constructed from materials containing slag (35%)			
Slag emanation fraction 5%			
Dose from slag component	2.27E-04	2.98E-04	5.25E-04
Total dose from building materials	5.59E-04	7.90E-04	1.35E-03
Total additional dose compared to 'standard materials'	4.85E-05	3.22E-05	8.07E-05

The total estimated dose to a member of the public living in a house constructed using slag is 1.35 mSv y^{-1}, this is assuming an emanation fraction of 5% for all building materials. The estimated dose from living in a house constructed with 'standard' building materials is 1.27 mSv y^{-1}, therefore the excess dose resulting from the use of slag is 0.08 mSv y^{-1}. When an emanation fraction of 0.7% for slag is used, the reduction in the radon concentration in the room air leads to a reduction in the radon inhalation dose and a dose that is lower overall by 0.12 mSv y^{-1} than that for 'standard' building materials. It can therefore be concluded that the use of slag in building materials does not have a significant radiological impact.

TABLE D11 Contribution of the radionuclide chains to the external dose from building materials

Scenario	Individual dose (Sv y^{-1})			
	^{238}U	^{232}Th	^{40}K	Total
House constructed from materials containing slag (35%)				
Dose from slag component	$1.56\ 10^{-4}$	$1.04\ 10^{-4}$	$3.71\ 10^{-5}$	$2.60\ 10^{-4}$
Total dose from building materials	$3.87\ 10^{-4}$	$2.22\ 10^{-4}$	$1.81\ 10^{-4}$	$7.90\ 10^{-4}$
House constructed from 'standard materials'				
Total dose from building materials	$3.55\ 10^{-4}$	$1.82\ 10^{-4}$	$2.21\ 10^{-4}$	$7.58\ 10^{-4}$
Total additional dose from using slag	$3.20\ 10^{-5}$	$4.04\ 10^{-5}$	$-4.02\ 10^{-5}$	$3.22\ 10^{-5}$

D4 References

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APPENDIX E

DOSES TO WORKERS AT THE STEEL WORKS

E1 Introduction

There are various areas at the steel works where exposure to radionuclides in wastes or slag could occur. Workers at the blast furnace, workers at the lagoons and workers digging out cooled slag are considered. The doses to each of these groups of workers are considered separately. This is because the size of the site makes it unlikely that workers exposed at the lagoons would also be exposed at the blast furnace. Also, the information from Scunthorpe steel works (Giles and Harvey, 2000) indicates that the workers have specific jobs and would only spend time in one area. The sintering process is not considered as it is an automated process and there is therefore limited worker exposure at the sinter plant. It is also assumed that most of the radionuclides are removed from the raw material during sintering and in the blast furnace and therefore exposures at the BOS furnace will be negligible.

Each set of workers will be exposed via four main pathways: external irradiation, inhalation of dust, inadvertent ingestion of dust and skin contamination. The total dose to a worker is then the sum of the doses from each of these pathways. The methodologies and data used to determine doses to the different workers and the results obtained are described in the following sections.

E2 Dose to workers at the blast furnace

E2.1 Methodology

External irradiation

External exposure is assumed to occur from a 100 tonne slag pile that is collected from the blast furnace and stored nearby awaiting removal. It is recognised that the piles of slag collected near the blast furnace are unlikely to be as large as this, however, there will be more than one slag pile and a typical worker is likely to be exposed to more than one pile of slag. Exposure to one, large source is intended to be a conservative representation. The geometry assumed for the source is a half cylinder. The worker is assumed to be 1.5m from the source. The external dose, D_{ext}, was determined using the following equation.

$$D_{ext} = \Sigma_R \; A_R \times DC_{ext,R} \times T$$

where

D_{ext} = Dose from external exposure to cooling slag, Sv y^{-1} (1.5m from source)

A_R = Activity concentration of radionuclide R in slag, Bq g^{-1}, see below

$DC_{ext,R}$ = Dose coefficient for external exposure, for radionuclide R, Sv h^{-1} per Bq g^{-1}

T = Duration of exposure, 2000 h y^{-1} is assumed (Penfold et al, 1997)

Values of $D_{ext,\ R}$ were determined using energy specific dose equivalent rate factors from IAEA (1992). The external dose from slag was calculated using the activity concentrations in slag of uranium-238, uranium-235 and thorium-232 given in Table E1, including all daughters in secular equilibrium.

