Drigg Site:
Basic Characteristics and Evaluation
of Restoration Options

Restoration Strategies for Radioactively Contaminated
Sites and their Close Surroundings
RESTRAT - WP1.1

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Executive Summary

TD.9: Drigg Site: Basic Characteristics and Evaluation of Restoration Options

The aim of this report is to demonstrate the applicability of the decision-making approach, proposed by RESTRAT, for selecting restoration techniques to treat low level waste disposal sites. The Drigg site (as it was prior to capping the waste trenches) has been used as an example of such a site.

The decision-making procedure is based on multi-attribute utility analysis. This is superior to convention cost-benefit analysis because criteria can be incorporated without the need to convert them into common units of measurement. For the Drigg site the attributes considered are radiation-induced health effects, monetary costs and social factors. These may be further broken down into a series of sub-attributes and weighting factors. A probabilistic approach was adopted to take account of the uncertainties with the utility functions and weighting factors.

The report provides a description of the steps followed in the decision-making process. These were as follows:

- Characterisation of the site in terms of:
  - description of the area in terms of geography and topography, geology and hydrogeology, pedology, meteorology, hydrology and demography;
  - physico-chemical characterisation of the radionuclides in the water column and in the sediment;
  - characterisation of the sources of contamination and the distribution of contamination throughout the site.

- Identification and characterisation of restoration options which are applicable to the site in terms of:
  - effectiveness against contaminants (radionuclides);
  - cost of applying the restoration technique;
  - the exposure times for restoration workers during restoration.

- Determination of the radiological impact of the site through:
  - the development of compartment model to describe the site;
  - the quantification of processes, exposure pathways and exposure groups;
  - the calculation of radiological doses to the public and restoration workers.

- Ranking of the restoration options through the calculation of the attributes, utilities and weighting factors needed to perform a multi-attribute utility analysis on the various restoration options.

The waste trenches at Drigg contain about $5 \cdot 5 \times 10^5 \, \text{m}^3$ of waste through which passes an estimated $5500 \, \text{m}^3 \, \text{a}^{-1}$ of water. The principal contaminants for the site are caesium, uranium, americium and plutonium. The are found in the water and sediment of the nearby stream and drainage system. The Drigg site is situated in a sparsely populated, agricultural area. The 500 year collective dose for the untreated site (before capping) was estimated by the RESTRAT model to be 120 manSv, with the ingestion of water being the principal exposure pathway.

A number of restoration options were considered applicable to remediating this site. Multi-attribute utility analysis showed that capping was the best option. However, uncertainties associated with parameters of the site and the restoration options meant that there was no clear-cut result; both sub-surface barriers and not remediating the site could not be entirely ruled out as options. These results are in broad agreement with the remediation options which were actually carried out at the site.

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1. Terms of reference

This report is submitted as Technical Deliverable 'TD9' against the requirements of Work Package 1.1 for the Restoration Strategies for Radioactive Sites and their Close Surroundings (RESTRAT) project.

The RESTRAT project, which is funded by the European Commission under the Nuclear Fission Safety Programme has the overall objective of developing generic methodologies for ranking restoration techniques as a function of contamination and site characteristics. The development of this generic methodology is based on an analysis of existing remediation methodologies and contaminated sites, and is structured in the following steps:

1. characterisation of relevant contaminated sites;

2. identification and characterisation of relevant restoration techniques;

3. assessment of the radiological impact;

4. development and application of a selection methodology for restoration options;

5. formulation of generic conclusions and development of a manual.

This Work Package, which is jointly funded by British Nuclear Fuels plc, is concerned with the characterisation of the Drigg low level waste site and the identification of appropriate restoration techniques for this site. The aims of this Work Package are to:

1. describe the main geographic, hydrographic and geological features of the site;

2. describe the local human activities;

3. describe the major sources of radioactive contamination;

4. identify possible restoration options for the site;

5. assess the impacts of the radioactive contamination and the effect of each restoration option;

6. evaluate and rank each remediation option, in terms of the suitability for the Drigg site.

2. Introduction

The Drigg waste disposal site is located near the Cumbrian coast in the north-west of England. This former Royal Ordnance Factory site at Drigg is now owned and operated by British Nuclear Fuels plc (BNFL) for the disposal of low-level radioactive wastes. The majority of the waste received arises from Sellafield, some 6 km to the north, but the site also receives wastes from other United Kingdom licensed nuclear sites and users of radioactivity such as hospitals, universities, etc.

The site was originally developed in 1939 for the manufacture and storage of high explosives but was abandoned soon after the 1939-45 war. The site was subsequently acquired by the Ministry of Defence and BNFL. Authorisation to dispose of low-level radioactive waste was granted in 1958 and disposals began in 1959. The authorisation was issued on the basis that the boulder clay, which exists at shallow depth, would form an effective seal preventing downward infiltration of radioactivity into the underlying Triassic sandstone aquifer. The authorisation is reviewed periodically. Currently, authorisation sets maximum limits for individual waste consignments and for the total quantities of groups of radionuclides disposed of each year. In addition, limits are set on the amount of radioactivity in leachates discharged through the marine pipeline and on concentrations in the Drigg stream which flows through the site.

The Drigg site is located on a coastal margin between the sea, to the west, and the Lake District National Park, to the east. The region is essentially rural. However, there are pockets of local industry, such as Sellafield.

Disposals of radioactive waste to the trenches and vaults at Drigg have all taken place in the north-western part of the 120 ha site and the present planning consent relates only to the 36 ha northern section. The locations of Trenches 1-7 and Vault 8 are shown in Figure 1.
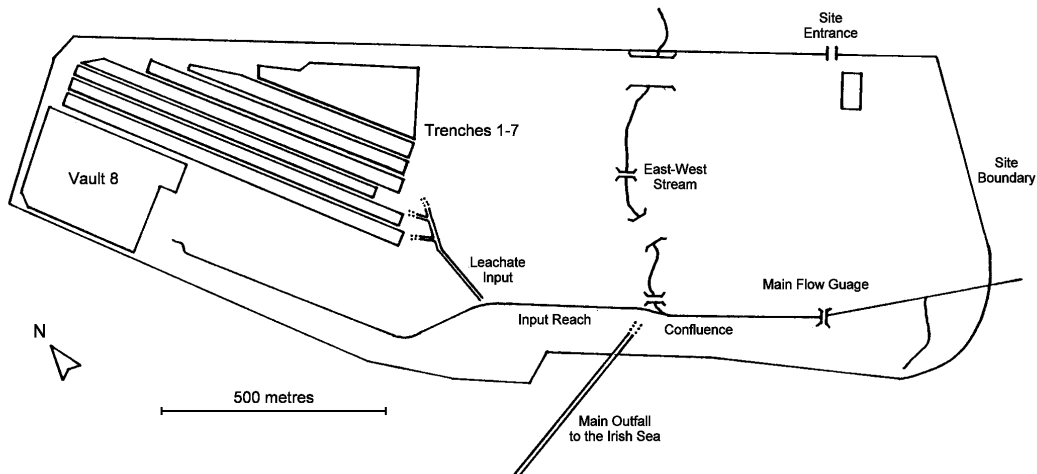
From Murdock (1992).

Figure 1   Disposal operations at the Drigg site (*circa* 1988).

Plutonium contaminated material, which was once stored in locations to the west of Trenches 1-7, is being completely removed from the site. It is intended that this area will be the site of further vaults once Vault 8 is full.

It is important to remember that Drigg is a working site where changes of practice are introduced. Consequently, the data compiled in this report, which was collected during the late 1980's and 1990's, may not be truly representative of earlier situations. The remediation of Trenches 1-7 is of particular interest to this project. However, remediation has already been applied to these. This has included:

1. capping with an impermeable layer to reduce intrusion by rain;

2. building impermeable walls on the north and east sides of the trenches to reduce horizontal intrusions of water;

3. lining parts of the trenches with impermeable clay to reduce water leaching into the surrounding rock;

4. discharging leachates through a marine outflow.

The behaviour of the trenches prior to remediation would have been different to that of their behaviour today. In order to examine the likely impact of alternative remediation approaches necessitates extrapolation back to the situation prior to remediation. It is also important to remember that this will be a hypothetical situation. Remediation to the trenches began before the Trench 7 was full.

3. General site characterisation

3.1 Description of the area

3.1.1 Geography and topography

To the north-east, the site is bounded by the Whitehaven-Barrow railway line and, to the south, by a road leading from Drigg village to the coast. Agricultural land to the north of the site is used for grazing cattle. Some vegetables are grown, with very limited cereal production (mainly for over-winter animal fodder). To the south the land is of poorer quality, suitable only for cattle and sheep grazing. An extensive Site of Special Scientific Interest (SSSI) nature reserve lies to the south of the site, characterised by sand dunes and is of local herpetological interest.

The site is near to the coast (see Figure 2). Ground elevation falls gently from about 20 m above Ordnance Datum (OD) in the north, to around 7 m in the south.

3.1.2 Geology and hydrogeology

The geology at Drigg consists of a thick and variable sequence of glacial sands and gravels, silts, clays and boulder clay, overlying a bedrock of Triassic sandstone. However, the surface of the site has been altered considerably by industrial activity and many areas consist of made ground, now extensively covered by blown sand and dense vegetation.

a) Surface soils and glacial deposits

The area around Gosforth is reported to have been subjected to two major phases of glaciation; the Main and the Scottish Re-advance Glaciation (Trotter *et al*., 1937). The deposits formed by the Main Glaciation are a complex, partially repeated, sequence of boulder clays and sands and gravels, suggesting that there was a period of withdrawal followed by a re-advance before the final phase of recession. Similarly, the Scottish Re-advance Glaciation is thought to contain periods of glacial oscillation.

An idealised glacial sequence is outlined in Table 1 but variations, especially with respect to the number of horizons and their thickness occur frequently. Geological cross sections of the site are represented in Figure 3.
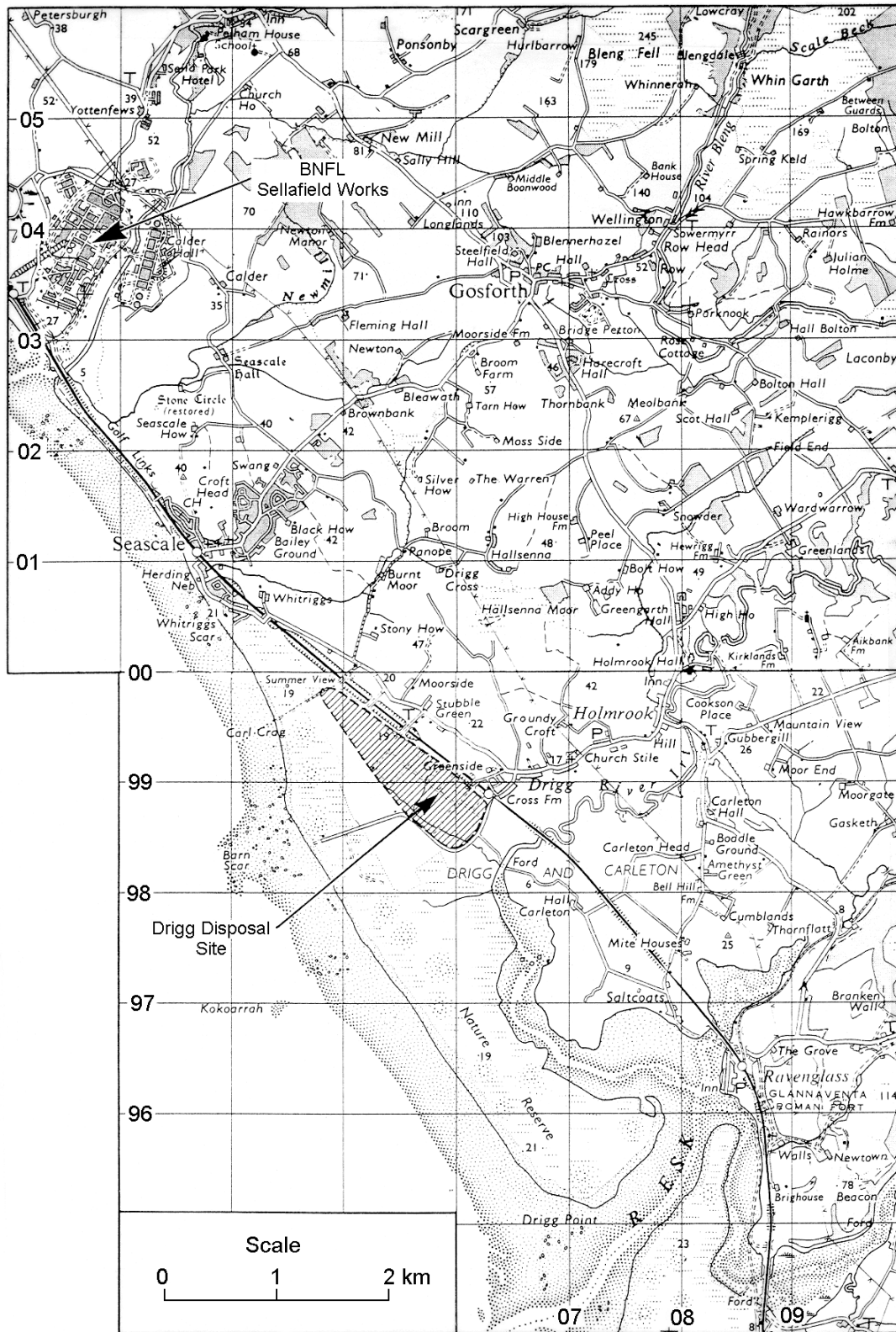
Adapted from Williams et al. (1985).

Figure 2   Location of the Drigg site.

Table 1   Generalised statigraphy of the Drigg site.

Age		Description	Maximum thickness (m)
Post Glacial		Made ground	2·0
		Blown sand	1·0
		Alluvium (in stream courses)	3·0
		Peat	3·7
Glacial	G8	Sands with pebbles, thought to be a solifluxion product	2·5
	G7	Soft, grey-brown, sandy/silty clay with traces of subangular fine to coarse gravel	2·0
	G6	Fine to medium gravel sand	3·0
	G5	Firm, brown, silty clay with sub-angular fine to coarse gravel	7·6
	G4	Fine to medium grained silty sand, occasionally with thin gravel lenses	6·0
	G3	Firm to stiff clay with interbedded sands and silts, containing pebbles and occasionally cobbles	6·8
	G2	Coarse sands and gravels with cobbles and large boulders	24·0
	G1	Stiff clay with interbedded sands and gravels	6·0
Triassic (St Bees Sandstone)			12·3[†]

† as penetrated.
From Williams et al. (1985).

a

b

Made Ground

Sand

Peat

Boulder Clay

Sand and Gravel

St Bees Sst.

------- Perched water levels encountered during drilling

—— —— Regional piezometric level

(a) Geological cross-section DDS1-Old Series (O.S.) 7-DDS33-32-13-5-31-O.S. 6DDS 8-9
(b) Geological cross-section O.S. 5-DDS24-23-O.S. 4-DDS22-6-7
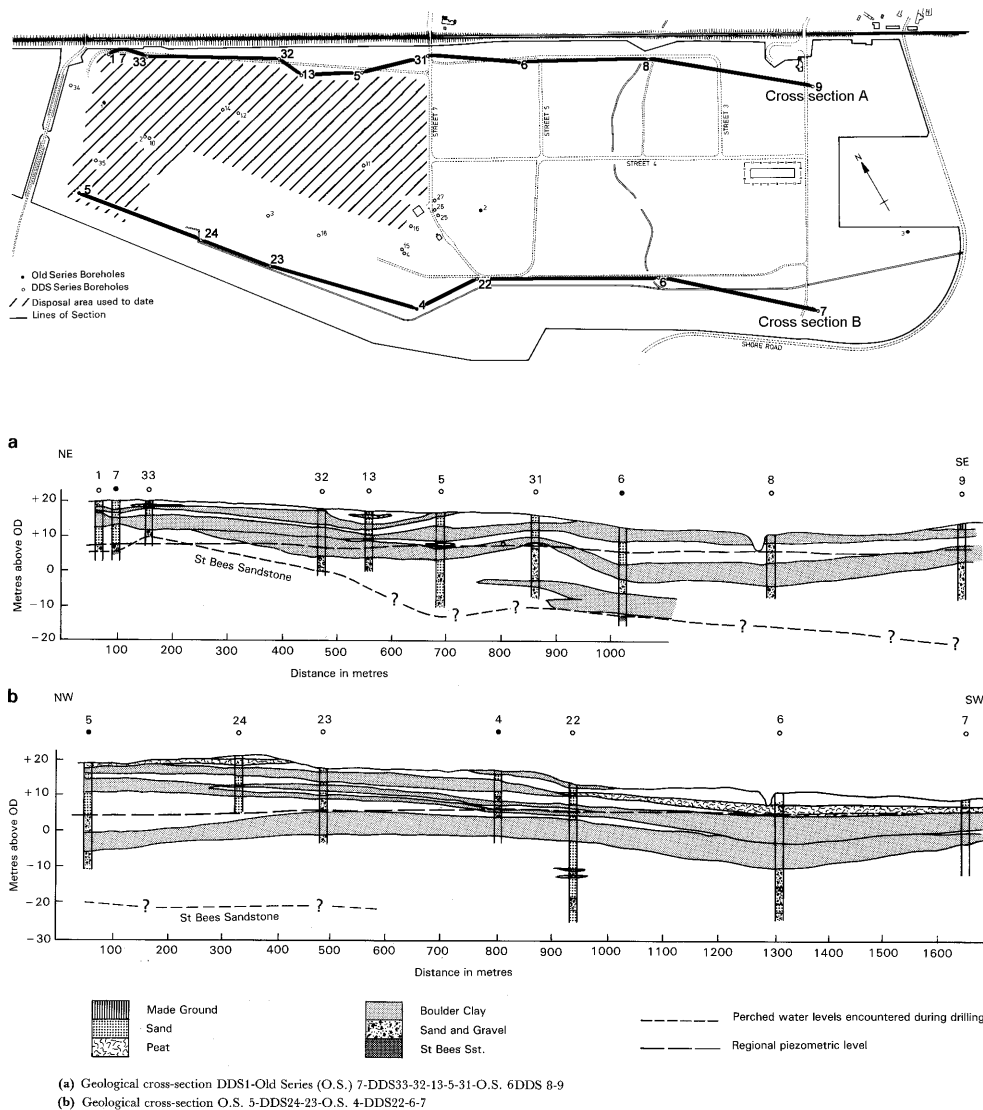
Adapted from Williams et al. (1985).

Figure 3   Geological cross sections of the Drigg site.

Those layers containing significant clay (G1, G3 and G5) are of relatively low permeability. Surface contours of the clay layers G3 and G5 are presented in Figures 4 and 5. G3 occurs over most of the site, except perhaps for an area in the north. Similarly, G5 occurs over the whole site, and is thickest in the northern half. By contrast, G1 appears in patches, directly overlying the sandstone, and is mostly restricted to the southern half of the site (Williams *et al.*, 1985).
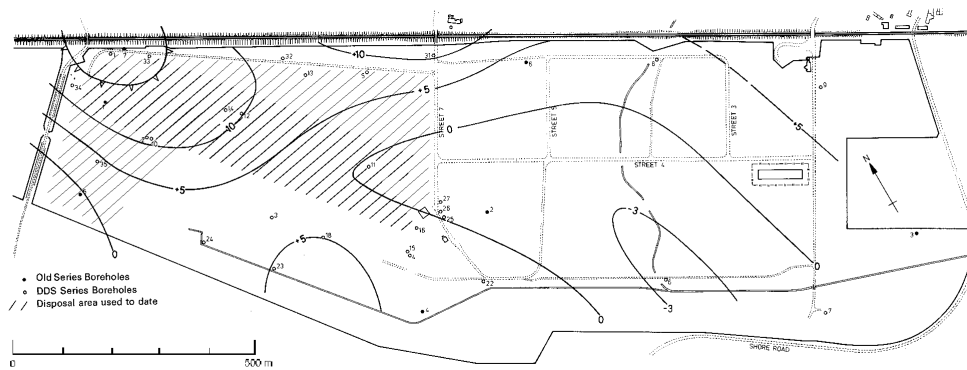
Contours in metres above Ordnance Datum.
Adapted from Williams et al. (1985).