TABLE E1 Radionuclide concentrations and resuspended dust concentrations

Worker location	Dust concentration in air (kg m^{-3})	Activity concentration of dust (Bq kg^{-1})	
Blast furnace	$7.5\ 10^{-7}$ *	^{210}Pb	$8\ 10^{3}$ †
		^{210}Po	$2.8\ 10^{3}$ †
		^{238}U ‡	88 *
		^{232}Th §	49 *
		^{235}U	4 *
Dust lagoons	$1.0\ 10^{-7}$ ¶	^{210}Pb	360 *
		^{210}Po	88 *
Digging cooled slag	$4.0\ 10^{-7}$ ¶	^{238}U	88 *
		^{232}Th	49 *
		^{235}U	4 *

* Giles and Harvey (2000)

† Harvey (1999)

‡ Including all daughters, except ^{210}Pb, ^{210}Bi and ^{210}Po, in secular equilibrium

§ Including all daughters in secular equilibrium

¶ Simmonds et al (1995)

Inhalation of dust

Workers at the blast furnace are assumed to be inhaling dust in the atmosphere for the full working year. In the absence of experimental data it has been assumed that the concentrations of lead-210 and polonium-210 in the dust in the working area around the blast furnace are the same as those in the furnace off gases. This is a conservative estimate as workers in the area will primarily be exposed to dust from the metal and slag. It was further assumed that, as these workers are also exposed to slag piles, the remaining members of the uranium-238, uranium-235 and thorium-232 decay chains are present with the activity concentrations measured in slag, see Table E1 for activity concentrations. It is assumed that workers at the blast furnace spend their time working outdoors, the inhalation rate for these workers is taken from Robinson (1996). The dust levels at the blast furnace are measured values from Corus (Giles and Harvey,

2000). The workers are known to wear protective equipment (Giles and Harvey, 2000). The effectiveness of respiratory equipment varies between industries depending on its type and perceived importance. In the absence of more detailed information it has been pessimistically assumed that the respiratory protection is 50% effective, this is intended to represent a poorly fitted fibre mask, not worn rigorously. The inhalation dose, D_{inh}, was determined using the following equation.

$$D_{inh} = \Sigma_R A_R \times DC_{inh,w,R} \times C_d \times I_{r,w} \times T \times P$$

where

D_{inh} = Dose from the inhalation of radionuclides in blast furnace dust, Sv y^{-1}

A_R = Activity concentration of radionuclide R in blast furnace dust, Bq g^{-1}, see Table E1

$DC_{inh,w.R}$ = Inhalation dose coefficient for workers, for radionuclide R, Sv Bq^{-1} (ICRP, 1994)

C_d = Concentration of dust in the air, 7.5 10^{-4} g m^{-3} (Giles and Harvey, 2000)

$I_{r,w}$ = Inhalation rate for workers, 1.18 m^3 h^{-1} (Robinson, 1996)

T = Time spent by workers inhaling dust, 2000 h y^{-1} (Penfold et al, 1997)

P = Protection factor for wearing respiratory equipment, 0.5 (Penfold et al, 1997)

Inadvertent ingestion of dust

It is pessimistically assumed that a worker ingests dust at a rate of 5 mg h^{-1} (Robinson, 1996) for 2000 hours per year. The dose from inadvertent ingestion has been estimated using the following equation.

$$D_{ing} = \Sigma_R A_R \times DC_{ing,w,R} \times Ing_{r,w} \times T$$

where

D_{ing} = Dose from the inadvertent ingestion of radionuclides in blast furnace dust, Sv y^{-1}

A_R = Activity concentration of radionuclide R in blast furnace dust, Bq g^{-1}, see Table E1

$DC_{ing,w,R}$ = Ingestion dose coefficient for workers, for radionuclide R, Sv Bq^{-1} (ICRP, 1994)

$Ing_{r,w}$ = Ingestion rate for workers, 5 10^{-3} g h^{-1} (Robinson, 1996)

T = Exposure duration, 2000 h y^{-1} (Penfold et al, 1997)

Skin contamination

Workers are assumed to be exposed to contaminated dust on their skin and clothing for an entire working year, 2000 h. The thickness of the deposit on the skin is assumed to be 0.01 cm^2. Beta irradiation of the skin and gamma irradiation of the skin are treated separately because of the greater penetration of gamma energy. For beta irradiation it is assumed that the hands, face and some of the arms and neck are covered with dust (1200 cm^2 in total of UVR exposed skin), since workers at the steel plant are usually wearing overalls. For gamma irradiation it is assumed that all of the UVR exposed skin will be irradiated as a result of dust on skin and clothing. The methodology described in NRPB (1997) has been used.

The activity concentration of radionuclide R per unit area on the skin (Bq cm^{-2}) is given by

$$A_{skin,R} = \rho \times d \times A_R$$

where

A_R = Activity concentration of radionuclide R in blast furnace dust, Bq g^{-1}, see Table E1

ρ = Density of dust, g cm^{-3}, see below

d = Thickness of dust on the skin, 0.01cm (Penfold et al, 1997)

Blast furnace dust, which has enhanced concentrations of lead-210 and polonium-210, has been assumed to have a density of 0.75 g cm^{-3}, which is a typical value for an ash like substance (James and Lord, 1992).

The equivalent dose, from radionuclide R, to the skin area that is exposed to beta and gamma radiation, $H_{skin,\beta,R}$ and $H_{skin,\gamma,R}$ respectively, can then be calculated as follows

$$H_{skin,\beta,R} = A_{skin,R} \times T \times \beta_{skin,R}$$

$$H_{skin,\gamma,R} = A_{skin,R} \times T \times \gamma_{skin,R}$$

where

T = Time of exposure, 2000 h y^{-1}

$\beta_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for β irradiation, for radionuclide R, Sv h^{-1} per Bq cm^{-2} (Harvey et al, 1993)

$\gamma_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for γ irradiation, for radionuclide R, Sv h^{-1} per Bq cm^{-2} (Harvey et al, 1993)

Effective doses to skin from beta and gamma irradiation, $D_{skin,\beta,R}$ and $D_{skin,\gamma,R}$ are calculated separately, summed together and then summed over radionuclides to give a total effective dose from skin contamination, D_{skin} (Sv y^{-1}).

$$D_{skin,\beta} = H_{skin,\beta R} \times W_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

Where

w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994)

EXP_{area} = Area of skin exposed to β irradiation, 1200 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2 (NRPB, 1997)

$$D_{skin,\gamma,R} = H_{skin,\gamma,R} \times W_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where

w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994)

EXP_{area} = Area of skin exposed to γ irradiation, 3000 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2

D_{skin} = Σ_R ($D_{skin,\beta,R}$ + $D_{skin,\gamma,R}$)

Total dose to a worker at the blast furnace

The total dose to a worker at the blast furnace is the sum from all the exposure pathways.

D_{BF} = D_{ext} + D_{inh} + D_{ing} + D_{skin}

E2.2 Results

The doses to workers at the blast furnace are presented in Table E2. The total dose to a worker at the blast furnace is 84 $\mu Sv\ y^{-1}$, the majority of this dose results from the inadvertent ingestion of dust containing enhanced levels of naturally occurring radionuclides, (74% of the total dose). The second most significant pathway is inhalation of dust, which is responsible for 23% of the total dose. Considering the conservative assumptions that have been made in the calculation of these doses, it is likely that the actual levels of exposure will be lower.

TABLE E2 Doses to workers at the blast furnace

	Dose (Sv y^{-1})				
Radionuclide	External	Inhalation	Ingestion	Skin contamination	Total
^{238}U	$4.1\ 10^{-7}$ *	$9.1\ 10^{-7}$ †	$4.1\ 10^{-7}$ †	$1.2\ 10^{-7}$	$1.8\ 10^{-6}$
^{234}U				$1.3\ 10^{-10}$	$1.3\ 10^{-10}$
^{230}Th				$1.0\ 10^{-9}$	$1.0\ 10^{-9}$
^{226}Ra				$4.6\ 10^{-8}$	$4.6\ 10^{-8}$
^{210}Pb		$8.2\ 10^{-6}$	$5.5\ 10^{-5}$	$1.3\ 10^{-6}$	$6.4\ 10^{-5}$
^{210}Po		$5.5\ 10^{-6}$	$6.7\ 10^{-6}$	$2.0\ 10^{-13}$	$1.2\ 10^{-5}$
^{232}Th	$3.2\ 10^{-7}$ *	$2.1\ 10^{-6}$ *	$4.3\ 10^{-7}$ *	$3.5\ 10^{-10}$	$2.8\ 10^{-6}$
^{228}Ra				$8.9\ 10^{-8}$	$8.9\ 10^{-8}$
^{228}Th				$3.4\ 10^{-8}$	$3.4\ 10^{-8}$
^{235}U	$5.1\ 10^{-9}$ *	$2.6\ 10^{-6}$ *	$7.9\ 10^{-8}$ *	$1.1\ 10^{-9}$	$2.7\ 10^{-6}$
^{231}Pa				$6.5\ 10^{-11}$	$6.5\ 10^{-11}$
^{227}Ac				$1.6\ 10^{-9}$	$1.6\ 10^{-9}$
Total	$7.3\ 10^{-7}$	$1.9\ 10^{-5}$	$6.2\ 10^{-5}$	$1.6\ 10^{-6}$	$8.4\ 10^{-5}$