Figure 4   Contours of surface of clay horizon G3.
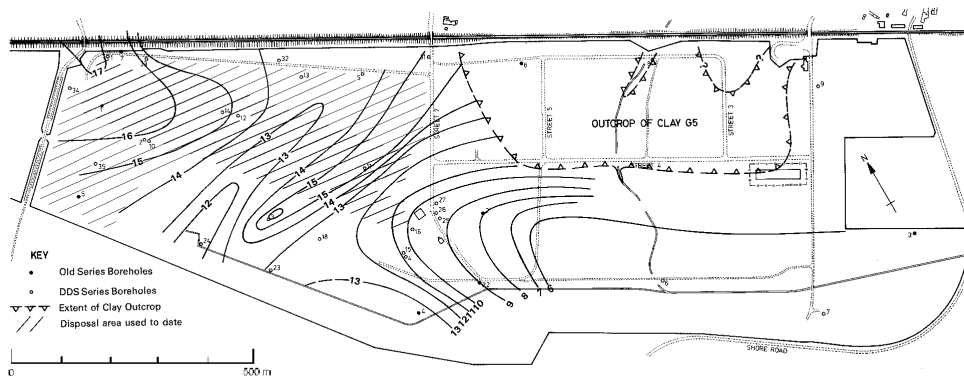
Contours in metres above Ordnance Datum.
Adapted from Williams et al. (1985).

Figure 5   Contours of surface clay horizon G5.

b)   *The St. Bees Sandstone*

The St. Bees Sandstone, does not outcrop in the immediate area of Drigg and has an irregular surface ranging from 10 m above OD in the north to 42·3 m below OD in the south (see Figure 6).
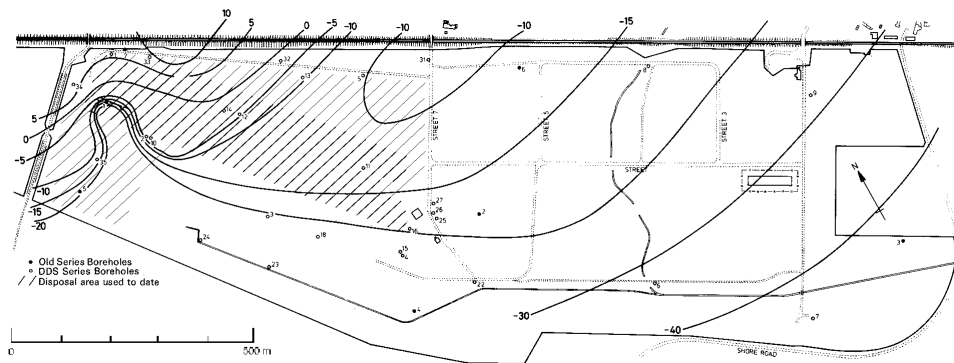
Contours in metres above Ordnance Datum.
Adapted from Williams et al. (1985).

Figure 6   Contours of the surface of the St. Bees Sandstone.

The St. Bees Sandstone consists of a dull red, medium grained, cross-bedded sandstone often containing white mica. It is considered to be a sub-aerial deposit (Trotter *et al.*, 1937). The sandstone is relatively well-cemented, with a calcareous cement, and has been used extensively as a building stone throughout West Cumbria. However, in places it can be poorly cemented and friable.

The Triassic sediments flank the Lower Palaeozoic rocks of the Lake District to the east. These increase rapidly in thickness towards the west in a series of step-faltered blocks. Beneath Drigg, the Trias is considered to be of the order of 1000 m thick (Gregory, 1915).

Information on faulting in the St. Bees Sandstone has been obtained mainly from the iron ore mining area to the north of Drigg and from data published by UK Nirex Ltd. The iron ore has been extracted from veins developed along faults in the Carboniferous Limestone Series, which often extend through into the Trias (Rose and Dunham, 1977). Where the St. Bees Sandstone is covered by thick glacial deposits outside the mined area, faults have not been detected, but it is likely that they occur with the same frequency (Trotter *et al*., 1937).

c)   Borrowdale Volcanic Group

A considerable amount of deep exploratory drilling has been undertaken by Nirex in the vicinity of Sellafield and the surrounding villages of Gosforth, Beckermet and Seascale in connection with a proposed rock characterisation facility and deep below ground intermediate level waste / low level waste repository[1]. Much of the data obtained have been summarised in publications intended for a non-scientific readership (e.g. Nirex, 1993).

Below the St. Bees Sandstone, and associated shale and evaporites layers, lie a non-continuous carboniferous limestone layer which overlies the Borrowdale Volcanic Group bedrock. The potential flow of groundwaters through this rock has been considered in some detail. However, for the purposes of characterising Drigg over time periods of hundreds of years it can be assumed that the Borrowdale Volcanic Group is impermeable and the overall migration of deep groundwaters is seaward (see Figure 7).

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[1] UK Nirex Ltd's application for the rock characterisation facility near to the Drigg site at Longlands Farm, Gosforth was rejected. The consequence of this will be that proposed deep level waste repository will not now be constructed.
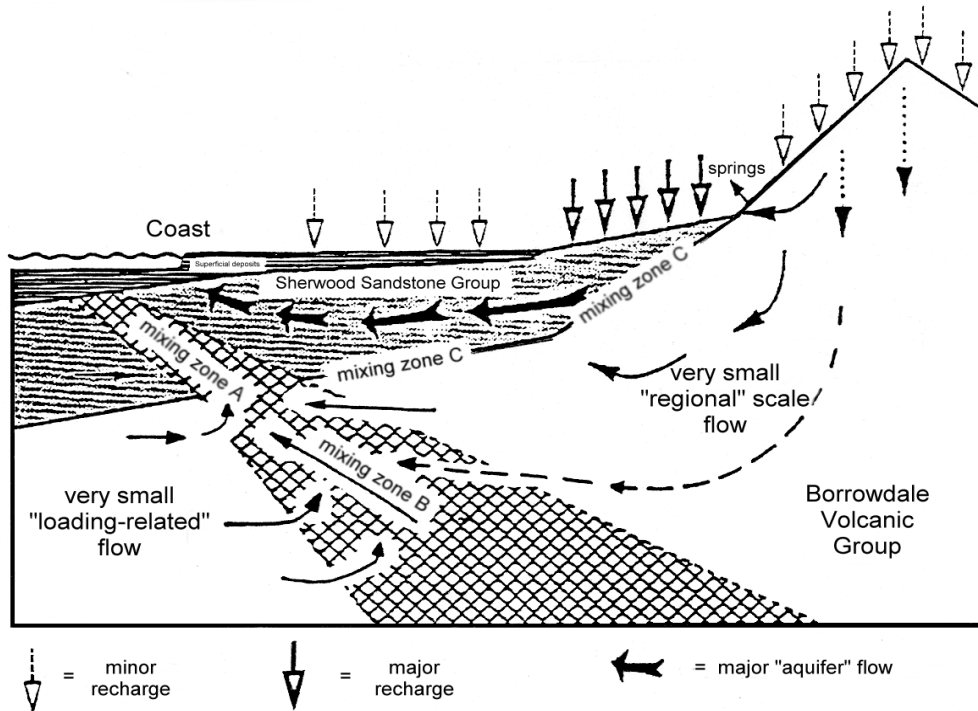
From Environmental Resources Limited (1992).

Figure 7   Schematic deep groundwater flow around Sellafield.

d)   Hydrogeology

i)   Regional and perched aquifers

The regional groundwater beneath the Drigg site lies within the sandstone bedrock and the overlying G2 sands and gravels. The G1 layer is absent in considerable areas so that the G2 aquifer has the potential to feed into the bedrock. The measurements of piezometric head in the regional aquifer confirm a consistent seaward migration of groundwater. It is thought that the regional groundwater emerges at sea near the Drigg beach to the south-west of the site. Regional groundwater contour levels are shown in Figure 8.
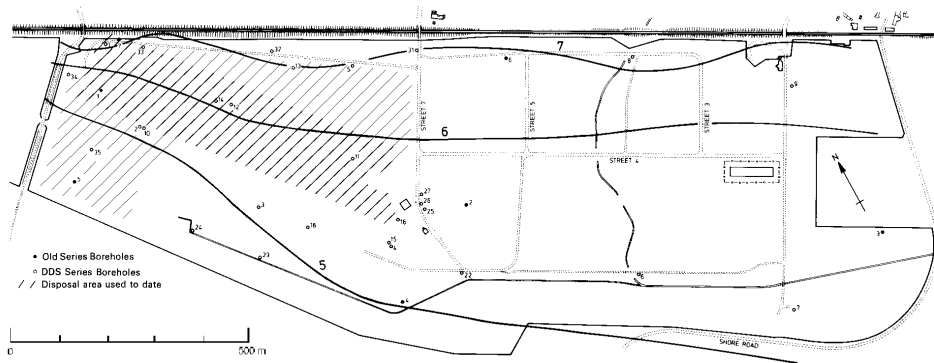
Contours in metres above Ordnance Datum.

Figure 8   Regional groundwater levels at the Drigg site.

The relatively impermeable layers above the G2 layer (i.e. particularly G3 and G5) result in perched aquifers in G4 and G6. Infiltrating water, from the surface, is unable to penetrate sufficiently rapidly through the G3 and G5 layers, and so tends to move horizontally according to the surface topography of the G3 and G5 layers. Various measurement made on the site indicate that water in the G6 aquifer in the area of the disposal trenches seeps, to some extent, through the G5 layer into the G4 layer (especially in some areas where the G5 layer is more permeable), while the rest emerges at the ground surface in the area of the East-West stream. Water in the G4 layer is thought to find its way downward, through the G3 layer, and then merge with water in the regional aquifer and emerge eventually on the Drigg beach. Areas of perched water are represented in Figure 9.
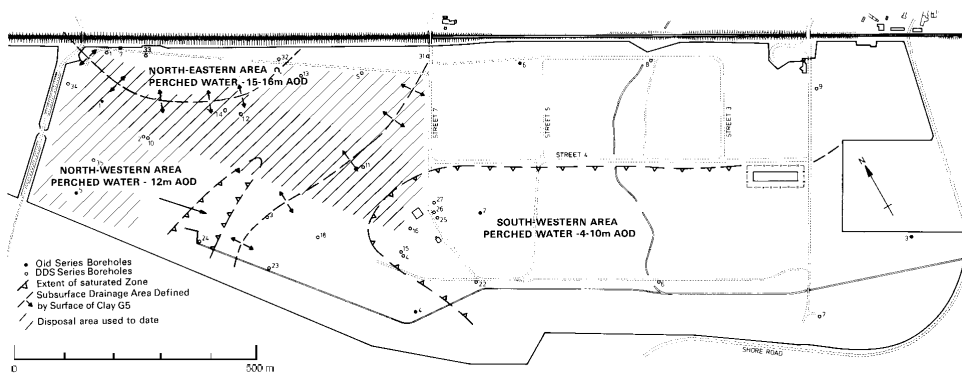
Figure 9   Areas of perched water tables at the Drigg site.

Smith *et al.* (1988) assumed that the G6 aquifer releases approximately 1×10^3 m^3 of water to the East-West stream/Drigg stream per year and that the G4 aquifer releases a further $2 \cdot 5 \times 10^5$ m^3 a^{-1}.

ii)   Groundwater characterisation

Of the metals present in groundwater, aluminium, iron and uranium, are believed to be at elevated levels compared with those of the surrounding area.   Peaks in phosphate, nitrate/nitrite and ammonium which occur may be due to the influence of agricultural application on adjacent farmland.

No experimentally derived, site-specific distribution coefficients are known to be available. Various reviews of data do tabulate generic soil/groundwater distribution coefficients, K_d. Values used are taken from Smith *et al.* (1988) and summarised in Table 2.

Table 2   Distribution coefficients for groundwater at the Drigg site.

Nuclide	Best estimate K_d (m^3 kg^{-1})	Range (m^3 kg^{-1})
Cs	1	0·03 to 50
Pu	2	0·01 to 300
Am	6	0·001 to 50

From Smith et al. (1988).

3.1.3   Pedology

Much of the area of the site has remained largely undisturbed during the last two decades. However, there has been significant, localised disruption, by heavy machinery etc., since 1939. This has resulted in a mixing of the overlying soils which are characterised by a high sand content (Rudge, 1989).

The soil types found on the site are consistent with the geology of the Drigg site. These are made up of a complex and irregular sequence of glacial deposits (made up of boulder clays and silts, coarse sands and gravels) interlaced with peat deposits (Murdock, 1992). Close to the stream, the dominant soil type is a Brown Podsol, this being replaced by Argillic Brown Earth away from the stream banks. The tree covered areas are primarily Stagnohumic Gleys, with severely impacted clay loam. As the result to the proximity of Drigg to the coast, it is exposed to a significantly salt-laden atmosphere.

3.1.4   Meteorology

Meteorological data has been collected at the automatic weather station on the Drigg site for the years 1991 to 1995. These have been used to calculate mean monthly and annual statistics for the temperature, the wind speed, the rainfall, the potential evaporation and irradiation statics for the Drigg site (see Table 3).

Table 3   **Meteorological data for Drigg trench cap (1991-1995).**

Month	Maximum Temperature °C	Minimum Temperature °C	Wind speed m s⁻¹	Irradiation MJ m⁻² day⁻¹	Rainfall mm month⁻¹	Total Potential Evaporation mm month⁻¹
January	6·5	2·2	4·85	1·8	89·6	14·7
February	6·2	2·0	4·28	3·7	81·8	15·4
March	7·3	2·7	4·57	6·7	89·3	34·0
April	10·0	4·3	3·72	11·6	67·0	55·7
May	13·7	6·0	3·45	16·3	47·2	83·0
June	16·3	9·1	3·03	18·0	55·9	81·3
July	17·4	11·3	3·22	15·8	85·9	90·9
August	17·8	11·0	3·28	14·1	76·4	78·9
September	15·4	8·7	3·39	9·1	72·2	50·5
October	12·2	5·9	3·86	4·9	106·0	34·3
November	9·1	4·1	4·06	2·4	72·2	15·3
December	6·4	1·9	4·17	1·4	107·7	9·5
Mean	11·5	5·8	3·82	8·8	79·3	47·0

Calculated from BNFL personal communication (1997a).

The differences between the annual distributions of rainfall and potential evaporation, shown in Table 3, suggest that infiltration is most likely to occur in the months August to April.

The data from the site also allowed a wind rose to be calculated for the site (see Figure 10). This shows that the predominant wind direction originates from the south.
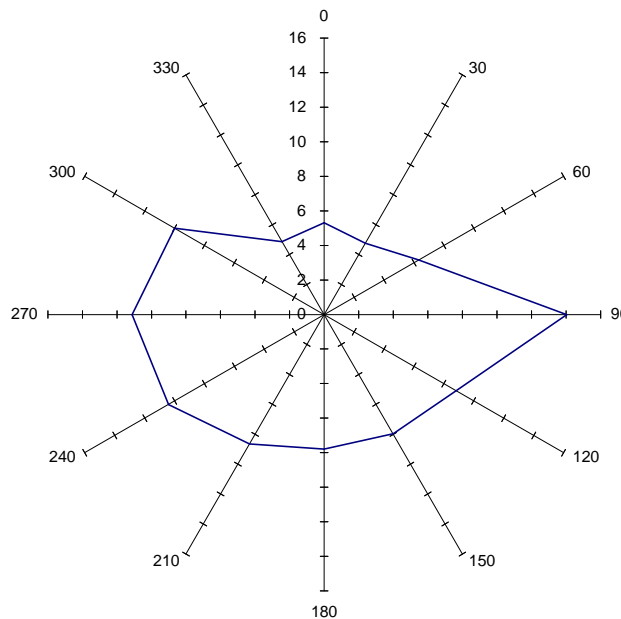
Note: The wind rose indicates the direction from which the wind originates.
Calculated from BNFL personal communication (1997a).

Figure 10   Wind rose for Drigg site.

3.1.5   Hydrology

a)   *Discharge courses and quantities from the Drigg site*

A number of surface water and discharge courses exist from the Drigg site. The two principal pathways are via the Drigg stream, which arises along the western boundary of the site, and the marine pipeline through which all collected leachates are drained.

Except for two streams which enter from the north-east via culverts under the railway embankment and immediately join, all other streams rise within or near the site boundary. Of the two streams entering from the north-east one is relatively polluted, probably by sewage effluent, and this often dominates the character of the downstream flow through the site. The internal site drainage is outlined in Figure 11.

A ditch evacuated along the northern half of the western boundary joins with the natural stream channel which eventually leaves the site at a point on the southern perimeter. Beyond the site fence the stream flows through about 300 m of agricultural land before entering, the tidal estuary of the River Irt. A minor catchment area of about 4 ha in the north-west corner of the site drainage channels westwards and spreads out at the shoreline some 400 m distant.

In addition to these natural drainage channels, surface water is intercepted by a series of drainage channels from the former Royal Ordnance Factory within the southern part of the site. The disposal trenches and newer vaults are deliberately drained to a marine outfall which discharges below the low water mark. Prior to 1991, collected leachates were drained to the Drigg stream.
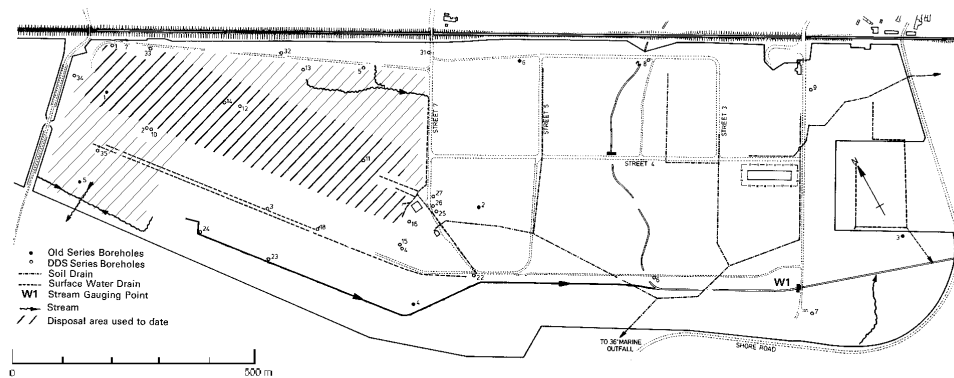
Adapted from Williams et al. (1985).