* Includes all members of the decay chain.

† Includes all members of the decay chain except ^{210}Pb, ^{210}Bi and ^{210}Po.

E3 Doses to workers at the lagoons

E3.1 Methodology

External irradiation

Since the best estimate assumption is that the only radionuclides present in the slurry disposed to the lagoons are lead-210 and polonium-210, see discussion in Section 3.3 of main text, external exposure to workers at the lagoons is assumed to be negligible. The intensity of gamma emissions from lead-210 and polonium-210 are very low and the workers at the lagoons would be inside a mechanical excavator, which would provide shielding. Compared with inhalation and inadvertent ingestion of dust this pathway is therefore considered to be insignificant.

Inhalation of dust

Workers at the lagoons are assumed to be inhaling dust in the atmosphere for the full working year. The dust in the lagoons comes from the wet gas cleaning system for the blast furnace and BOS furnace. It is assumed that workers at the lagoons spend all of their time outdoors, the inhalation rate for these workers is taken from Robinson (1996). The dust levels at the lagoons are assumed to be $1\ 10^{-4}$ g m^{-3} (Simmonds et al, 1995). As with the workers at the blast furnace, workers at the lagoons are known to wear respiratory equipment, which is assumed to be 50% effective. The inhalation dose, D_{inh}, was determined using the following equation.

$D_{inh} = \Sigma_R A_R \times DC_{inh,w,R} \times C_d \times I_{r,w} \times T \times P$

where

D_{inh} = Dose from the inhalation of radionuclides in dust resuspended from lagoons, Sv y^{-1}

A_R = Activity concentration of radionuclide R in dust, Bq g^{-1}, see Table E1

$DC_{inh,w.R}$ = Inhalation dose coefficient for workers, for radionuclide R, Sv Bq^{-1} (ICRP, 1994)

C_d = Concentration of dust in the air, 1 10^{-4} g m^{-3} (Simmonds et al, 1995)

$I_{r,w}$ = Inhalation rate for workers, 1.18 m^3 h^{-1} (Robinson, 1996)

T = Exposure duration, 2000 h y^{-1} (Penfold et al, 1997)

P = Protection factor for wearing protective equipment, 0.5 (Penfold et al, 1997)

Inadvertent ingestion of dust

The inadvertent ingestion rate of dust for workers at the lagoons has been assumed to be 5 mg h^{-1} (Robinson, 1996). The dose from inadvertent ingestion, D_{ing}, was determined using the following equation.

$D_{ing} = \Sigma_R A_R \times DC_{ing,w,R} \times Ing_{r,w} \times T$

where

D_{ing} = Dose from the ingestion of radionuclides in dust at the lagoons, Sv y^{-1}

A_R = Activity concentration of radionuclide R in dust, Bq g^{-1}, see Table E1

$DC_{ing,w,R}$ = Ingestion dose coefficient for workers, for radionuclide R, Sv Bq^{-1} (ICRP, 1994)

$Ing_{r,w}$ = Ingestion rate for workers, 5 10^{-3} g h^{-1} (Robinson, 1996)

T = Exposure duration dust, 2000 h y^{-1} (Penfold et al, 1997)

Skin contamination

The same assumptions for workers at the blast furnace have been made for workers at the lagoons. Workers are assumed to be exposed to contaminated dust on their skin and clothing for an entire working year, 2000 h. The thickness of the deposit on the skin is assumed to be 0.01 cm (Penfold et al, 1997). Beta irradiation of the skin and gamma irradiation of the skin are treated separately because of the greater penetration of gamma energy. For beta irradiation it is assumed that the hands, face and some of the arms and neck are covered with dust (1200 cm^2 in total of UVR exposed skin), since workers at the steel plant are usually wearing overalls. For gamma irradiation it is assumed that all of the

UVR exposed skin will be irradiated as a result of dust on skin and clothing. The methodology described in NRPB (1997) has been used.