Figure 11   Site plan showing location of boreholes and drainage features.

a)   Drigg stream

The Drigg stream arises along the north-western edge of the Drigg site and flows out to the south and west (see Figure 12). Approximately 300 m after leaving the site, the Drigg stream joins the tidal regions of the River Irt. Indeed, the Drigg stream may be tidally influenced some way onto site under conditions of high spring tides, although the monitoring weir introduced some 200 m inside the site perimeter forms an effective limit of tidal influence.
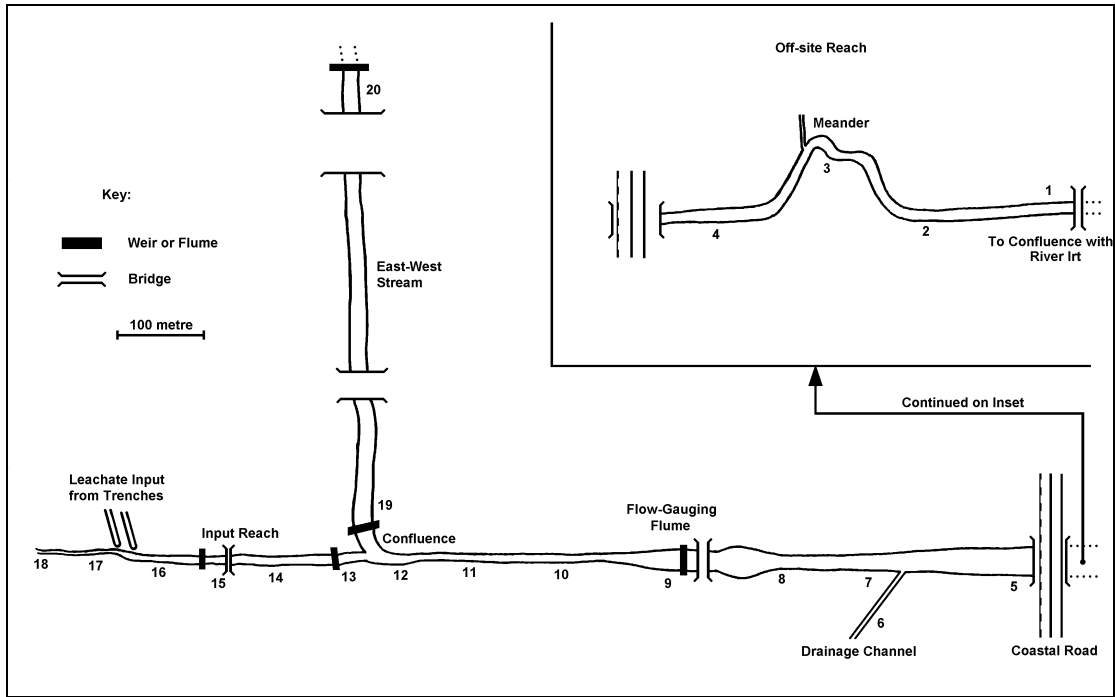
Sampling points described by Murdock (1992) are shown.

Figure 12   The Drigg stream.

i)   Flow rate

Smith *et al.* (1988) estimated the total length of the Drigg stream to be 1090 m, with a volume of 690 m³, a river water velocity of 1.5×10^6 m a^{-1} (0·05 m s^{-1}) and a bed sediment velocity of 150 m a^{-1} (5 10^{-6} m s^{-1}).

Murdock *et al.* (1995) published velocity data for individual locations on the Drigg stream for the period January 1990 to March 1991 (see Table 4).

Table 4   Velocity of stream at sampling points.

Sampling point	Velocity m s^{-1}
1	0·38
3	0·14
5	0·12
7	0·51
8	0·12
9	0·06
10	0·17
12	0·26
14	0·25

Note: position of sampling points correspond to those given in Figure 11.
From Murdock (1992)

The total annual flow of water from the Drigg site in the Drigg stream has been published by BNFL since 1988 (BNFL 1989-1997). These are given in Table 5.

Table 5   **Annual off-site flow through the Drigg stream.**

Year	Annual flow (m^3)
1988	$1\cdot4 \times 10^6$
1989	$1\cdot1 \times 10^6$
1990	$1\cdot1 \times 10^6$
1991	$9\cdot7 \times 10^5$
1992	$1\cdot4 \times 10^6$
1993	$8\cdot9 \times 10^5$
1994	$1\cdot5 \times 10^6$
1995	$1\cdot6 \times 10^6$
1996	$8\cdot5 \times 10^6$

From BNFL (1989-1997).

Within the broad range of flow rates recorded over nine years, the diversion of $0\cdot6$-$2\cdot1 \times 10^5$ m^3 (BNFL, 1991-1997) to the sea tanks, which began in 1991, is not apparently significant.

Murdock (1992) recorded instantaneous flow rates for the Drigg stream on ten sampling occasions. These varied between $0\cdot004$ and $0\cdot14$ m^3 s^{-1}, with an arithmetic mean of all values of $0\cdot0525$ m^3 s^{-1}. This approximates to an annual flow of $1\cdot7 \times 10^6$ m^3 over the sample period (range $1\cdot3 \times 10^5$ to $4\cdot4 \times 10^6$ m^3 a^{-1}), which is in reasonably close agreement with the total flow values given in Table . Smith *et al.* (1988) have also estimated the flow rate in the Drigg stream, and assumed a uniform value of $9\cdot5 \times 10^5$ m^3 a^{-1}, again in reasonable agreement.

ii)   Suspended solids and water characterisation

It may be assumed that the total suspended solids loading of about $0\cdot02$ kg m^{-3}, with a range from about $0\cdot002$ to $0\cdot2$ kg m^{-3}. Exceptionally, suspended solids may rise to 2 kg m^{-3} or more at points near to the confluence of the East-West stream and the Drigg stream, but such high concentrations are not thought to persist. Data published by Murdock *et al.* (1995) indicate a suspended sediment loading of $0\cdot006$ to $0\cdot12$ kg m^{-3} at the monitoring weir over the period 1990 to 1991, with a range of $0\cdot001$ to $0\cdot26$ kg m^{-3} inclusive of monitoring of the tidal reach and input reach of the Drigg stream over the same period. Rudge (1989) reported suspended solid concentrations for the period 1986 to 1987. These were $0\cdot0096$ to $0\cdot0122$ kg m^{-3}, for the East-West stream, $0\cdot0261$ to $0\cdot0640$ kg m^{-3} for the Drigg stream below the confluence and $0\cdot0190$ to $0\cdot0232$ kg m^{-3} for the water entering the River Irt.

The conductivity of Drigg stream water has also been measured by Murdock (1992). Little temporal variation was found over a year, although conductivities were generally highest in the input reach of the stream (up to 750 µS cm^{-1}) and lowest in the tidal reach (down to 400 µS cm^{-1}). This indicates that the salt content of the leachate to be slightly higher than that of the stream water.

iii)   Distribution Coefficients

Murdock *et al.* (1995) have published specific distribution coefficients for caesium, americium and plutonium with suspended sediment in the Drigg stream, covering the period over which leachates were diverted away from the stream for outfall via the marine pipeline. For comparison, generic data used by Smith *et al.* (1988) in assessing the Drigg site are also included in Table 6.

Table 6   **Distribution coefficients (K_d) for ^{137}Cs, ^{241}Am and ^{239,240}Pu for suspended sediment in the Drigg stream.**

Date		K_d (m^3 kg^{-1}) †								
		^{137}Cs			^{241}Am			^{239,240}Pu		
		Tidal reach	F-G station‡	Input reach	Tidal reach	F-G station‡	Input reach	Tidal reach	F-G station‡	Input reach
1990	avg.	43	29	16	11	65	77	23	9·5	25
	min.	8·2	9·8	1·5	13	2·0	1·1	1·8	2·2	0·20
	max.	110	91	35	480	140	300	100	33	110
1991	avg.	67	130	250	33	71	400	78	80	230
	min.	24	-	-	6·4	0·51	25	7·0	6·7	15
	max.	110	-	-	80	140	560	170	180	600
NRPB generic f/w values		30			1000			1000		

† *Where K_d is defined as*   :   <u>*Conc nuclide in suspended sediments (Bq kg^{-1} dry weight)*</u>
<div align="center">Conc nuclide in filtrate (Bq m^{-3})</div>

‡ *F-G; flow-gauge*

From Murdock et al. (1995) and Smith et al. (1988)

It can be seen that prior to 1991, the generic data for caesium are roughly in agreement with the experimentally derived values, but that the generic distribution coefficient for the actinides is in all cases substantially in excess of the experimental values. Following re-routing of the leachates, the apparent distribution coefficient values rose significantly. However, the generic values for plutonium and americium remained higher than the observed values. This apparent rise is particularly noticeable for caesium and is due to the very rapid fall in radionuclide concentrations in stream water immediately following leachate diversion to the marine pipeline. Generally, levels fell by around two orders of magnitude in the first two weeks (Murdock *et al.*, 1995). Since the distribution coefficient is effectively intended as an equilibrium value, it is clear that the disequilibrium caused by the slow desorption of nuclides from the sediment compared to flushing out of the dissolved phase in water must be taken into account.

b)   *River Irt and the Ravenglass estuary*

The Ravenglass estuary area, including the lower reaches of the River Irt, is characterised in Technical Deliverable TD12 (Bousher, 1998). Summary information only is presented below.

i)   *Flow rate*

Smith *et al.* (1988) define the Irt estuary as a water body of $2·5 \times 10^6$ m^3, with a mean depth of 3 m and a volumetric exchange of $9·1 \times 10^8$ m^3 a^{-1}.

ii)   *Suspended solids and water characterisation*

The Irt estuary is assumed to have a sediment loading of 4×10^{-5} kg m^{-3} and a sedimentation rate of $0·15$ kg m^{-3} a^{-1} (Smith *et al.*, 1988).

iii)   *Distribution coefficients*

A distribution coefficient of 28 m^3 kg^{-1} has been reported by Assinder *et al.* (1985) for the River Esk, based on a single sample. This value is in fairly close agreement with the experimentally derived values reported for the Drigg stream and may be reasonably

representative of freshwaters in this region in general. For coastal waters Smith *et al.* (1988) assume a distribution coefficient of $3 \, m^3 \, kg^{-1}$ for caesium, which may indeed be more applicable to brackish waters due to the competition for binding sites from sodium and potassium ions present in the marine water component.

3.1.6 Population (characteristics)

a) Demography

Drigg site is contained within the Parish of Drigg and Carlton, which extends northwards to divide the village of Seascale into two, the more populous part of Seascale being in the relatively recent Parish of Seascale (formed c.1909). As a consequence, population data, which are generally reported by Parish, are approximate for Seascale and the farms around Drigg (Copeland Borough Council, 1996).

The west coast of Cumbria around Sellafield and Drigg is sparsely populated. The total population of the local authority area (Copeland Borough Council) is about 71,000, the major part of which reside in the three local towns of Whitehaven (22 km north of Drigg), Egremont (13 km north of Drigg) and Cleator Moor (18 km north-east of Drigg).

Three villages lie close to the Drigg site. Seascale (3 km to the north of the site) has a population of about 1,000, Holmrook (2 km to the east) has a population of 150 and Drigg village (immediately adjacent to the south-east of the site) has a population of 325. A number of scattered farms and clusters of a few cottages are in the vicinity of Drigg, which may have a combined population of about 125.

The Drigg stream, which carries surface water from the Drigg site, crosses the fields of Greenmoorside Farm to the south of the site.

For risk assessment purposes, Smith *et al.* (1988) assume that the nearest habitation to the Drigg site is c.250 to 350 m distant. The major habitation (Drigg village) is taken to be 1 km distant.

b) Habits

Local habits data are well documented with respect to 'critical groups' for the radiological impact of discharges from the Sellafield and Drigg sites. Information directly relevant to Drigg is summarised in Table 7.

Table 7   Critical groups characterised for the Drigg site.

Exposure pathway	Critical group characteristics	Comments	Source of information
Terrestrial foodstuffs	Infants assumed to consume local milk at $0 \cdot 24 \, m^3 \, a^{-1}$.	Milk and inhalation are the only pathways considered for the Drigg critical group. Other characteristics can be given for the Sellafield critical group.	BNFL (1997), derived from local surveys and NRPB defined general consumption habits
Marine foodstuffs	No marine critical group of consumers is defined for Drigg.	The majority of activity recorded in local environmental surveys is attributed to Sellafield.	BNFL (1997)
Local occupancy	No external radiation critical group is defined.	Radiation measurements are made at a number of points near the site perimeter, but no local occupancy of these areas is explicitly assumed.	BNFL (1997)

On the basis of measured concentrations of radionuclides in local milk, the dose to the most exposed individual from Drigg (assumed to be an infant) was calculated by BNFL (1997) to be 5·7 µSv in 1996; however, this could well be dominated by the contribution from discharges of the nearby Sellafield site . Other pathways were insignificant.

Although the Drigg stream is not a source of potable water, Smith *et al.* (1988) assumed that up to 5×10^{-4} m³ a⁻¹ may be ingested accidentally, or roughly 0·1% of typical annual intake of drinking water.

Local external gamma dose rates are generally near to background (range 0·08 to 0·11 µSv h⁻¹ on sand dunes, beach and grass banks around the Drigg stream), except for an area of mud at the confluence of the Drigg stream and the River Irt, where dose rates measured 0.26 µSv h⁻¹ in 1996 (BNFL, 1997). No critical group for this pathway is defined.

Critical group characteristics for Sellafield are defined in Table 8. These groups may be applicable to Drigg for the purposes of calculating limiting doses which may be incurred, but it should be noted that most dose calculations in the vicinity of Drigg would, in practice, be dominated by Sellafield derived discharges.

Table 8   Critical group consumption and occupancy rates defined for Sellafield.

Foodstuff	Consumption rate (kg person⁻¹ a⁻¹)		
	Adults	**Children**	**Infants**
Cod & plaice[a]	34·75	Marine consumption pathways expressed on basis of adult consumption[i]	
Crab & lobster[b]	8·25		
Winkles	7·79		
Other molluscs[c]	2·16		
Milk	258·9	258·8	260[d]
Beef	29·3	14·7	7
Mutton	9·2	2·7	3
Sheep liver	1·6	0·9	2
Pig meat[h]	100	81·6	0
Green veg	22·2	14·4	30
Root veg[e]	48·5	30·2	50
Mushrooms[h]	0·8	0	0
Fruit[f]	11·0	5·6	1·2
Poultry	5·9	3·7	3
Eggs[g]	28·9	15·2	14·5
Honey[j]	6·72	na	
Venison[j]	134·4		
Other game[j]	17		

Occupancy pathways	Occupancy of area (h a⁻¹)		
	Adult	**Children**	**Infants**
Whitehaven boat dweller	610	na	
Site perimeter	full occupancy		
Beach/leisure	1000		
Farmer (ploughing)	300		

a. A mix of 50% cod and 50% plaice is assumed.
b. A mix of 70% crab and 30% lobster is assumed.
c. Assumed to be an equal mix of cockles, mussels and limpet.
d. Based on generalised Heywood (1987) recommendations
e. Local potatoes only
f. Soft fruits only
g. Hens eggs only
h. From Stewart et al. (1990), data for infants relate to 1-2 years.
i. The critical group may include children 'scaled' to adults.
j   From Fulker et al. (1998); maximum consumption rates reported

From BNFL (1995), except where otherwise stated

It should also be noted that critical group consumption rates are not simply additive. In fact, relatively little crossover between pathways is thought to occur. For dose calculation

purposes it is usual to assume that marine pathway groups consume all seafoods at critical group rates, but these are not the same group as the Whitehaven boat dwellers. Critical group consumers of food do not generally eat elevated levels of fish etc. Caution should thus be exercised in calculating reasonable upper dose limits based on the critical group consumption rates given above.

Habits data applicable to the general population around Drigg can probably be modelled from UK national consumption data (e.g. Byrom *et al.*, 1995). A summary of mean consumption data for UK adults are presented in Table 9.

Table 9   Mean consumption of food by adults (16 to 64 years) in the UK.

Food group	Percentage of population as consumers	Mean consumption of consuming group (kg person^{-1} a^{-1})	97.5th percentile consumption of consuming group (kg person^{-1} a^{-1})
Domestic fruit	76·3	20	50
Potatoes and root vegetables	100	50	95
Green and other domestic vegetables	96·6	15	35
Mushrooms	17·1	1·5	4·5
Honey	3·9	2	7·5
Pig meat	94·6	8·5	25
Cattle meat	96·9	15	30
Sheep meat	41·0	4	10
Offal	36·6	3	10
Poultry	68·0	5·5	15
Game	0·3	4	7·5[†]
Milk	99·9	110	240
Butter, cheese & other milk products	98·3	15	45
Eggs	96·3	6·5	20
Fish	69·3	6	20
Shellfish	3·2	2·5	7

† Based on a very small sample size (less than 40 consumers from the national survey).
From Byrom et al. (1995).

Data are specifically omitted from the above table which relate to imported foodstuffs. Cereal consumption is also omitted as cereal crops are not grown for human consumption in this area (see Section 3.1.6.c).

c)   Activities

The land to the south and west affords poor grazing to sheep only. To the north and east the land is more fertile, with mainly dairy cattle. Beef cattle and over-wintering sheep may also be farmed in this area. A few root crops are grown for human consumption (mainly potatoes), although the majority of such crops (e.g. beets) are used as over-wintering fodder for animals. Silage is also produced for over-wintering, both as bale and clamp silage. Very limited cereal crops are grown anywhere near Drigg and such cereals are intended as animal fodder (e.g. some maize is grown approximately 8-10 km north-east of the site at Gosforth).

For modelling purposes, the main potential source of contamination arising from the Drigg site entering the food chain is via the Drigg stream which flows off-site to join the River Irt, and in doing so traverses land grazed by dairy cattle.

There are no clear data to suggest that surface waters arising from the Drigg site are used for local agricultural land irrigation. Nonetheless, for the purposes of risk assessment, Smith *et al.* (1988) assume that up to 0·5 km^2 of land may be so irrigated, utilising a fraction of 0·04

of the water flow. A further fractional bottom sediment transfer rate of $0 \cdot 05 \, a^{-1}$ to land is assumed. These assumptions would appear to be upper limits.

Very limited commercial fishing is undertaken off the coast at Drigg. One or more part time fishermen maintain boats at Ravenglass, but the main fishing ports are at Whitehaven, Workington and the Isle of Man. Mussel beds at Ravenglass have at various times been exploited commercially. Collection of shellfish (mussels, winkles, limpets, etc.) off the Cumbrian coast for personal consumption remains relatively common. Scallop beds to the south and east of the Isle of Man are unlikely to receive any measurable input of radioactivity from the Drigg site, even in the absence of the influence of Sellafield discharges.

Commercial exploitation of the edible seaweed *Porphyra umbilicalis* ceased during the 1960's and no collection for personal consumption is known to occur. Seaweed derived fertilisers are not known to be utilised by local farmers, but may be used for local domestic produce.

3.2   Physico-chemical characteristics

3.2.1   The water column

Water samples from the Drigg stream and the drainage channel were analysed by Westlakes Scientific Consulting and FZ Rossendorf (see Appendix A). Typical compositions of the stream and drainage channels are summarised in Table 10 along with their uncertainties (standard deviations) associated with these measurements.

Table 10   Typical aqueous phase compositions for the Drigg stream and drainage channels.

Component	Compartment			
	Drainage channel		Drigg stream	
	Mean	Stand. Dev.	Mean	Stand. Dev.
F^-	-	-	$4 \cdot 545 \times 10^{-7}$	-
PO_4^{3-}	$2 \cdot 001 \times 10^{-6}$	$4 \cdot 212 \times 10^{-7}$	$1 \cdot 259 \times 10^{-6}$	$1 \cdot 346 \times 10^{-6}$
NO_3^-	$7 \cdot 096 \times 10^{-5}$	$5 \cdot 483 \times 10^{-5}$	$4 \cdot 720 \times 10^{-4}$	$2 \cdot 675 \times 10^{-4}$
NO_2^-	$3 \cdot 695 \times 10^{-7}$	$1 \cdot 087 \times 10^{-7}$	$3 \cdot 036 \times 10^{-6}$	$2 \cdot 771 \times 10^{-6}$
NH_3	$2 \cdot 877 \times 10^{-5}$	$2 \cdot 349 \times 10^{-6}$	$1 \cdot 687 \times 10^{-5}$	$1 \cdot 486 \times 10^{-5}$
SO_4^{2-}	$5 \cdot 674 \times 10^{-4}$	$3 \cdot 123 \times 10^{-5}$	$4 \cdot 015 \times 10^{-4}$	$4 \cdot 469 \times 10^{-5}$
CO_3^{2-}	$2 \cdot 500 \times 10^{-3}$	$3 \cdot 333 \times 10^{-5}$	$1 \cdot 618 \times 10^{-3}$	$1 \cdot 481 \times 10^{-4}$
Cl^-	$2 \cdot 398 \times 10^{-3}$	$1 \cdot 692 \times 10^{-4}$	$1 \cdot 629 \times 10^{-3}$	$7 \cdot 184 \times 10^{-5}$
SiO_2	$1 \cdot 714 \times 10^{-4}$	$3 \cdot 329 \times 10^{-6}$	$1 \cdot 262 \times 10^{-4}$	$1 \cdot 326 \times 10^{-5}$
K^+	$1 \cdot 228 \times 10^{-4}$	$2 \cdot 558 \times 10^{-6}$	$2 \cdot 107 \times 10^{-4}$	$3 \cdot 876 \times 10^{-5}$
Na^+	$2 \cdot 358 \times 10^{-3}$	$1 \cdot 740 \times 10^{-5}$	$1 \cdot 435 \times 10^{-3}$	$1 \cdot 714 \times 10^{-4}$
Ca^{2+}	$1 \cdot 854 \times 10^{-3}$	$1 \cdot 372 \times 10^{-4}$	$1 \cdot 302 \times 10^{-3}$	$1 \cdot 975 \times 10^{-4}$
Mg^{2+}	$5 \cdot 513 \times 10^{-4}$	$1 \cdot 234 \times 10^{-5}$	$3 \cdot 058 \times 10^{-4}$	$6 \cdot 895 \times 10^{-5}$
Fe	$4 \cdot 118 \times 10^{-5}$	$7 \cdot 162 \times 10^{-6}$	$1 \cdot 561 \times 10^{-5}$	$2 \cdot 631 \times 10^{-5}$
Al^{3+}	$1 \cdot 731 \times 10^{-4}$	$3 \cdot 706 \times 10^{-7}$	$1 \cdot 931 \times 10^{-6}$	$2 \cdot 721 \times 10^{-6}$
Zn^{2+}	-	-	$3 \cdot 370 \times 10^{-7}$	$4 \cdot 402 \times 10^{-7}$
U	-	-	$8 \cdot 132 \times 10^{-8}$	$1 \cdot 354 \times 10^{-7}$
Th	-	-	$8 \cdot 862 \times 10^{-8}$	$1 \cdot 150 \times 10^{-7}$
Pb^{2+}	-	-	$3 \cdot 517 \times 10^{-8}$	$6 \cdot 008 \times 10^{-8}$
Ni^{2+}	-	-	$3 \cdot 087 \times 10^{-7}$	$3 \cdot 409 \times 10^{-7}$
Mn^{2+}	-	-	$1 \cdot 000 \times 10^{-7}$	$1 \cdot 732 \times 10^{-7}$
Cd^{2+}	-	-	$4 \cdot 326 \times 10^{-8}$	$7 \cdot 339 \times 10^{-8}$
As	-	-	$7 \cdot 071 \times 10^{-8}$	$7 \cdot 575 \times 10^{-8}$
pH:	$6 \cdot 45$	-	$6 \cdot 80$	$0 \cdot 53$
Eh / mV:	$59 \cdot 7$	-	$96 \cdot 00$	$14 \cdot 03$

In general the aqueous phases both of the drainage channel and of the Drigg stream are dominated by chlorides and (bi)carbonates of sodium and calcium. The iron and aluminium were mostly present in colloidal form, making exact analysis difficult.

The values for the redox potential imply reducing conditions, which is normally not to be expected from water with free contact to atmosphere. A reason for this may be the leaching and subsequently degradation of organic matter from inside the trenches (see Section 3.3.1.c). However, the redox potential given would be surprisingly low. Computation of E_h values from the concentrations for the redox pairs NO_3^-/NH_4^+ and NO_2^-/NH_4^+ gave mean redox potentials of 424 mV and 394 mV, respectively, suggesting a less reducing environment.

The ionic strength is around $0 \cdot 0099$ for the drainage channel and $0 \cdot 0067$ for the Drigg stream (diluted with water from the East-West stream). The pH is very close to neutral conditions. The temperature for the two waters was found to be $7 \cdot 0°C$.

3.2.2   Solid phases

The characterisation of the relevant solid phases is not as satisfactory as compared to the aqueous phase. There are only rather general mineralogical investigations for this site

published. General conclusions to be drawn from such information, partly supported by modelling results, indicate, that the sediments in contact with the aqueous phase are mostly mixtures of quartz and weathered alumosilicates, like illite, muscovite or kaolinite. The high colloidal portion of iron also points at the presence of iron oxides / hydroxides.

3.3 Radiological characteristics

All disposals up to 1988 were into shallow trenches, about 5 to 8 m deep, 25 m wide and up to 700 m in length. Trench 1 was a former railway cutting, but subsequent trenches were purposely excavated. The bases of the trenches are founded mostly in the G5 low permeability stratum and are sloped to drain from north to south. In Trenches 6 and 7 a central drainage channels has been constructed running north to south and incorporating a perforated plastic pipe with a gravel surround.

Disposals began in 1959. Earth firebreaks were introduced from November 1967, at which time about one-third of Trench 2 had been used. These firebreaks were constructed about 50 m apart, were of about 5 m mean length and had drainage pipes installed through their bases. Similar drainage arrangements exist at the end of the trench, the trench drainage systems then connecting to a common drainage system. Following revised operating aspects and reviewed safety studies, the firebreaks were discontinued in May 1983; after the first one had been constructed in Trench 6.

Groundwater can enter the volume occupied by waste by infiltrating through the cover material and, in addition, water can enter the trenches laterally, particularly via the more permeable layers such as G6. Much of the water percolates down through the waste, collects at the G5 base and runs down the slope and into the drainage system. However, this system is not 100% effective, for two main reasons. Firstly, not all the trenches are based entirely in G5 clay. In these areas leachate can run readily through the base of the trench directly into the G6 aquifer. This is mitigated in the later trenches, where clay was recorded as being absent, with the introduction of a bentonite layer (which has the same effect as the clay). Secondly, for a number of possible reasons the central drainage channels appear to be at least partly blocked in some trench sections. The main evidence for this comes from measurements of the water level in the trenches. Instead of water running away rapidly through the drainage channels the level has built up in parts of some trenches. In these areas there is therefore another direct route for leachate loss, laterally this time, into the G6 aquifer. The blockages may be due to the rather simple construction of the initial drainage channels or to failure of the drainage channels put into the firebreaks. The improved drainage channels adopted laterally and the presence of just a single firebreak in Trench 6 probably explain the absence of a build-up of the water level in that trench. Similarly, there has been no build up of water in Trench 7.

The compacted wastes in the trenches have been covered with a domed interim cap of maximum thickness 1·5 m at its periphery. The cap surface has a slope of approximately 1 in 25 and is a good deal thicker (maximum 6 m) at its centreline. The capping procedure required $440,000 \text{ m}^3$ of earth (for Trenches 1-6). This material, whilst not strictly representing engineered clay cap, is principally boulder clay from the site. The upper 20 cm of the cap profile, however, has a loam-like texture, due to the input of organic material from surface vegetation (Moores, 1992).

A plastic membrane has been laid approximately one metre below the trench cap surface to compliment the natural low permeability of the clay soil. The cap has been planted with a mixture of grass and shrubs both to increase evapotranspiration and for aesthetic considerations.

All trenches are now completely filled and have been covered with an impermeable membrane, capped with soil and planted. This has had the effect of dramatically reducing

water percolation into the trenches and, as a consequence, the volumes of leachate arising. Other measures have been introduced to the north-east of the site (e.g. constructing a curtain wall) to reduce off-site horizontal migration of waters from the trenches.

The hydraulic conductivity of the capping soil was found to be $2\cdot3$ - $2\cdot5 \times 10^{-6}$ m s^{-1} (Moores, 1992). These are well above the US EPA standard of 1×10^{-9} m s^{-1}. Therefore, the performance of the trench cap depends on the integrity of the plastic membrane (Moores, 1992).

The plastic membrane is essentially impermeable. However, a cap efficiency of 85-90% is assumed to be the case.

From 1988 waste has been placed in a concrete lined disposal area, in iso-freight containers (although Trench 7 continued to operate until 1995 for Sellafield wastes). Waste in Vault 8 is compacted and grouted within containers. When full, it will be capped with an impermeable membrane.

3.3.1 Source term

a) *Distribution in the Trenches*

The speciation of these radionuclides in the trenches has not be defined. However, given that considerable amounts of organic materials (e.g. cellulose) has been deposited in the trenches, it is expected that the conditions will be strongly anaerobic due to bacterial action. Therefore, these radionuclides will be predominantly present in low oxidation states.

The wastes disposed of at Drigg consist of biodegradable materials such as paper and cloth, as well as redundant plant items and steelwork, soil, rubble and some plastics and rubber. Volumes of waste accepted at Drigg over the past few years are presented in Table 11. The majority of wastes were consigned from Sellafield.

Table 11   Volumes of waste disposals at Drigg.

Source of Waste	Volume consigned for disposal (m³)					
	1989	**1990**	**1991**	**1992**	**1993**	**1994**
BNFL: Sellafield	24843	18835	17095	17902	15559	17435
Springfields	272	477	153	95	18	0
Chapelcross	76	436	112	76	200	235
Capenhurst	522	558	756	450	285	380
Westlakes (G. S. Labs)	0	0	0	0	0	22
AEA Technology:						
Harwell	0	0	536	1094	727	995
Winfrith	1351	354	144	270	480	438
Risley	37	173	20	0	0	0
Windscale	1126	1439	1081	918	732	383
MOD AWE sites	2354	2789	1237	1607	2174	2280
Other MOD sites	540	511	291	396	393	755
Scottish Nuclear:						
Hunterston	413	315	429	376	316	316
Torness	0	114	0	76	0	114
Nuclear Electric:						
Dungeness	228	511	284	115	36	36
Wylfa	38	77	77	171	38	0
Bradwell	192	340	315	169	90	131
Berkeley Power Stn	320	153	179	239	198	380
Berkeley Laboratories	189	430	133	171	113	146
Trawsfynydd	0	264	192	189	36	72
Hinkley Point	92	210	322	190	0	36
Oldbury	153	133	189	18	54	18
Sizewell	110	182	36	38	18	56
Heysham	115	38	77	38	95	76
Hartlepool	77	115	77	295	18	0
Wythenshaw	0	0	0	0	1	0
HSD	0	0	0	0	126	180
Amersham International plc:						
Amersham & Cardiff	1151	1147	1095	466	1044	1320
Miscellaneous[a]	316	2864	222	86	434	547
Total[b]	32400	32465	25052	25445	23185	26351[c]

a.   Including non-nuclear industries and also (for 1991) disposal of vent equipment from the BNFL ship "Mediterranean Shearwater".

b.   Disposals to Vault 8 are based on standard gross container volumes.

c.   Disposals have increased compared with recent years for both Sellafield and non-Sellafield sites. In the former case this is due to the introduction of extensive new plants, including Thorp, and an increasing level of work on decommissioning of old plants; excluding these changes, there is a continuing downward trend in routine arisings due to the continuation of waste minimisation initiatives. The introduction of the Waste Monitoring and Compaction (WAMAC) plant at Sellafield for treatment of LLW, and the end of direct disposals into trenches, means that the basis on which volumes are reported will change for 1995.

From BNFL (1990-1995).

All wastes are characterised for radionuclide content, external gamma dose rate and for contents of a "difficult" or "hazardous" nature[2]. As a result of waste minimisation measures, and with the advent of waste compaction, the volume of waste disposals to Drigg has fallen over the past five to ten years, from a peak of nearly 40,000 m³ in 1987 to around a present day 10,000 m³. Previously, wastes were packaged in plastic, paper sacks or steel drums and tipped into the trenches. With the advent of Vault 8, all wastes are now containerised.

As noted in Footnote a to Table 11, heavier and bulky waste materials from decommissioning nuclear facilities and old industrial sites have also, on occasion, been disposed of at Drigg.

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[2] The term "difficult" has no statutory definition. It is used to cover a range of industrial wastes, the disposal of which calls for special handling and operational procedures because of their toxic nature or physical characteristics.

The annual disposal inventories for the major nuclides of interest are presented in Table 12. Notwithstanding an increasing level of work decommissioning old plants, a continuing downward trend in waste volumes is expected.

Table 12   Radionuclide inventories of waste disposals at Drigg.

Nuclide	Activity disposed (TBq)					Authorised limit
	1992	**1993**	**1994**	**1995**	**1996**	**(TBq)**
Uranium	0·057	0·058	0·061	0·024	0·10	0·3
Radium-226 + Thorium 232	0·007	0·003	0·010	0·005	0·005	0·03
Other alpha[a]	0·24	0·11	0·14	0·09	0·10	0·3
Carbon-14	0·009	0·008	0·015	0·010	0·005	0·05
Iodine-129	0·0001	0·0001	0·0001	0·0018	0·00017	0·05
Tritium	0·17	0·18	0·22	0·34	0·41	10
Others[b]	4·1	4·4	7·0	6·0	3·6	15
Cobalt-60[c]	0·43	0·72	0·59	0·36	0·25	2

a.   *Alpha emitting radionuclides with half-lives greater than three months excluding uranium, radium-226 and thorium-232.*
b.   *Defined in the current authorisations as:*
   i. iron-55 and beta emitting radionuclides with half-lives greater than three months (excluding carbon-14, iodine-129 and tritium);
   ii. not more than 2 TBq may be cobalt-60.
c.   *The cobalt-60 figure is included in the "others" as well as shown separately.*
From BNFL (1995 - 1997).

An estimate of the total inventories of caesium, plutonium, americium and uranium is described in Appendix B. These are summarised in Table 13. A comparison between the calculated inventories with the range suggested by BNFL shows good agreement but indicates that there is a considerable degree of uncertainty associated with these values (particularly for ^{137}Cs) (Murdock, 1992).

Table 13   Estimated inventory of radionuclides in Trenches 1-7.

Nuclide	Solid Waste in Trenches	
	Inventory **TBq**	**BNFL estimate** **TBq**
Caesium-137	2·4	0·2 - 20
Uranium-234,235,238	41·3	--
Plutonium-239,240	1·6	0·8 - 4
Americium-241	0·7	0·03 - 1

3.3.2   Discharges/effluents

a)   Discharge courses and quantities from the Drigg site

Concentrations of radionuclides in the Drigg stream (water and bottom sediment fractions) are presented in Table 14 and historical trends in concentration for principal contaminants are summarised in Figure 13. Activity flowing off-site in the Drigg stream flows through about 300 m of agricultural land and enters the River Irt, The concentrations of radionuclides in the River Irt are also summarised in Table 14.

Table 14   Radioactivity in the Drigg stream and River Irt.

	Drigg stream site boundary: water (kBq m^{-3})							
Year	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**
Total alpha	<1	4	0·73	<0·64	<0·16	<0·14	<0·07	<0·09
Total beta	58	34	5	<3·4	<2·9	<0·98	1·2	3·8
^{3}H	26000	5000	540	243	256	210	210	250
^{90}Sr	37	18	2·5	<2·2	0·75	0·62	<0·56	<0·63
^{137}Cs	4·2	1·9	<0·64	<0·72	<0·19	<0·17	<0·23	<0·22
U alpha	1·3	0·9	0·52	<0·29	<0·25	<0·066	0·054	0·075
Pu alpha	0·07	0·09	0·032	<0·014	<0·011	<0·0051	<0·005	<0·006
^{241}Am	0·21	0·46	0·15	0·042	0·036	0·017	0·012	0·011
	Drigg stream bottom sediment (Bq kg^{-1} d.w.)							
Year	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**
Total alpha		1700					na	
Total beta		5200						
^{3}H								
^{90}Sr			<250	196	<150			
^{137}Cs		3400	1578	151	686			
U alpha			121	99	70			
Pu alpha			174	96	91			
^{241}Am			91	113	67			
	River Irt (kBq m^{-3})							
Year	**1989**	**1990**	**1991**	**1992**	**1993**	**1994**	**1995**	**1996**
Total alpha	<0·5	<0·52	<0·49	<0·41	<0·25			
Total beta	<1·6	<2	<0·49	<0·36	<0·13			
^{3}H	<830	<430	<7·8	11	<15	<5·1	<5·2	<0·50
^{90}Sr	0·86	0·76	0·046	0·05	<0·015	0·024	0·022	0·023
^{137}Cs	<1·3	<0·92	<0·44	<0·14	<0·045	<0·022	<0·019	<0·02
U alpha			<0·0037	0·0029	0·0013	<0·0014	0·0026	<0·003
Pu alpha	<0·005	<0·003	<0·0088	<0·0029	<0·0018	<0·002	<0·0021	0·003
^{241}Am			<0·0095	<0·0067	<0·0025	<0·0038	<0·0040	<0·003

na not analysed
From BNFL (1990-1995).
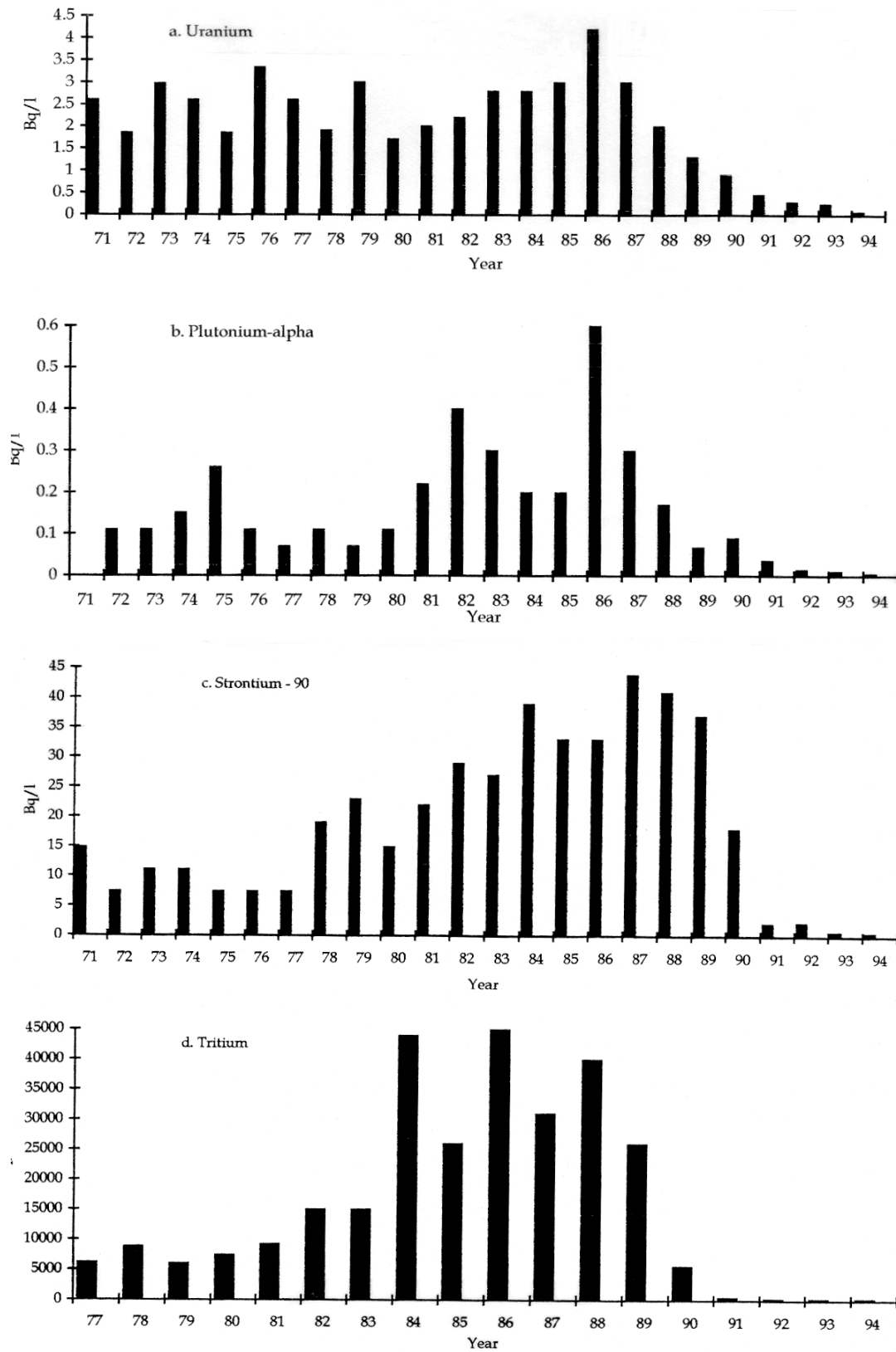
From BNFL (1995).