The activity concentration of radionuclide R per unit area on the skin (Bq cm^{-2}) is given by

$A_{skin,R} = \rho \times d \times A_R$

where

A_R = Activity concentration of radionuclide R in dust, Bq g^{-1}, see Table E1

ρ = Density of dust collected from the wet gas cleaning system, 0.75 g cm^{-3} (a typical value for an ash like substance (James and Lord, 1992)

d = Thickness of dust on the skin, 0.01cm (Penfold et al, 1997)

The equivalent dose, from radionuclide R, to the skin area that is exposed to beta and gamma radiation, $H_{skin,\beta,R}$ and $H_{skin,\gamma,R}$ respectively, can then be calculated as follows

$H_{skin,\beta,R} = A_{skin,R} \times T \times \beta_{skin,R}$

$H_{skin,\gamma,R} = A_{skin,R} \times T \times \gamma_{skin,R}$

where

T = Exposure duration, 2000 h y^{-1}

β_{skin} = The skin equivalent dose rate to the basal layer of the skin epidermis for β irradiation, for radionuclide R, Sv h^{-1} per Bq cm^{-2} (Harvey et al, 1993)

γ_{skin} = The skin equivalent dose rate to the basal layer of the skin epidermis for γ irradiation, for radionuclide R, Sv h^{-1} per Bq cm^{-2} (Harvey et al, 1993)

Effective doses to skin from beta and gamma irradiation, $D_{skin,\beta,R}$ and $D_{skin,\gamma,R}$ are calculated separately, summed together and then summed over radionuclides to give a total effective dose from skin contamination, D_{skin} (Sv y^{-1}).

$$D_{skin,\beta.R} = H_{skin,\beta,R} \times W_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

Where

w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994)

EXP_{area} = Area of skin exposed to β irradiation, 1200 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2 (NRPB, 1997)

$$D_{skin,\gamma,R} = H_{skin,\gamma,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where

w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994)

EXP_{area} = Area of skin exposed to γ irradiation, 3000 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2

D_{skin} = Σ_R ($D_{skin,\beta,R}$ + $D_{skin,\gamma,R}$)

Total dose to a worker at the lagoons

The total dose to a worker at the lagoons is the sum from all the exposure pathways.

D_{lagoon} = D_{inh} + D_{ing} + D_{skin}

E3.2 Results

The doses to workers at the slurry lagoons are presented in Table E3. The total dose to a worker at the dust lagoons is 2.8 μSv y^{-1}, the majority of this dose, 95%, results from the inadvertent ingestion of dust resuspended from the lagoons. Considering the conservative assumptions made, especially regarding inadvertent ingestion, it is expected that actual levels of exposure would be significantly lower.

TABLE E3 Doses to workers at slurry lagoons

	Dose (Sv y^{-1})			
Radionuclide	Inhalation	Ingestion	Skin contamination	Total
^{210}Pb	4.9 10^{-8}	2.5 10^{-6}	5.7 10^{-8}	2.6 10^{-6}
^{210}Po	2.3 10^{-8}	2.1 10^{-7}	6.3 10^{-15}	2.3 10^{-7}
Total	7.2 10^{-8}	2.7 10^{-6}	5.7 10^{-8}	2.8 10^{-6}

E4 Doses to workers digging cooled slag

E4.1 Methodology

External irradiation

External exposure to cooling slag is assumed to occur from a 100 tonne slag pile that is collected from the blast furnace/BOS furnace and stored, awaiting removal. It is recognised that the piles of slag are unlikely to be as large as this, however, there will be more than one slag pile and a typical worker is likely to be

exposed to more than one pile of slag. Exposure to one, large source is intended to be a conservative representation. The worker is assumed to be 1.5m from the source. The worker will also be inside an excavator truck, or a dump truck, which will provide shielding.

$$D_{ext} = \Sigma_R \; A_R \times DC_{ext,R} \times T \times SF$$

where

D_{ext} = Dose from external exposure to cooling slag, Sv y^{-1} (1.5m from source)

A_R = Activity concentration of radionuclide R in slag, Bq g^{-1}, see Table E1

$DC_{ext,R}$ = Dose coefficient for external exposure, for radionuclide R, Sv h^{-1} per Bq g^{-1}

T = Duration of exposure, 2000 h y^{-1} is assumed (Penfold et al, 1997)

SF = Shielding factor of 0.7 is used to represent the excavator (Wilkins et al, 2002)

Values of $DC_{ext,R}$ were determined using energy specific dose equivalent rate factors from IAEA (1992). The shielding factor of 0.7 was used to represent the shielding afforded to an operative by his excavator (13mm of steel was assumed). This factor is strictly applicable to the radionuclides caesium-137 and americium-241 (Wilkins, 2002). An excavator may provide better shielding for the lower energy gamma radiation present in this scenario, however 0.7 will be assumed as a conservative estimate.

Inhalation of dust

Workers digging out slag piles are assumed to be inhaling dust in the atmosphere for the full working year. The activity concentrations in the dust are assumed to be the same as the concentrations in the slag. It is assumed that workers at the slag piles spend all of their time outdoors, the inhalation rate for these workers is taken from Robinson (1996). The dust levels at the slag piles are assumed to be 4 10^{-4} g m^{-3} (Simmonds et al, 1995). As with the workers at the blast furnace, workers at the slag piles are known to wear respiratory equipment, which is assumed to be 50% effective.

$$D_{inh} = \Sigma_R \; A_R \times DC_{inh,w,R} \times C_d \times I_{r,w} \times T \times P$$

where

D_{inh} = Dose from the inhalation of radionuclides in dust from the slag, Sv y^{-1}

A_R = Activity concentration of radionuclide R in dust, Bq g^{-1}, see Table E1

$DC_{inh,w.R}$ = Inhalation dose coefficient for workers, for radionuclide R, Sv Bq^{-1} (ICRP, 1994)

C_d = Concentration of dust in the air, 4 10^{-4} g m^{-3} (Simmonds et al, 1995)

$I_{r,w}$ = Inhalation rate for workers, 1.18 m^3 h^{-1} (Robinson, 1996)

T = Exposure duration, 2000 h y^{-1} (Penfold et al, 1997)

P = Protection factor for wearing protective equipment, 0.5 (Penfold et al, 1997)

Inadvertent ingestion of dust

The inadvertent ingestion rate of dust assumed for workers at the slag piles is 5 mg h^{-1}, Robinson (1996).

$$D_{ing} = \Sigma_R A_R \times DC_{ing,w,R} \times Ing_{r,w} \times T$$

where

D_{ing} = Dose from the ingestion of radionuclides in slag dust, Sv y^{-1}

A_R = Activity concentration of radionuclide R in slag dust, Bq g^{-1}, see Table E1

$DC_{ing,w,R}$ = Ingestion dose coefficient for workers, for radionuclide R, Sv Bq^{-1} (ICRP, 1994)

$Ing_{r,w}$ = Ingestion rate for workers, 5 10^{-3} g h^{-1} (Robinson, 1996).

T = Exposure duration, 2000 h y^{-1} (Penfold et al, 1997).

Skin contamination

The same assumptions for workers at the blast furnace have been made for workers at the slag piles. Workers are assumed to be exposed to contaminated dust on their skin and clothing for an entire working year, 2000 h. The thickness of the deposit on the skin is assumed to be 0.01 cm (Penfold et al, 1997). Beta irradiation of the skin and gamma irradiation of the skin are treated separately because of the greater penetration of gamma energy. For beta irradiation it is assumed that the hands, face and some of the arms and neck are covered with dust (1200cm^2 in total of UVR exposed skin), since workers at the steel plant are usually wearing overalls. For gamma irradiation it is assumed that all of the UVR exposed skin will be irradiated as a result of dust on skin and clothing. The methodology described in NRPB (1997) has been used.

The activity concentration of radionuclide R per unit area on the skin (Bq cm^{-2}) is given by:

$$A_{skin,R} = \rho \times d \times A_R$$

where

A_R = Activity concentration of radionuclide R in dust, Bq g^{-1}, see Table E1

ρ = Density of dust, g cm^{-3} (see below)

d = Thickness of dust on the skin, 0.01cm (Penfold et al, 1997)

British Standards (BSI, 1990; 1992) describe the density of fine aggregates to be between 1.44 – 1.12 g cm^{-3}; 1.3 has been chosen to represent slag dust.