Figure 13   Radioactivity in the Drigg stream.

Data relating to sediment sampling are clearly incomplete and, therefore, should be treated as uncertain. For 1996, BNFL stated that levels of ^{137}Cs and the actinides in sediments of the Drigg stream bed are "now typically a few hundred Bq kg^{-1} with actinide concentrations averaging 100 Bq kg^{-1}" (BNFL, 1997). The Drigg stream is not a source of human drinking water and doses which may be incurred from accidental ingestion are considered to be negligible.

Seasonal surface water seepage points are also known to occur. These are often dry during the summer months and are minor sources of potential contamination to local areas. Two such seepage points are monitored periodically, when possible, to the north and north-west of the site. Annual flow estimates are not available. Reported concentrations of radionuclides in these minor watercourses are presented in Table 15.

Table 15   Radioactivity in minor water courses at Drigg: 1996.

Nuclide	Mean activity (kBq m^{-3})		
	British Rail culvert	**North site seepage point**	**North-west site seepage point**
Total-alpha	<0·14	<0·05	0·04
Total-beta	9·3	1·7	0·67
^{3}H	360	<110	<110

From BNFL (1997).

Periodic samples are also taken of water which has passed through the local water company's sewage treatment plant at Drigg village. Results for 1996, as in previous years, confirm that levels of radioactivity are generally below the limits of detection.

For completeness, quantities and volumes of radionuclides discharged via the marine pipeline are presented in Table 16.

Table 16   Radioactivity in marine pipeline discharges.

Nuclide	Quantity discharged (TBq)				
	1992	**1993**	**1994**	**1995**	**1996**
Total-alpha	0·00078	0·00067	0·0012	0·00085	0·00027
Total-beta	0·028	0·021	0·025	0·034	0·009
^{3}H	3·0	2·5	2·3	2·7	2·0

From BNFL (1995-1997).

b)   Distribution in streams

The addition of the trench caps and instigation of disposal of the trench leachates via a marine out-fall means that the current situation is not a true reflection of the discharges from Trenches 1-7 before 1991. However, Murdock (1992) provided activity data for the Drigg stream which covered the period before and after the installation of the marine out-fall. This data enables the concentrations and inventories of radionuclides in the leachate from the trenches, prior to capping and use of the marine out-fall, to be estimated. These calculations are described in Appendix B. A summary of the results are given in Table 17.

Table 17   Estimated radionuclide concentrations and inventories of leachate from Trenches 1 to 7.

Nuclide	Concentration kBq m^{-3}	Amount MBq a^{-1}
Caesium-137	48·6	267
Uranium-234,235,238	28·1	93
Plutonium-239,240	8·6	47
Americium-241	6·5	36

The distribution of particulate and dissolved radionuclides at different points on the stream have been measured (see Table 18). The data suggests that both particulate and dissolved phases are important.

Table 18   Stream water activity data.

Site	n	^{137}Cs (kBq m^{-3})		^{239,240}Pu (kBq m^{-3})		^{241}Am (kBq m^{-3})	
		Particulate	Dissolved	Particulate	Dissolved	Particulate	Dissolved
1	4	0·15	0·11	0·012	0·033	0·003	0·014
9	4	0·12	0·08	0·010	0·027	0·005	0·027
15	4	0·16	0·08	0·011	0·028	0·007	0·049

From Murdock (1992)

The sources of these radionuclides are likely to be re-equilibrium with the sediment in the stream bed.

3.3.3   Contamination

a)   Sediment

The re-routing of the leachate from the trenches through the marine out-fall is likely to indirectly affect the distribution of radionuclides in the sediment. If the concentration in the stream water declines then it is expected that the concentration in the sediment will also decline as radionuclides are released.

The caesium and gross alpha activity for different depths of stream sediment were measured during 1991 when discharge through the marine pipeline was first introduced (Murdock, 1992). The mean results for the year are given in Table 19. Over the year there was a gradual decline in the concentration in the top 2 cm (particularly at Site 15). However, the lower sediment concentrations appeared largely unaffected by this change. Therefore, this suggests that, unless the sediment is significantly disturbed, the re-equilibrium of the sediment with the stream water will be slow.

Table 19   Variation of activity with depth of sediment.

Site	Radionuclide		n	Mid point Depth range (cm)				
				2	6	10	14	18
7	^{137}Cs	kBq kg^{-1}	5	0·54	0·97	0·83	0·74	0·51
	α	kBq kg^{-1}	5	0·26	0·72	0·89	0·89	0·77
15	^{137}Cs	kBq kg^{-1}	5	0·56	0·68	0·76	1·72	1·05
	α	kBq kg^{-1}	5	0·51	0·61	1·01	2·00	1·35

From Murdock (1992)

It is assumed that this situation will also be the case for plutonium and americium.

The activity of radionuclides associated with different size fractions of the sediment has been investigated (see Table 20). The results suggest that much of the caesium activity is

associated with the clay fraction of the sediment. However, plutonium and americium were largely associated with both the clay and the sand fractions of the sediment.

Table 20   Distribution of radionuclides with size fractions.

Site	Sediment Fraction	n	^{137}Cs kBq kg^{-1}	^{239,240}Pu kBq kg^{-1}	^{241}Am kBq kg^{-1}
5	Clay (3-6%)	2	38·3 - 59·1	5·21 - 9·03	2·97 - 6·99
	Silt (4-7%)	2	11·2 -- 24·9	0·48 - 1·35	0·36 - 0·51
	Sand (87-93%)	2	1·2 - 1·7	0·1	0·06 - 0·07
	Total sediment		3·4 - 10·7	0·07 - 0·21	0·04 - 0·21
14	Clay (1-2%)	2	125·5 - 347·7	2·02 - 2·83	0·75 - 1·02
	Silt (2%)	2	40·1 - 67·2	0·71 - 2·46	0·41 - 0·73
	Sand (96-97%)	2	1·4 - 2·5	0·04 - 0·27	0·02 - 0·19
	Total sediment		2·7 - 6·8	0·38 - 0·49	0·27 - 0·29

Based on data taken from Murdock (1992)

The activity of each radionuclide in the total sediment is also included in Table 20.

The geochemical speciation of nuclides bound to Drigg stream bottom sediment has been determined by Murdock *et al.* (1993), based on sequential extraction, and is summarised in Table 21.

Table 21   Sequential fractionation of sediments.

Site	Fraction	n	^{137}Cs %	^{239,240}Pu %	^{241}Am %
1	Exchangeable	2	1·1	0·1	0·4
	Specifically adsorbed	2	0·6	0·4	4·3
	Organically bound	2	8·4	47·3	51·1
	Oxide bound	2	5·0	46·3	42·0
	Residual	2	85·0	6·1	2·4
9	Exchangeable	2	0·6	0·1	0·2
	Specifically adsorbed	2	0·3	1·5	2·4
	Organically bound	2	7·2	50·8	61·0
	Oxide bound	2	9·4	45·5	26·8
	Residual	2	87·1	3·3	9·8

Based on data taken from Murdock (1992)

The high proportion of irreversibly bound caesium is thought to reflect the high illite content of the Drigg stream sediment, to which it is known that caesium attaches strongly and preferentially. Both the americium and plutonium exist predominantly in the organically bound and oxide / hydroxide bound fractions.

b)   Radionuclide contour map and depth profiles

Jackson (1989) published data from a series of soil cores sampled from land immediately adjacent to the western edge of the Drigg site, during 1986 and 1987. Concentrations of ^{137}Cs were reported to be around 350 Bq kg^{-1} d.w. in the top 4 cm (range 234 to 456 Bq kg^{-1} d.w.), declining to approximately 100 Bq kg^{-1} at 4-6 cm, 60 Bq kg^{-1} at 6-10 cm, 50 Bq kg^{-1} at 10-14 cm and 30 Bq kg^{-1} at >14 cm. The soil in this area is a sand pararendzina with a dry bulk density of 710 kg m^{-3} and a total organic content of around 15%.

These data can be compared to caesium profiles obtained from an upland peaty soil area, approximately 15 km south-east of Drigg (Table 22), and to data from other studies in and around the Drigg site.

Table 22   Caesium concentration profiles in soil at Drigg and Corney Fell.

Sample site	Date of sample	Depth of soil sample (cm)	Concentration (Bq kg^{-1} d.w.)		Soil density (kg m^{-3})
			^{137}Cs	^{134}Cs	
Stoneside Hill, Corney Fell	22/11/76	0-4 4-6 6-10 10-14	11 57 42 21	not reported	not measured
	27/6/86 to 14/2/87 (n=4)	0-4 4-6 6-10 10-14	450 256 142 83	217 88 c. 48 <40	840 970 1070 1030
Drigg site	June 1986	0-4 4-6 6-8 8-10 10-12 12-14	449 222 162 63 33 34	113 19 13 5 2·1 1·5	
Drigg Headland	June 1986	0-4 4-6 6-8 8-10 10-12 12-14	416 223 167 150 99 65	110 13 5 5 4 0·6	
Drigg coastal pasture	3/10/86 to 31/10/87 (n=5)	0-4 4-6 6-10 10-14 14$^+$	350 100 60 50 30	107 10 5 3 nd	710

Note: Considerable uncertainty attach to the comparability of the data, since they were originally presented in differing units (e.g. Bq m^{-2}) and, particularly for the earlier sample, conversion to units of weight requires an assumed soil bulk density.

From Jackson et al. (1987), Jackson (1989) and Rudge (1989).

Although reservations may be expressed about the data as presented, the dominant source of caesium at both sites post-1986 appears to be from Chernobyl deposition. This appears to be confirmed by the ^{134}Cs/^{137}Cs ratios measured and by a separate observation (Rudge, 1989) that the apparent concentration of ^{137}Cs in surface soil on the Drigg site doubled between November 1985 and June 1986. Therefore, it is likely that groundwater migration through the soil at Drigg (e.g. arising from the trench disposals) is not a major source of contamination, at least in the near surface layers.

4.   Restoration options

The radioactive waste at the Drigg waste repository is largely confined to Trenches 1 to 7, with some now in Vault 8.   The waste mostly contains radioactive objects rather than contaminated soil.

Public access to the site is restricted.   However, accessibility to the waste for remediation is good.   Therefore, treatment on site would be the most appropriate option.

The remediation techniques which are available for treating the site are described in Technical Deliverable 3+4 (Zeevaert and Bousher, 1998).   Of the techniques descibed the ones considered appropriate to this site are as follows:

Physical separation	Filtration (for leachates only)	
Chemical separation	Chemical solubilisation (including solvent extraction) Ion exchange (for leachates only)	
Biological separation	Biosorption (for leachates only)	
Containment	Capping	
	Sub-surface barriers	Slurry walls Grout curtains
Physical immobilisation	*Ex-situ* *In-situ*	
Chemical immobilisation	*Ex-situ* *In-situ*	

A number of remediation techniques were rejected as being inappropriate to this site.   In particular, since the waste is already located in a repository the removal of the source to another site is not considered a sensible option.   The physical state of the waste also meant that separation approaches such as soil washing and flotation are rejected.

Technical Deliverable 3+4 gave ranges of values for the cost, performance and number of man hours required to remediate the waste.   These ranges were generic, with the expectation that they would be used to derive site-specific parameters for each remediation technique.

The values selected for a number of factors including: the type of waste material, the radionuclides present, their accessibility, the ease of excavation and the location of the site with respect to a suitable waste storage repository.

The site-specific values used for Drigg are summarised in Tables 24 to 26.

Table 23   Site-specific values for the performance of remediation techniques appropriate to the Drigg site.

Remediation Technique	Unit[†]	Value
Physical separation (*ex-situ*) Filtration	DF	7 (Cs) 100 (U, Am, Pu)
Chemical Separation (*ex-situ*) (including solvent extraction) Chemical solubilisation	DF	10
Ion exchange (liquids)	DF DF	33 (Cs) 5 (U, Am, Pu)
Biological Separation Biosorption (liquids)	DF	10
Containment Capping	k	1×10^{-10} m s^{-1}
Subsurface barriers	k	1×10^{-9} m s^{-1}
Physical Immobilisation	MRF	10
Chemical Immobilisation	MRF	10

[†] DF = decontamination factor; k = permeability coefficient; MRF = mobility reduction factor.

Table 24   **Site-specific values for the costs of remediation techniques appropriate to the Drigg site.**

Remediation Technique	Cost
Physical Separation (*ex-situ*) Filtration (liquids)	0·75 EUR m^{-3} (liquid)

Chemical Separation (*ex-situ*)
    Chemical solubilisation (including solvent extraction)
        Separation costs
        Cost of excavation and transport (prior to solubilisation)
            NB This does not include the cost of disposal
 	
    Ion exchange (liquids)
	
	550 EUR m^{-3}
	65 EUR m^{-3}
	
	2 EUR m^{-3}
	(liquid)

Biological Separation Biosorption (liquids)	2·5 EUR m^{-3} (liquid)

Containment
    Capping
 	
    Subsurface Barriers
            Slurry walls
 	
            Grout curtains
	
	35 EUR m^{-2}
	(surf. area)
	
	550 EUR m^{-2}
	(barrier)
	350 EUR m^{-2}
	(barrier)

Physical Immobilisation
    Ex-situ
        Cost of excavation and transport (prior to immobilisation)
 	
    In-situ
	
	35 EUR m^{-3}
	65 EUR m^{-3}
	
	350 EUR m^{-3}

Chemical Immobilisation
    Ex-situ
        Cost of excavation and transport (prior to immobilisation)
 	
    In-situ
	
	180 EUR m^{-3}
	65 EUR m^{-3}
	
	100 EUR m^{-3}

Table 25   Site-specific values for the exposure times (restoration workers) of remediation techniques appropriate to the Drigg site.

Remediation Technique	Exposure Time (Restoration Workers)
Physical separation (ex- situ) 　　Filtration (liquids)	0·7 manh m^{-3} (liquid)
Chemical Separation (ex-situ) (including solvent extraction) 　　Chemical solubilisation 　　　　Excavation and transport of the soil (prior to separation) 　　Ion exchange (liquids)	2·3 manh m^{-3} 0·7 manh m^{-3} 0·7 manh m^{-3} (liquid)
Biological Separation 　　Biosorption (liquids)	0·7 manh m^{-3} (liquid)
Containment 　　Capping 　　Subsurface barriers	0·05 manh m^{-2} (surf. area) 0·1 manh m^{-3} (barrier volume)
Physical Immobilisation 　　*Ex-situ* (including prior excavation and transport of the soil) 　　*In-situ*	0·6 manh m^{-3} 0·2 manh m^{-3}
Chemical Immobilisation 　　*Ex-situ* (including prior excavation and transport of the soil) 　　*In-situ*	0·6 manh m^{-3} 0·2 manh m^{-3}

The performance values and costs associated with each each techniques may be used to calculate the cost of treating the material in Trenches 1 to 7 and also the cost of disposal of any residue from the remediation procedure. These calculations are described in detail in Appendic C. The results of these calculations are summarised in Table 26.

Table 26   Cost of applying remediation technologies to the Drigg site.

Remediation strategy Drigg	Monetary costs of remediation [EUR]	
	Remediation (incl. labour)	Waste disposal (incl. transport)
Filtration	$3 \cdot 8 \times 10^8$	$3 \cdot 1 \times 10^7$
Chemical Solubilisation	$3 \cdot 0 \times 10^8$	$1 \cdot 0 \times 10^8$
Ion Exchange	$1 \cdot 0 \times 10^8$	$3 \cdot 1 \times 10^7$
Biosorption	$1 \cdot 9 \times 10^9$	$3 \cdot 1 \times 10^7$
Capping	$3 \cdot 5 \times 10^6$	0
Subsurface Barrier	$6 \cdot 3 \times 10^6$	0
Physical Immobilisation (*ex-situ*)	$5 \cdot 5 \times 10^7$	0
Physical Immobilisation (*in-situ*)	$1 \cdot 9 \times 10^8$	0
Chemical Immobilisation (*ex-situ*)	$1 \cdot 3 \times 10^8$	0
Chemical Immobilisation (*in-situ*)	$5 \cdot 5 \times 10^7$	0

5. Radiological impact assessment

5.1 Compartment scheme

The source is the low-level radioactive waste in solid form deposited at the Drigg site. After interaction between the solid phase and the infiltration water the activity is transported as leakage via drains to Drigg stream. Irrigation, was assumed, of agricultural soil of water from the stream and dredging of the sediments occurring on a yearly basis.

The compartment scheme for Drigg site is based on site information and is shown in Figure 14.
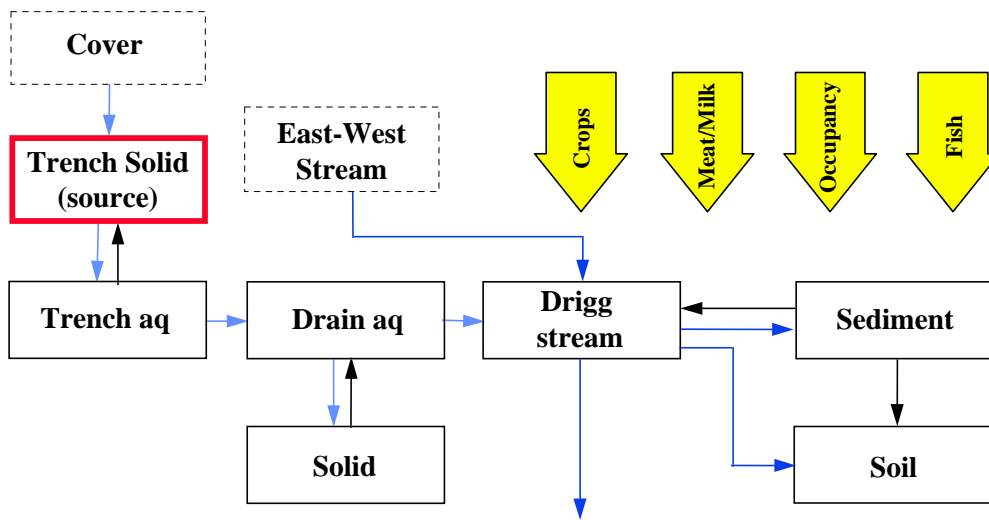
Figure 14   Compartment scheme for the Drigg site.

The radioisotopes of most interest are caesium-137, uranium-238, plutonium-239 and americium-241.

It should be noted that the half-life of caesium-137 is approximately 28 years. Consequently, over periods of 100 and 500 years it is the case that due to radioactive decay, the contribution of caesium-137 to the overall activity of a site will naturally decline. However, as a first approximation, for the purposes of the present calculations this was not taken into account.

5.2 Processes, exposure pathways and exposure groups

Each transfer coefficient can consist of several more or less complicated processes. For each major compartment the processes considered for the turnover of radionuclides in the model for the Drigg site are given in Table 27.

The equations and data used are given in Stiglund and Nordlinder (1999).

The important exposure pathways for the critical group are indicated in the compartment scheme (Figure 14). The exposure pathways considered are

- consumption of water;
- consumption of fish;
- consumption of milk and meat via the drinking water of the cattle;
- consumption of milk and meat via the pasturage for the cattle;
- consumption of root crops and vegetables;
- occupancy and inhalation.

Table 27   Processes considered within compartments.

Compartment	Processes
Water	Irrigation Sedimentation of particles Outflow
Sediment	Resuspension Dredging
Surface soil	Precipitation Evaporation Migration

The critical group are taken to be local farmers who eat locally produced food and drink water from the Drigg stream. They also consume fish from this stream.

The committed collective dose is based on production data.

Only external exposure from the soil is considered in the dose to the restoration workforce.

5.3   Doses to public/remediation workers

The doses calculated consist of individual and committed collective doses.

5.3.1   Individual doses to critical group: base case first year

The results of the dose calculations for the individual doses during the first year are shown in Table 28.

Table 28   Individual doses at Drigg site during the first year.

Exposure pathway	Individual dose (Sv a^{-1})				
	Cs-137	U-238	Pu-239	Am-241	Total
Water	$2 \cdot 9 \times 10^{-5}$	$7 \cdot 3 \times 10^{-5}$	$7 \cdot 4 \times 10^{-4}$	$9 \cdot 9 \times 10^{-5}$	$8 \cdot 4 \times 10^{-4}$
Fish	$2 \cdot 4 \times 10^{-4}$	$5 \cdot 5 \times 10^{-6}$	$1 \cdot 2 \times 10^{-5}$	$1 \cdot 4 \times 10^{-5}$	$2 \cdot 7 \times 10^{-4}$
Milk	$7 \cdot 5 \times 10^{-6}$	$5 \cdot 3 \times 10^{-7}$	$1 \cdot 4 \times 10^{-8}$	$3 \cdot 6 \times 10^{-8}$	$8 \cdot 0 \times 10^{-6}$
Meat	$4 \cdot 4 \times 10^{-6}$	$3 \cdot 3 \times 10^{-7}$	$2 \cdot 9 \times 10^{-8}$	$3 \cdot 9 \times 10^{-8}$	$4 \cdot 7 \times 10^{-6}$
Root crops	$1 \cdot 2 \times 10^{-6}$	$3 \cdot 4 \times 10^{-7}$	$2 \cdot 0 \times 10^{-6}$	$2 \cdot 6 \times 10^{-7}$	$3 \cdot 8 \times 10^{-6}$
Vegetables	$5 \cdot 5 \times 10^{-7}$	$1 \cdot 2 \times 10^{-6}$	$1 \cdot 2 \times 10^{-5}$	$1 \cdot 6 \times 10^{-6}$	$1 \cdot 5 \times 10^{-5}$
External exposure	$5 \cdot 1 \times 10^{-8}$	$4 \cdot 7 \times 10^{-35}$	$2 \cdot 2 \times 10^{-35}$	$1 \cdot 7 \times 10^{-11}$	$5 \cdot 1 \times 10^{-8}$
Inhalation	$8 \cdot 2 \times 10^{-13}$	$5 \cdot 2 \times 10^{-10}$	$3 \cdot 9 \times 10^{-9}$	$5 \cdot 2 \times 10^{-10}$	$4 \cdot 9 \times 10^{-9}$
Total dose	$2 \cdot 8 \times 10^{-4}$	$8 \cdot 1 \times 10^{-5}$	$7 \cdot 7 \times 10^{-4}$	$1 \cdot 1 \times 10^{-4}$	$1 \cdot 3 \times 10^{-3}$

It should be noted that irrigation is included in the calculations and this is the reason why the dose from root crops and vegetables are quite large. The inclusion of irrigation may not be so realistic since the Drigg site is situated in the north-west part of Britain where the rainfall is very large. Furthermore, these calculations have been based on the assumption that no restoration technique has been applied to the Drigg site, which is not a true reflection of the current situation (see Section 3.3.2.b).

The main contributor to the total dose is the intake of caesium-137 in fish.   Since caesium-137 is a gamma emitter it also dominates the external exposure pathway.

The contribution to the total dose from each pathway is shown in Figure 15.
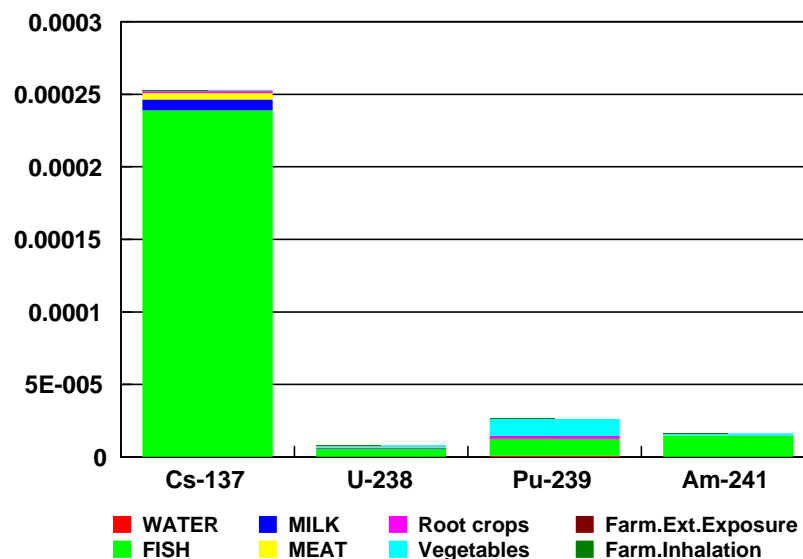
Figure 15   The contribution to the total dose (Sv a^{-1}) from different pathways.

Consumption of fish is the main exposure pathway for all nuclides. For caesium-137 fish is the overall dominating exposure pathway. For uranium-238 and plutonium-239 intake of vegetables makes a major contribution to the total dose while for americium-241 root crops also provides an important pathway.

5.3.2   Collective doses to public and restoration workers over 100 years and 500 years

Relevant restoration options for the Drigg site have been considered in Section 4. These are summarised in Table 29.

Table 29   Restoration options considered for the Drigg site.

Case	Restoration option
A	Base case (no restoration)
C2	Filtration
D1	Chemical solubilisation
D2	Ion exchange
D3	Biosorption
E1	Capping
E3	Subsurface barriers
F1	Cement-based solidification, *ex-situ*
F2	Cement-based solidification, *in-situ*
G1	Chemical immobilisation, *ex-situ*
G2	Chemical immobilisation, *in-situ*

The results of the dose calculations for the collective intakes/doses at 100 years and 500 years for the different restoration options are shown in Table 30. It should be noted that Option A refers to a hypothetical situation since restoration methods have already been applied at this site.

Table 30 **Mean Collective doses at Drigg site at 100 years and 500 years.**

Option	Group	Collective dose (manSv)					
		Year	Cs-137	U-238	Pu-239	Am-241	Total
A	Farmers	100	0·88	12	27	9·3	49
		500	0·95	27	76	17	$1·2 \times 10^2$
	Workers	-	-	-	-	-	-
C2	Farmers	100	0·14	$2·6 \times 10^{-2}$	0·39	0·36	0·93
		500	0·16	0·12	1·8	1·2	3·3
	Workers	-	$4·9 \times 10^{-10}$	$3·3 \times 10^{-11}$	$5·0 \times 10^{-10}$	$5·0 \times 10^{-10}$	$1·5 \times 10^{-9}$
D1	Farmers	100	0·12	2·3	4·9	2·6	9·9
		500	0·13	8·8	18	5·9	33
	Workers	-	$1·2 \times 10^{-9}$	$1·1 \times 10^{-10}$	$2·0 \times 10^{-10}$	$1·6 \times 10^{-10}$	$1·7 \times 10^{-9}$
D2	Farmers	100	$4·0 \times 10^{-2}$	4·0	8·4	3·9	16
		500	$4·4 \times 10^{-2}$	13	29	8·4	51
	Workers	-	$1·0 \times 10^{-10}$	$6·7 \times 10^{-11}$	$1·0 \times 10^{-10}$	$1·0 \times 10^{-10}$	$3·7 \times 10^{-10}$
D3	Farmers	100	0·16	3·2	6·6	3·2	13
		500	0·18	11	23	7·0	42
	Workers	-	$5·0 \times 10^{-10}$	$4·7 \times 10^{-11}$	$9·0 \times 10^{-11}$	$7·0 \times 10^{-11}$	$7·1 \times 10^{-10}$
E1	Farmers	100	$4·0 \times 10^{-3}$	$8·8 \times 10^{-2}$	0·19	0·15	0·43
		500	$4·4 \times 10^{-3}$	0·43	0·93	0·49	1·9
	Workers	-	$4·0 \times 10^{-12}$	$3·3 \times 10^{-13}$	$7·0 \times 10^{-13}$	$5·0 \times 10^{-13}$	$5·5 \times 10^{-12}$
E3	Farmers	100	$3·0 \times 10^{-2}$	0·63	1·4	0·87	2·9
		500	$3·3 \times 10^{-2}$	2·6	5·9	2·1	11
	Workers	-	$5·0 \times 10^{-10}$	$4·0 \times 10^{-11}$	$8·0 \times 10^{-11}$	$7·0 \times 10^{-11}$	$6·9 \times 10^{-10}$
F1	Farmers	100	$7·4 \times 10^{-2}$	1·2	2·2	0·78	4·2
		500	$8·0 \times 10^{-2}$	2·2	6·4	1·4	10
	Workers	-	$2·1 \times 10^{-9}$	$1·9 \times 10^{-10}$	$3·6 \times 10^{-10}$	$2·7 \times 10^{-10}$	$2·8 \times 10^{-9}$
F2	Farmers	100	$7·4 \times 10^{-2}$	1·1	2·2	0·78	4·2
		500	$8·0 \times 10^{-2}$	2·4	6·4	1·4	10
	Workers	-	$1·0 \times 10^{-9}$	$6·7 \times 10^{-11}$	$2·0 \times 10^{-10}$	$1·0 \times 10^{-10}$	$1·4 \times 10^{-9}$
G1	Farmers	100	$5·2 \times 10^{-2}$	0·75	1·6	0·55	2·9
		500	$5·6 \times 10^{-2}$	1·7	4·5	1·0	7·2
	Workers	-	$1·4 \times 10^{-9}$	$1·2 \times 10^{-10}$	$2·3 \times 10^{-10}$	$1·7 \times 10^{-10}$	$1·9 \times 10^{-9}$
G2	Farmers	100	$5·2 \times 10^{-2}$	0·75	1·6	0·55	2·9
		500	$5·6 \times 10^{-2}$	1·7	4·5	1·0	7·2
	Workers	-	$7·0 \times 10^{-10}$	$6·0 \times 10^{-11}$	$1·0 \times 10^{-10}$	$8·0 \times 10^{-11}$	$9·4 \times 10^{-10}$

It can be seen in Table 29 that, for the farmers, the contribution that plutonium-239 makes the biggest contribution to the total dose.

In Figure 16 the total collective dose for the critical group and the restoration workers are shown for plutonium-239 for the different restoration options at 100 years and 500 years.
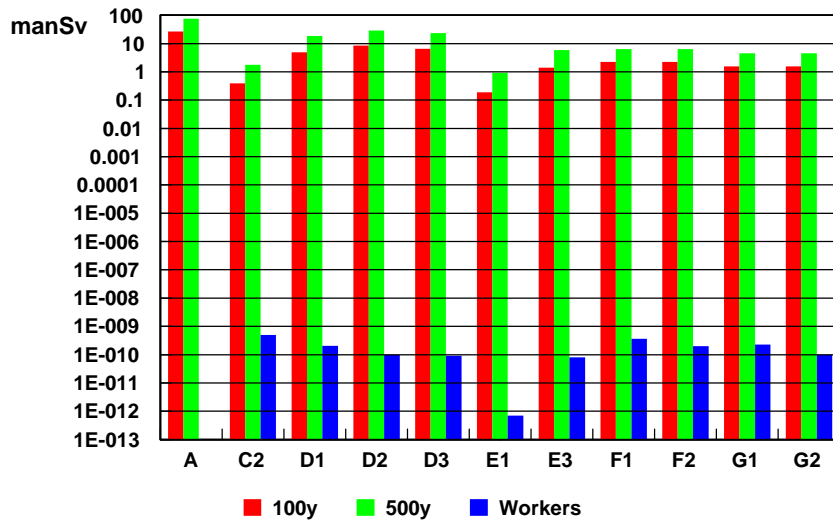
Figure 16 **The total collective dose of plutonium-239 for different restoration options at 100 and 500 years.**

It can be seen in Figure 16 that the reduction of dose to the critical group is quite small for the different restoration options except for Option E1 and, especially, Option C2. Restoration Option C2 is the filtration of the leachate from the waste in the trenches and Option E1 is the capping of the trenches.

The lowest dose to the remediation workers, when remediation is applied, occurs when capping of the trenches is being carried out.

5.4   Uncertainty analysis

Uncertainty analysis was carried out using PRISM (Stiglund and Nordlinder, 1999). This is a random sampling method, based upon the Latin Hyper Cube sampling method, whereby the response of the system to each individual input parameter is evaluated. The results show that for the Drigg site the amount of water flowing from the trenches and out of the system dominates the uncertainty for all nuclides concerned especially for caesium-137 and uranium-238. This is due to overestimated ranges for the different flows. For plutonium-239 and americium-241 the appointed distribution factor between solid and soluble fraction in the trenches, distribution constant value, also plays a major role for the uncertainty.

6. Ranking of restoration options

Decisions on clean-up in long-lasting exposure situations may well go far beyond purely radiological protection considerations. Satisfying the justification principle requires that the overall effect of the actions involved should do more good than harm, taking account of relevant radiological and non-radiological factors. Most decisions require multiple criteria to be taken into account. The field of multiple criteria analysis offers a number of approaches which take explicit account of multiple criteria in providing structure and support to the decision making process. In case of restoration of contaminated sites there are several criteria or attributes that need to be considered when choosing an 'optimum' restoration strategy. When the performance and costs of all the protection options have been assessed, a comparison is needed to define the *optimum* protection option. When the optimum is not self evident, the comparison can be carried using a quantitative decision aiding technique. Of the different techniques available, cost-benefit analysis has been used to evaluate if the remediation options are justified on economic grounds and multi-attribute utility (MAU) analysis for ranking different remediation options. Moreover, annual individual doses to critical groups before implementation of remedial measures have been assessed for comparison with the clean-up criteria recommended by IAEA.

6.1 Evaluation of remediation options (MAU analysis)

6.1.1 Attributes

The attributes that has been considered in this study includes:

- *Health attributes:*
    - collective doses to population;
    - doses to remediation workers;
    - non-radiological health factors.

- *Economic attributes:*
    - costs of remedial actions (including costs of labour and monitoring);
    - costs of monitoring of remedial options;
    - costs of disposal of generated waste (in broad categories);
    - loss/gain of taxes due to loss/gain of income.

- *Social attributes:*
    - reassurance of the public;
    - discomfort, disturbance and anxiety from the remedial action;
    - loss/gain of income.

The restoration options included for the Drigg site have been identified in Chapter 4 (Table 23). The economic and radiological data for quantifying the various attributes for each of those options are shown in Table 31.

6.1.2 Utilities

Utility functions for the attributes *monetary costs* and *radiation doses* have been calculated from the figures in Table 31 on monetary cost components and residual collective doses after remediation. Linear (risk neutral) utility functions have been used.

Table 31 **Remediation costs and collective doses to population and workers for different restoration strategies at the Drigg site.**

Restoration strategy	Collective dose to population [manSv]		Collective dose to workers [manSv]	Monetary costs of restoration [kEUR]			Fraction of activity left on-site	Waste volume [m³]
	100 y	500 y		Remedi-ation	Moni-toring	Waste disposal		
A	49	120	0	0	75,000	0	1	0
C2	0·93	3·3	$1·5 \times 10^{-9}$	380,000	750	31,000	0·01	12,500
D1	9·9	33	$1·7 \times 10^{-9}$	300,000	7,500	100,000	0·1	41,000
D2	16	51	$3·7 \times 10^{-10}$	1,000,000	15,000	31,000	0·2	12,500
D3	13	42	$7·1 \times 10^{-10}$	1,300,000	7,500	31,000	0·1	12,500
E1	0·43	1·9	$5·5 \times 10^{-12}$	3,500	75,000	0	1	0
E3	2·9	11	$6·9 \times 10^{-10}$	6,300	75,000	0	1	0
F1	4·2	10	$2·8 \times 10^{-9}$	55,000	75,000	0	1	0
F2	4·2	10	$1·4 \times 10^{-9}$	190,000	75,000	0	1	0
G1	2·9	7·2	$1·9 \times 10^{-9}$	130,000	75,000	0	1	0
G2	2·9	7·2	$9·4 \times 10^{-10}$	55,000	75,000	0	1	0

a)	*Utility functions for monetary costs*

Utility functions have been determined for remediation costs (including labour costs), waste disposal costs (including transport costs), monitoring costs and loss of taxes due to loss of income:

$$u_{remedia}(x) = 100 \cdot \left(1 - \frac{x}{1,300,000}\right) \quad \text{for } 0 \le x \le 1,300,000 \text{ kECU}$$

$$u_{waste}(x) = 100 \cdot \left(1 - \frac{x}{100,000}\right) \quad \text{for } 0 \le x \le 100,000 \text{ kECU}$$

$$u_{monitor}(x) = 100 \cdot \left(1 + \frac{750 - x}{75,000 - 750}\right) \quad \text{for } 750 \le x \le 75,000 \text{ kECU over 500 y}$$

b)	*Utility functions for health factors*

The following utility functions for the radiological health components have been determined for the exposed population and workers implementing the remedial actions. Only radiological health factors are considered for the Drigg site as no heavy metals are found.

$$u_{dose, \, pop,100}(x) = 100 \cdot \left(1 + \frac{0.43 - x}{49 - 0.43}\right) \text{ for } 0.43 \le x \le 49 \text{ man Sv}$$

$$u_{dose, \, pop,500}(x) = 100 \cdot \left(1 + \frac{1.9 - x}{120 - 1.9}\right) \text{ for } 1.9 \le x \le 120 \text{ man Sv}$$

$$u_{dose, \, work}(x) = 100 \cdot \left(1 - \frac{x}{2.8 \cdot 10^{-9}}\right) \text{ for } 0 \le x \le 2.8 \cdot 10^{-9} \text{ man Sv}$$

c) Utility functions for social factors

The utility function u_{reas} for reassurance would be linked to both the residual dose and the fraction of activity remaining on the site after the remedial measure has been implemented. However, the residual dose and remaining activity are not necessarily correlated. A remedial measure that has left all the activity on-site in a contained form (capping, surface barriers, etc.) might give a substantial dose reduction and thus a low value of the residual doses. Detailed information on how social factors like reassurance are linked with individual doses and activity concentration on site is not available. Therefore, utility functions for 100 years and 500 years integration time have been proposed which gives a low value only when both sub-utilities have low values:

$$u_{reas,100}(x, y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{0.43 - x}{49 - 0.43}\right)_{dose} + \frac{1}{2} \cdot \left(1 + \frac{0.01 - y}{1.0 - 0.01}\right)_{activity}\right)$$

for $0.43 \le x \le 49$ man Sv and $0.01 \le y \le 1$

$$u_{reas,500}(x, y) = 100 \cdot \left(\frac{1}{2} \cdot \left(1 + \frac{1.9 - x}{120 - 1.9}\right)_{dose} + \frac{1}{2} \cdot \left(1 + \frac{0.01 - y}{1.0 - 0.01}\right)_{activity}\right)$$

for $1.9 \le x \le 120$ man Sv and $0.01 \le y \le 1$

where y is the fraction of activity remaining on site after the remedial measures has been implemented. The value of the utility function u_{reas} will be 100 for a residual dose of 0·43 (1·9) manSv and a remaining fraction of the initial activity of 0·01 (best strategy) and 0 for a residual dose of 49 (120) manSv and a remaining activity fraction of 1·0 (worst strategy).