The equivalent dose, from radionuclide R, to the skin area that is exposed to beta and gamma radiation, $H_{skin,\beta,R}$ and $H_{skin,\gamma,R}$, respectively, can then be calculated as follows

$$H_{skin,\beta,R} = A_{skin,R} \times T \times \beta_{skin,R}$$

$$H_{skin,\gamma,R} = A_{skin,R} \times T \times \gamma_{skin,R}$$

where

T = Exposure duration, 2000 h y^{-1} (Penfold et al, 1997).

$\beta_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for β irradiation, for radionuclide R, Sv h^{-1} per Bq cm^{-2} (Harvey et al, 1993)

$\gamma_{skin,R}$ = The skin equivalent dose rate to the basal layer of the skin epidermis for γ irradiation, for radionuclide R, Sv h^{-1} per Bq cm^{-2} (Harvey et al, 1993)

Effective doses to skin from beta and gamma irradiation, $D_{skin,\beta,R}$ and $D_{skin,\gamma,R}$, are calculated separately, summed together and then summed over radionuclides to give a total effective dose from skin contamination, D_{skin} (Sv y^{-1}).

$$D_{skin,\beta,R} = H_{skin,\beta,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where

w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994)

EXP_{area} = Area of skin exposed to β irradiation, 1200 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2 (NRPB, 1997)

$$D_{skin,\gamma,R} = H_{skin,\gamma,R} \times w_{skin} \times \frac{EXP_{area}}{TOTAL_{area}}$$

where

w_{skin} = Tissue weighting factor for skin, 0.01 (ICRP, 1994)

EXP_{area} = Area of skin exposed to γ irradiation, 3000 cm^2

$TOTAL_{area}$ = Total area of UVR exposed skin, 3000 cm^2

D_{skin} = Σ_R ($D_{skin,\beta,R}$ + $D_{skin,\gamma,R}$)

Total dose to a worker digging cooled slag

The total dose to a worker digging out a cooled slag heap is the sum from all the exposure pathways.

D_{Slag} = D_{ext} + D_{inh} + D_{ing} + D_{skin}

E4.2 Results

The doses to workers digging cooled slag are presented in Table E4. The total dose to a worker is 5.7 $\mu Sv\ y^{-1}$, the dominant pathway is the inhalation of dust. Inhalation of dust accounts for 55% of the total dose, the second most significant pathway is inadvertent ingestion of dust, which accounts for 30% of the total dose. Given the conservative assumptions that have been made, it is anticipated that the actual levels of exposure would be lower.

TABLE E4 Doses to workers digging cooled slag

	Dose (Sv y^{-1})				
Radionuclide	External	Inhalation	Ingestion	Skin contamination	Total
^{238}U	$2.9\ 10^{-7}$ *	$6.3\ 10^{-7}$ *	$1.2\ 10^{-6}$ *	$1.2\ 10^{-7}$	$2.3\ 10^{-6}$
^{234}U				$1.3\ 10^{-10}$	$1.3\ 10^{-10}$
^{226}Th				$1.0\ 10^{-9}$	$1.0\ 10^{-9}$
^{226}Ra				$4.6\ 10^{-8}$	$4.6\ 10^{-8}$
^{232}Th	$2.3\ 10^{-7}$ *	$1.1\ 10^{-6}$ *	$4.3\ 10^{-7}$ *	$3.5\ 10^{-10}$	$1.8\ 10^{-6}$
^{228}Ra				$8.9\ 10^{-8}$	$8.9\ 10^{-8}$
^{228}Th				$3.4\ 10^{-8}$	$3.4\ 10^{-8}$
^{235}U	$3.6\ 10^{-9}$ *	$1.4\ 10^{-6}$ *	$7.9\ 10^{-8}$ *	$1.1\ 10^{-9}$	$1.5\ 10^{-6}$
^{231}Pa				$6.5\ 10^{-11}$	$6.5\ 10^{-11}$
^{227}Ac				$1.6\ 10^{-9}$	$1.6\ 10^{-9}$
Total	$5.1\ 10^{-7}$	$3.1\ 10^{-6}$	$1.7\ 10^{-6}$	$2.9\ 10^{-7}$	$5.7\ 10^{-6}$

* Includes all members of the decay chain.

E5 Conclusion

The estimated doses to workers digging cooled slag are very small, as are the doses to workers at the slurry lagoons. The highest estimated doses to workers on the steel production site are those at the blast furnace. This is a result of the conservative assumptions made about the activity concentrations in the dust and the dust concentration in the air being higher than anywhere else on the steel

production site. Inhalation and ingestion of resuspended dust are the most significant exposure pathways for every worker scenario.

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