The utility function u_{distur} for disturbance has been related to the volume of soil and sediment waste to be transported to the waste disposal site:

$$u_{distur}(x) = 100 \cdot \left(1 - \frac{x}{41,000}\right) \text{ for } 0 \le x \le 41,000 \text{ m}^3$$

6.1.3 Weighting factors

Weighting factors are scaling factors that reflect the relative importance of each of the attributes. The weighting factors assigned to the different attributes at the same hierarchy level can be either the ratio of value ranges, R, of the attributes or by assigning values to the ratio of the weighting factors at that level. Both methods have been used here.

a) Weighting factors for major attributes

The major weighting factors considered in this study include those for monetary costs, health and social factors. The sum of these weighting factors should respect the following conditions:

$$w_{health} + w_{economic} + w_{social} = 1$$

The assessment of the weighting factors is discussed in Hedemann Jensen (1999) where conversion/scaling constants between weighting factors has been expressed as:

$$C_1 = \frac{w_{economic}}{w_{health}} \cong \frac{w_{economic}}{w_{dose,pop}} = \frac{R_{economic}}{\alpha \cdot R_{dose,pop}}$$

$$C_2 = \frac{w_{social}}{w_{health}} \approx \frac{r_{psy}}{r_{rad}}$$

C_1 can be determined for a 100 and 500 years integration time for the collective dose from the values given in Table 31. The value of C_2 is more difficult to assess but a value of $0\cdot2$-$0\cdot3$ has been argued for in Hedemann Jensen (1999). From these values of C_1 and C_2 the weighting factors for health, economic and social factors have been calculated as shown in Table 32.

b)   *Weighting factors for health sub-attributes*

The weighting factors for health sub-attributes include those of radiation induced stochastic health effects to the affected population and workers and non-radiation induced stochastic health effects to the affected population. The sum of these weighting factors should respect the following conditions:

$$w_{dose,pop} + w_{dose,work} + w_{non-rad} = 1$$

The conversion/scaling constant, C, for the health sub-attributes can according to Hedemann Jensen (1999) be expressed as:

$$w_{dose,pop} = C \cdot R_{dose,pop} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{dose,work} = C \cdot R_{dose,work} \cdot l \cdot r_{rad} \cong C \cdot R_{dose,pop}$$

$$w_{non-rad,pop} = C \cdot R_{non-rad,pop} \cdot l \cdot r_{non-rad}$$

Exposure to heavy metals is not relevant for the Drigg site and $R_{non-rad}$ is therefore zero. The value of C is given by (Hedemann Jensen, 1999):

$$C \cong \frac{1}{R_{dose,pop} + R_{dose,work}}$$

From the calculated values of C_{100} and C_{500} (for 100 and 500 years integrating time for the collective dose to the population) the weighting factors for collective population and worker doses have been calculated as shown in Table 32.

c)   *Weighting factors for economic sub-attributes*

The weighting factors for economic sub-attributes include those for cost of remediation, cost of waste disposal and costs of monitoring. The sum of these weighting factors should respect the following conditions:

$$w_{remedia} + w_{waste} + w_{monitor} = 1$$

The conversion/scaling constant, C, for the economic attributes can according to Hedemann Jensen (1999) be expressed as:

$$w_{remedia} = C \cdot R_{remedia}$$

$$w_{waste} = C \cdot R_{waste}$$

$$w_{monitor} = C \cdot R_{monitor}$$

The conversion/scaling constant, *C*, for the economic sub-attributes can be determined from the cost ranges in Table 31 and the weighting factors for remediation costs, waste disposal costs and monitoring costs have been calculated as shown in Table 32.

d)   *Weighting factors for social sub-attributes*

The weighting factors include those for reassurance and disturbance. The sum of these weighting factors should respect the following conditions:

$$w_{distur} + w_{reas} = 1$$

The conversion/scaling constants for the social sub-attributes can according to Hedemann Jensen (1999) be expressed as:

$$C_1 = \frac{w_{reas}}{w_{distur}}$$

In Hedemann Jensen (1999) it is argued that $w_{reas} > w_{distur}$ and that $C_1 \approx$ 5-7. From these values the weighting factors for disturbance and reassurance have been calculated as shown in Table 32.

Table 32   Weighting factors for attributes and sub-attributes applied in the optimisation of remediation of the Drigg site.

Health factors			Economic factors			Social factors		
0·0034	0·0083		0·996	0·990		0·00086	0·0021	
Dose population	1	1	Remediation costs		0·882	Reassurance		0·86
Dose workers	0	0	Waste disposal costs		0·068	Disturbance		0·14
Non-radiation	-	-	Monitoring costs		0·050	Loss/gain of income		-
			Loss/gain of taxes		-			

Note: The values in the left of the double columns are for an integration time of 100 years and in the right column for an integration time of 500 years.

It should be emphasised that value setting of weighting factors is the crucial issue of any optimisation because subjective judgements inevitably will enter the process.

6.2   Results

IAEA has proposed clean-up criteria in terms of individual doses. The individual doses assessed at the Drigg site are of the order of 1·3 mSv a^{-1} at the time of decision to introduce remediation (year 1). In practice, however, doses are known to be lower than this. According to the IAEA criteria, clean-up is almost always needed for an individual dose range of 1-10 mSv a^{-1} if a constraint for controlled practices is applied. Even without the application of a constraint, IAEA suggests that for individual doses of 1-10 mSv a^{-1} clean-up would usually be needed. Based on these recommendations it can therefore be concluded that some kind of remediation would almost always be justified for the Drigg site.

In a cost-benefit approach, the monetary costs, *X*, of the remediation strategies are compared with the benefit of the collective dose reduction, ΔS. The net benefit, ΔB, is given as:

$$\Delta B = \alpha \cdot \Delta S - (\alpha \cdot S_{work} + X_{remedia} + X_{waste} + X_{monitor})$$

None of the remedial options are justified on economic grounds alone when only the central estimates of collective dose are used together with an α-value of 100,000 EUR manSv^{-1}. A higher value of α (*e.g.* 200,000 EUR manSv^{-1}) and more extreme values from the calculated collective dose distribution (*e.g.* the 95th percentile) would make Options E1 and E3 economically justified, but only when the avertable collective dose is taken over 500 years.

In the multi-attribute approach, overall scores, U_i, of the remediation options *i* has been determined from the weighted sum of utilities for each of the attributes considered:

$$U_i = \sum_{j=1}^{3} w_j \cdot u_{ij}$$

$$= w_{health} \cdot (w_{dose,pop} \cdot u_{dose,pop} + w_{dose,work} \cdot u_{dose,work})$$

$$+ w_{economic} \cdot (w_{waste} \cdot u_{waste} + w_{remedia} \cdot u_{remedia} + w_{monitor} \cdot u_{monitor})$$

$$+ w_{social} \cdot (w_{distur} \cdot u_{distur} + w_{reas} \cdot u_{reas})$$

The weighting factors above have all been sampled in a triangular distribution between 0·67-1·5 times the most probable value given in Table 32. Similarly, the values of all the utilities, *u(x)*, are determined from the utility functions in which the values of *x* are sampled in a triangular distribution between 0·67-1·5 times the central values of *x* given in Table 31. Negative correlation between collective doses and remediation costs has been applied with a correlation coefficient of −0·8. The evaluation of the different strategies has been made with the forecasting and risk analysis program CRYSTAL BALL. Latin Hypercube Sampling technique was used and the number of trials were 10,000. The results for the scores, U_i, for the Options A-G2 are shown in Figure 17. The error bars represent the 5th and 95th percentiles of the distributions of U_i.
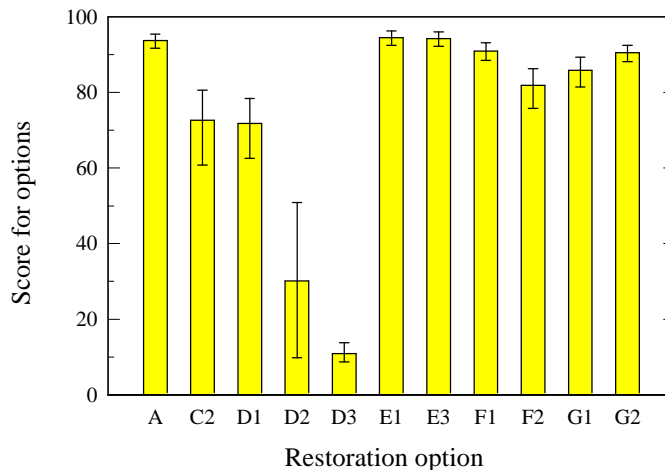
Note: Identical scores are found for an integration time of 100 years.

Figure 17   Overall evaluation of scores for different remediation strategies for the Drigg site for an integration time of 500 years for the collective dose.

It appears from Figure 17 that the Options E1 (capping) and E3 (sub-surface barriers)have the highest score, closely followed by the Option A (no remediation). Also the Options F1 (*ex-*

situ cement-based solidification) and G2 (*in-situ* chemical immobilisation) have a high and practically an equal score. Therefore, it might be difficult to pick an optimum solution.

The ranking of the different remedial measures using multi-attribute utility analyses allows the inclusion of factors that are not easy to quantify in monetary terms as is required in cost-benefit analysis. The weighting factors assigned to the different attributes have been determined by use of scaling factors in terms of weighting factor ratios, and their values were sampled around a most probable value. Notwithstanding this advantage of the multi-attribute method there are difficulties with the determination of weighting factors for the different attributes. Without any terms of reference for the weighting between attributes, value settings by a decision maker could lead to 'optimised' results that might be useless because of a subjective bias of the decision maker in the selection of weighting factors. Therefore, the outcome of any multi-attribute analysis, including the present study, should be judged very carefully in the light of the values assigned to the weighting factors before any firm conclusions could be drawn.

7. Conclusions

The purpose of this technical deliverable has been to provide an example site to demonstrate the application of the decision-making procedure, developed by RESTRAT, to remediate the Drigg low level waste disposal site. Each step of the proposed procedure has been carried out. The site was characterised through a review of the available data, and a number of areas of contamination were identified along with pathways through which man can be exposed to these contaminants. Characterisation enabled the doses, arising from the site, to be quantified through the development of a model. Restoration techniques which were appropriate to this site were identified and the impact of these doses on the locality were calculated from the model. These performances of the techniques were combined with the economic and social costs in a multi-attribute utility analysis to rank each restoration technique.

This exercise was performed on a hypothetical case, as it was assumed that no remediation (capping) had previously been carried out on the site. Thus applying the conclusions of this particular study to the current situation must be viewed with some caution.

The results from multi-attribute utility analysis show that there was three of options which could have been applied to the site. This included the 'no remediation' option whereby the site was simply monitored. However, the options which achieved the highest scores were containment options (capping and sub-surface barrier methods). This is in agreement with the restoration options which were actually carried out at the Drigg site.

Although the results from the multi-attribute utility analysis show that capping was the best option for this example site, the high level of uncertainties meant that there was significant overlap in the ranking of the next two restoration techniques (sub-surface barriers and the 'no remediation options). The uncertainty arose both from the lack of precision in characterising the site and from the uncertainties associated with the restoration techniques themselves.

The uncertainties associated with the parameters of the site reflect the level of assessment detail in the current study. As a consequence, the migration of the radionuclides from the trenches, and their subsequent distribution throughout the site, includes high levels of uncertainty. This turn leads to a lack of precision with regard to defining the performance of each restoration method. These can only be defined in terms of a range of values which generates the high levels of uncertainty in the multi-attribute utility analysis.

In spite of these problems presented by the lack of precision of the characterisation of the site, the multi-attribute utility analysis demonstrates that it is a fairly robust approach which produces realistic decisions which are consistent with previous studies of the Drigg site.

8. Acknowledgements

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Appendix A   The water column

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The compartment structure (see Section 5.1) includes three sections with an aquatic phase, namely the trenches, the drainage channels, and the Drigg stream. The composition of the trenches is extremely heterogeneous, and, as a consequence, it is not possible to measure and characterise or to model. For this compartment a chemical speciation computation to unfold the K_d is not feasible. For this special case the K_d concept will be retained.

Water samples were taken by Westlakes Scientific Consulting Ltd (WSC) from the East-West stream just outside the Drigg site and from the Drigg stream after leaving the site. These samples have been analysed in parallel by the Forschungszentrum Rossendorf (FZR) and WSC, and have shown good conformity, inside the experimental uncertainties. Additionally, two rain water samples were analysed, and also information about groundwater composition from various perimeters at the side borders are available.

A1   Rainwater analysis

Rainwater was collected over a four week period ending March 4 1997 from the Drigg site. This was analysed for cations and anions in a threefold analysis of an accumulated sample (Samples Ia, Ib and Ic). The results are summarised in Table A1.

Table A1   Constituents of rain water from the Drigg site

Component	Sample			Molecular. Weight.	Sample:			Method
	Ia $mg\ m^{-3}$	Ib $mg\ m^{-3}$	Ic $mg\ m^{-3}$	$g\ mol^{-1}$	Ia $mol\ L^{-1}$	Ib $mol\ L^{-1}$	Ic $mol\ L^{-1}$	
PO_4^{3-}	< 2000			94·971	$<2·1 \times10^{-5}$			IC
NO_3^-	< 1000			62·005	$<1·5 \times10^{-5}$			IC
NO_2^-	< 1000			46·0055	$<2·2 \times10^{-5}$			IC
SO_4^{2-}	1600			96·058	$1·666 \times10^{-5}$			IC
HCO_3^-	880			61·017	$1·442 \times10^{-5}$			IC
Cl^-	19900			35·453	$5·613 \times10^{-4}$			IC
K^+	440	420	440	39·098	$1·125 \times10^{-5}$	$1·074 \times10^{-5}$	$1·125 \times10^{-5}$	F-AAS
Na^+	10400	10100	10300	22·99	$4·524 \times10^{-4}$	$4·393 \times10^{-4}$	$4·480 \times10^{-4}$	F-AAS
Ca^{2+}	500	460	490	40·08	$1·248 \times10^{-5}$	$1·148 \times10^{-5}$	$1·223 \times10^{-5}$	ICP-MS
Mg^{2+}	1260	1140	1253	24·305	$5·184 \times10^{-5}$	$4·690 \times10^{-5}$	$5·155 \times10^{-5}$	F-AAS
Fe	< 50	< 50	< 50	55·847	$<9 \times10^{-7}$	$<9 \times10^{-7}$	$<9 \times10^{-7}$	F-AAS
Zn^{2+}	5·3	8·5	5·4	65·38	$8·106 \times10^{-8}$	$1·300 \times10^{-7}$	$8·259 \times10^{-8}$	ICP-MS
Pb^{2+}	0·5	0·2	0·4	207·2	$2·413 \times10^{-9}$	$9·653 \times10^{-10}$	$1·931 \times10^{-9}$	ICP-MS
Ni^{2+}	< 10	< 10	< 10	58·7	$<1·7 \times10^{-7}$	$<1·7 \times10^{-7}$	$<1·7 \times10^{-7}$	ICP-MS
Mn^{2+}	1·6	1·2	1·5	54·938	$2·912 \times10^{-8}$	$2·184 \times10^{-8}$	$2·730 \times10^{-8}$	ICP-MS
Cd^{2+}	< 0·1	< 0·2	< 0·1	112·41	$<8·9 \times10^{-10}$	$<1·8 \times10^{-9}$	$<8·9 \times10^{-10}$	ICP-MS

Note: Upper limit values denote, that the analytical value was below the respective detection limit.
The analytical methods are: IC = Ion Chromatography, ICP-MS = Inductively Coupled Plasma - Mass Spectrometry, F-AAS = Flame - Atom Adsorption Spectrometry.

The concentrations of silica, arsenic, aluminium, uranium and thorium were below detection limit. These have not been included in Table A1.

A2   Analysis of the East-West stream water

Several sets of analytical data have been generated of the composition of water from the East-West stream. Table A2 gives the values for a point (0621 East; 9928 North) upstream, shortly before the stream enters the Drigg site. Samples were collected by WSC on

December 4, 1996, and analysed in parallel from FZR (Sample II) and WSC (Sample III). FZR also filtered part of sample II: with a 10 kDalton filter for the determination of cations, and with a 25 kDalton filter for the determination of anions (Sample IIa). The coincidence between the results from FZR and Westlakes was considered to be good. The only difference concerning the iron content may have been due to the fact that WSC did not filter the sample before analysis.

Table A2   Constituents of East-West stream water before entering the Drigg site.

Component	Sample			Molecular. Weight.	Sample:			Method
	II mg m^{-3}	**IIa** mg m^{-3}	**III** mg m^{-3}	g mol^{-1}	**II** mol L^{-1}	**IIa** mol L^{-1}	**III** mol L^{-1}	
PO$_4^{3-}$	<2000	<2000	500	94·971	<2·1 ×10^{-5}	<2·1 ×10^{-5}	5·265 ×10^{-6}	IC
NO$_3^-$	56100	52000	45300	62·005	9·048 ×10^{-4}	8·386 ×10^{-4}	7·306 ×10^{-4}	IC
NO$_2^-$	n.d.	< 1000	521	46·0055	<2·2 ×10^{-5}	<2·2 ×10^{-5}	1·132 ×10^{-5}	IC
NH$_4^+$	n.d.	n.d.	1060	18·0383			5·876 ×10^{-5}	
SO$_4^{2-}$	30700	26500	29000	96·058	3·196 ×10^{-4}	2·759 ×10^{-4}	3·019 ×10^{-4}	IC
HCO$_3^-$	66600	67500	63500	61·017	1·091 ×10^{-3}	1·106 ×10^{-3}	1·041 ×10^{-3}	IC
Cl$^-$	48500	49200	52000	35·453	1·368 ×10^{-3}	1·388 ×10^{-3}	1·467 ×10^{-3}	IC
Si	2560	3160	2500	28·086	9·115 ×10^{-5}	1·125 ×10^{-4}	8·901 ×10^{-5}	ICP-MS
K$^+$	10810	10650	11000	39·098	2·765 ×10^{-4}	2·724 ×10^{-4}	2·813 ×10^{-4}	F-AAS
Na$^+$	20000	20080	21700	22·99	8·699 ×10^{-4}	8·734 ×10^{-4}	9·439 ×10^{-4}	F-AAS
Ca^{2+}	46770	45300	43000	40·08	1·167 ×10^{-3}	1·130 ×10^{-3}	1·073 ×10^{-3}	ICP-MS
Mg^{2+}	6030	6205	6440	24·305	2·481 ×10^{-4}	2·553 ×10^{-4}	2·650 ×10^{-4}	F-AAS
Fe	0	0	150	55·847	0	0	2·686 ×10^{-6}	F-AAS
Al^{3+}	56·7	27·9	n.d.	26·9815	2·101 ×10^{-6}	1·034 ×10^{-6}		ICP-MS
Zn^{2+}	0	43·9	n.d.	65·38	0	6·715 ×10^{-7}		ICP-MS
U	0·1	0·1	n.d.	238·03	4·201 ×10^{-10}	4·201 ×10^{-10}		ICP-MS
Th	<1	1·7	n.d.	232·0381	<4 ×10^{-9}	7·326 ×10^{-9}		ICP-MS
Ni^{2+}	0	6·8	n.d.	58·7	0	1·158 ×10^{-7}		ICP-MS
Mn^{2+}	0	2	n.d.	54·938	0	3·640 ×10^{-8}		ICP-MS
As	2·8	1·5	n.d.	74·922	3·737 ×10^{-8}	2·002 ×10^{-8}		ICP-MS

Note: Upper limit values denote, that the analytical value was below the respective detection limit.
n.d. stands for "not determined".
The analytical methods are: IC = Ion Chromatography, ICP-MS = Inductively Coupled Plasma - Mass Spectrometry, F-AAS = Flame - Atom Adsorption Spectrometry.

The values presented in the table below, the pH was measured *in-situ* and found to be 6·43, which is smaller then the values reported by Rudge (1989) (6·60) and Murdock (1992) (7·20). However, it is not known whether those values were also determined *in-situ*. It was found that after allowing the water to stand for three weeks, pH rose from 6·43 to 7·62, so external pH measurements may give values significantly too large.

Rudge (1989) reported a K$^+$ concentration of 14·6 g m^{-3}. This gives reasonable agreement with values from this project.

Murdock (1992) also reported a value of 1·2 g m^{-3} for total iron. This is much larger than the upper limit for truly dissolved iron, again indicating a measurement of the sum of dissolved and colloidal material with insufficient filtering.

The redox potential,, measured *in-situ* to be 188 mV, with a dissolved oxygen content of 74%. Suspended material was determined as 20·4 g m^{-3}, the conductivity was 368 ms cm^{-1}, which is in close agreement to the value of 351 ms cm^{-1} reported by Murdock (1992).

γ-Spectroscopy gave values of 0·82 Bq kg^{-1} for ^{226}Ra, and 0·05 Bq kg^{-1} for ^{232}Th, corresponding to 9·9 ×10^{-14} mol L^{-1} and 5·3 ×10^{-8} mol L^{-1}, respectively.

WSC sampled the East-West stream (Sample IV) at a point shortly before it entered the Drigg site on December 4 1996. The cationic and anionic constituents were analysed and the results are summarised in Table A3. The reproducibilities are also given.

Table A3   Constituents of East-West stream water before entering the Drigg site.

Component	Molecular Weight	Sample IV		Sample IV (error)		
	g mol^{-1}	mg m^{-3}	mol L^{-1}	mg m^{-3}	mol L^{-1}	%
PO_4^{3-}	94.971	160	1.685×10^{-6}	30	3.159×10^{-7}	18.75
NO_3^-	62.005	47300	7.628×10^{-4}	3800	6.129×10^{-5}	8.03
NO_2^-	46.0055	162	3.521×10^{-6}	5	1.087×10^{-7}	3.09
NH_4^+	17.0306	520	3.053×10^{-5}	40	2.349×10^{-6}	7.69
SO_4^{2-}	96.058	38500	4.008×10^{-4}	2900	3.019×10^{-5}	7.53
HCO_3^-	60.01	83000	1.383×10^{-3}	2000	3.333×10^{-5}	2.41
Cl^-	35.453	56000	1.580×10^{-3}	6000	1.692×10^{-4}	10.71
SiO_2	60.085	11600	1.931×10^{-4}	300	4.993×10^{-6}	2.59
K^+	39.098	6400	1.637×10^{-4}	400	1.023×10^{-5}	6.25
Na^+	22.99	34900	1.518×10^{-3}	600	2.610×10^{-5}	1.72
Ca^{2+}	40.08	55500	1.385×10^{-3}	5800	1.447×10^{-4}	10.45
Mg^{2+}	24.305	9700	3.991×10^{-4}	400	1.646×10^{-5}	4.12
Fe	55.847	1000	*1.791×10^{-5}*	100	*1.791×10^{-6}*	*10.00*
Al^{3+}	26.9815	130	*4.818×10^{-6}*	70	*2.594×10^{-6}*	*53.85*

The conductivity was 408 mS cm^{-1}, the pH was 6.05, dissolved oxygen was 112% and the redox potential was 126.1 mV.

As indicated by both their high concentration values and their large internal error, the iron and aluminium concentrations are the sum of dissolved and colloidal material. This indicates insufficient filtering.

A comparison between the values in Tables A2 and A3 show little significant difference. This is a clear sign that during its course through the Drigg site, the water of the East-West stream does not undergo large changes in composition.

A3   Analysis of Drigg stream water

The Drigg stream was sampled three times during 1996 and 1997 and provided three sets of data.

The Drigg stream was sampled, on March 4, 1997 (Sample V), shortly after the confluence with the East-West stream on the Drigg site. The water sample was analysed by WSC and the results given in Table A4. Reproducibilities are again included.

The conductivity was 487 mS cm^{-1}, the pH was 6.23, dissolved oxygen was 112.1% and the redox potential was 100.0 mV. Iron and aluminium concentrations are the sum of dissolved and colloidal material (see Section A2).

Table A4   Constituents of the Drigg stream shortly after the confluence with the East-West stream on the Drigg site

Component	Molecular Weight g mol^{-1}	Sample V		Sample V (error)		
		mg m^{-3}	mol L^{-1}	mg m^{-3}	mol L^{-1}	%
PO_4^{3-}	94·971	210	$2·211 \times 10^{-6}$	40	$4·212 \times 10^{-7}$	19·05
NO_3^-	62·005	38300	$6·177 \times 10^{-4}$	3400	$5·483 \times 10^{-5}$	8·88
NO_2^-	46·0055	142	$3·087 \times 10^{-6}$	5	$1·087 \times 10^{-7}$	3·52
NH_4^+	17·0306	490	$2·877 \times 10^{-5}$	40	$2·349 \times 10^{-6}$	8·16
SO_4^{2-}	96·058	41000	$4·268 \times 10^{-4}$	3000	$3·123 \times 10^{-5}$	7·32
HCO_3^-	60·01	90000	$1·500 \times 10^{-3}$	2000	$3·333 \times 10^{-5}$	2·22
Cl^-	35·453	53000	$1·495 \times 10^{-3}$	6000	$1·692 \times 10^{-4}$	11·32
SiO_2	60·085	6800	$1·132 \times 10^{-4}$	200	$3·329 \times 10^{-6}$	2·94
K^+	39·098	6400	$1·637 \times 10^{-4}$	100	$2·558 \times 10^{-6}$	1·56
Na^+	22·99	37200	$1·618 \times 10^{-3}$	400	$1·740 \times 10^{-5}$	1·08
Ca^{2+}	40·08	65600	$1·637 \times 10^{-3}$	5500	$1·372 \times 10^{-4}$	8·38
Mg^{2+}	24·305	10200	$4·197 \times 10^{-4}$	300	$1·234 \times 10^{-5}$	2·94
Fe	55·847	1000	*$1·791 \times 10^{-5}$*	400	*$7·162 \times 10^{-6}$*	*40·00*
Al^{3+}	26·9815	200	*$7·412 \times 10^{-6}$*	10	*$3·706 \times 10^{-7}$*	*5·00*

Table A5 contains values from an analysis of water from the Drigg stream after leaving the Drigg site to the south at the location 0629 East, 9827 North. The water was collected by WSC on December 4, 1996, and analysed in parallel with FZR (Sample VI) and WSC (Sample VII). FZR also filtered part of sample VI: with a 10 kDalton filter, for the determination of cations, and with a 25 kDalton filter for the determination of anions (Sample VIa).

The coincidence of the results from FZR and WSC is considered as being very good, the only difference concerning the iron content may be due to the fact that WSC did not filter the sample before analysing.

The values reported by Murdock (1992) are also included in Table A4. These are in good agreement with the values from the 1996 sampling campaign by WSC. Again, the iron content is the exception for the reasons described above.

As with the East-West stream, the pH differs significantly. *In-situ* measurements by WSC yielded 6·68 (drifting to 7·67 during three weeks), which is in good agreement to Rudge (1989) (6.80), but not to Murdock (1992) (7.40). However, it is not known whether the latter value was determined *in-situ*.

The differences in the redox potential are not significant. Murdock (1992) reported 107·8 mV, whereas *in-situ* measurements in 1996 gave 80·4 mV.

Thorium concentration was determined with γ-spectroscopy at 0·16 Bq kg^{-1}, corresponding to $1·7 \times 10^{-7}$ mol L^{-1}. Radium concentrations were below the detection limit.

Table A5   Constituents of the Drigg stream after leaving the Drigg site.

Component	Sample				Molecular Weight	Sample				Method
	VI	VIa	VII	Murdock, 1992		VI	VIa	VII	Murdock, 1992	
	mg m^{-3}	mg m^{-3}	mg m^{-3}	mg m^{-3}	g mol^{-1}	mol L^{-1}	mol L^{-1}	mol L^{-1}	mol L^{-1}	
PO$_4^{3-}$	<2000	<2000	<300	n.d.	94·971	<2·1 ×10^{-5}	<2·1 ×10^{-5}	<3·2 ×10^{-6}		IC
NO$_3^-$	38600	37700	31700	n.d.	62·005	6·225 ×10^{-4}	6·080 ×10^{-4}	5·112 ×10^{-4}		IC
NO$_2^-$	n.d.	<1000	266	n.d.	46·0055		<2·2 ×10^{-5}	5·782 ×10^{-6}		IC
NH$_4^+$	n.d.	n.d.	390	n.d.	18·0383			2·162 ×10^{-5}		
SO$_4^{2-}$	34500	33900	36000	n.d.	96·058	3·592 ×10^{-4}	3·529 ×10^{-4}	3·748 ×10^{-4}		IC
HCO$_3^-$	102000	110000	91300	n.d.	61·017	1·672 ×10^{-3}	1·803 ×10^{-3}	1·496 ×10^{-3}		IC
Cl$^-$	59000	60600	58000	n.d.	35·453	1·664 ×10^{-3}	1·709 ×10^{-3}	1·636 ×10^{-3}		IC
Si	3290	4008	3200	n.d.	28·086	1·171 ×10^{-4}	1·427 ×10^{-4}	1·139 ×10^{-4}		ICP-MS
K$^+$	8180	8290	9700	6100	39·098	2·092 ×10^{-4}	2·120 ×10^{-4}	2·481 ×10^{-4}	1·560 ×10^{-4}	F-AAS
Na$^+$	28200	28960	33100	38900	22·99	1·227 ×10^{-3}	1·260 ×10^{-3}	1·440 ×10^{-3}	1·692 ×10^{-3}	F-AAS
Ca^{2+}	51580	51150	46500	46100	40·08	1·287 ×10^{-3}	1·276 ×10^{-3}	1·160 ×10^{-3}	1·150 ×10^{-3}	ICP-MS
Mg^{2+}	6780	7050	7430	5700	24·305	2·790 ×10^{-4}	2·901 ×10^{-4}	3·057 ×10^{-4}	2·345 ×10^{-4}	F-AAS
Fe	0	0	430	4400	55·847	0	0	7·700 ×10^{-6}	7·879 ×10^{-5}	F-AAS
Al^{3+}	28·5	16·8	n.d.	n.d.	26·9815	1·056 ×10^{-6}	6·226 ×10^{-7}			ICP-MS
Zn^{2+}	0	54·6	n.d.	n.d.	65·38	0	8·351 ×10^{-7}			ICP-MS
U	0·7	0·8	n.d.	n.d.	238·03	2·941 ×10^{-9}	3·361 ×10^{-9}			ICP-MS
Th	< 0·1	1·7	n.d.	n.d.	232·0381	<4 ×10^{-9}	7·326 ×10^{-9}			ICP-MS
Pb^{2+}	0	0·2	n.d.	n.d.	207·2	0	9·653 ×10^{-10}			ICP-MS
Ni^{2+}	0	39·6	n.d.	n.d.	58·7	0	6·746 ×10^{-7}			ICP-MS
Cd^{2+}	0	0·2	n.d.	n.d.	112·41	0	1·779 ×10^{-7}			ICP-MS
As	2·8	1·3	n.d.	n.d.	74·922	3·737 ×10^{-8}	1·735 ×10^{-8}			ICP-MS

An additional check for filtering effects was carried out in November 1997. This showed a reduction of the aluminium content by 37% and of the iron content by 19% when applying a 200 nm filter.

The results are given in Table A6, with Sample VIII for the unfiltered and Sample VIIIa for the filtered sample. At FZR, the finest filters used for sample preparation had an approximate pore size of 15 nm, which then excluded nearly all of the colloidal material.

Table A6   Filtering effects on analysis of water from the Drigg stream after leaving the Drigg site to the south.

Component	Sample		MW	Sample	
	VIII	VIIIa		VIII	VIIIa
	mg m^{-3}	mg m^{-3}	g mol^{-1}	mol L^{-1}	mol L^{-1}
SO$_4^{2-}$	42000	44000	96·058	4·372 ×10^{-4}	4·581 ×10^{-4}
Cl$^-$	58000	58000	35·453	1·636 ×10^{-3}	1·636 ×10^{-3}
SiO$_2$	7810	8410	60·08	1·300 ×10^{-4}	1·400 ×10^{-4}
K$^+$	9000	10000	39·098	2·302 ×10^{-4}	2·558 ×10^{-4}
Na$^+$	32600	31900	22·99	1·418 ×10^{-3}	1·388 ×10^{-3}
Fe	625	508	55·847	1·119 ×10^{-5}	9·096 ×10^{-6}
Al^{3+}	39	24·6	26·9815	1·445 ×10^{-6}	9·117 ×10^{-7}

Table A7 reports the concentration values for a water sample from the drainage channel short before the confluence with the East-West stream at the Drigg site (Sample IX). The water was collected on March 4, 1997, and analysed by WSC. The reproducibilities are also given.

Table A7 **Constituents of the drainage channel shortly before the confluence with the East-West stream on the Drigg site**

Component	Molecular Weight	Sample IX		Sample IX (error)		
	g mol^{-1}	mg m^{-3}	mol L^{-1}	mg m^{-3}	mol L^{-1}	%
PO$_4^{3-}$	94·971	190	$2·001 \times 10^{-6}$	30	$3·159 \times 10^{-7}$	15·79
NO$_3^-$	62·005	4400	$7·096 \times 10^{-5}$	1700	$2·742 \times 10^{-5}$	38·64
NO$_2^-$	46·0055	17	$3·695 \times 10^{-7}$	3	$6·521 \times 10^{-8}$	17·65
NH$_4^+$	17·0306	490	$2·877 \times 10^{-5}$	40	$2·349 \times 10^{-6}$	8·16
SO$_4^{2-}$	96·058	54500	$5·674 \times 10^{-4}$	3600	$3·748 \times 10^{-5}$	6·61
HCO$_3^-$	60·01	150000	$2·500 \times 10^{-3}$	150	$2·500 \times 10^{-6}$	0·10
Cl$^-$	35·453	85000	$2·398 \times 10^{-3}$	85	$2·398 \times 10^{-6}$	0·10
SiO$_2$	60·085	10300	$1·714 \times 10^{-4}$	300	$4·993 \times 10^{-6}$	2·91
K$^+$	39·098	4800	$1·228 \times 10^{-4}$	400	$1·023 \times 10^{-5}$	8·33
Na$^+$	22·99	54200	$2·358 \times 10^{-3}$	1400	$6·090 \times 10^{-5}$	2·58
Ca^{2+}	40·08	74300	$1·854 \times 10^{-3}$	5700	$1·422 \times 10^{-4}$	7·67
Mg^{2+}	24·305	13400	$5·513 \times 10^{-4}$	400	$1·646 \times 10^{-5}$	2·99
Fe	55·847	2300	*$4·118 \times 10^{-5}$*	700	*$1·253 \times 10^{-5}$*	*30·43*
Al^{3+}	26·9815	4670	*$1·731 \times 10^{-4}$*	1630	*$6·041 \times 10^{-5}$*	*34·90*

In-situ measurements yielded a conductivity of 594 mS cm^{-1}, a pH of 6·45, dissolved oxygen at 83·9% and a redox potential of 59·7 mV. The iron and aluminium concentrations are the sum of dissolved and colloidal material (see above).

A4   Selected concentration for the waters on the Drigg site

The set of selected best values for the drainage channel and Drigg stream compartments is listed in Table A8. Mean values and standard deviations were calculated assigning equal weights to all listed values. Exceptions are aluminium and iron, where only the values from filtered samples were taken into consideration, and the pH value, where the unusually high values from Murdock (1992) were discarded.

Table A8   **Selected best set of aqueous phase compositions for the Drigg site.**

Component	Compartment			
	Drainage channel		Drigg stream	
	Mean	Stand. Dev.	Mean	Stand. Dev.
F^-			$4\cdot545 \times 10^{-7}$	-
PO_4^{3-}	$2\cdot001 \times 10^{-6}$	$4\cdot212 \times 10^{-7}$	$1\cdot259 \times 10^{-6}$	$1\cdot346 \times 10^{-6}$
NO_3^-	$7\cdot096 \times 10^{-5}$	$5\cdot483 \times 10^{-5}$	$4\cdot720 \times 10^{-4}$	$2\cdot675 \times 10^{-4}$
NO_2^-	$3\cdot695 \times 10^{-7}$	$1\cdot087 \times 10^{-7}$	$3\cdot036 \times 10^{-6}$	$2\cdot771 \times 10^{-6}$
NH_3	$2\cdot877 \times 10^{-5}$	$2\cdot349 \times 10^{-6}$	$1\cdot687 \times 10^{-5}$	$1\cdot486 \times 10^{-5}$
SO_4^{2-}	$5\cdot674 \times 10^{-4}$	$3\cdot123 \times 10^{-5}$	$4\cdot015 \times 10^{-4}$	$4\cdot469 \times 10^{-5}$
CO_3^{2-}	$2\cdot500 \times 10^{-3}$	$3\cdot333 \times 10^{-5}$	$1\cdot618 \times 10^{-3}$	$1\cdot481 \times 10^{-4}$
Cl^-	$2\cdot398 \times 10^{-3}$	$1\cdot692 \times 10^{-4}$	$1\cdot629 \times 10^{-3}$	$7\cdot184 \times 10^{-5}$
SiO_2	$1\cdot714 \times 10^{-4}$	$3\cdot329 \times 10^{-6}$	$1\cdot262 \times 10^{-4}$	$1\cdot326 \times 10^{-5}$
K^+	$1\cdot228 \times 10^{-4}$	$2\cdot558 \times 10^{-6}$	$2\cdot107 \times 10^{-4}$	$3\cdot876 \times 10^{-5}$
Na^+	$2\cdot358 \times 10^{-3}$	$1\cdot740 \times 10^{-5}$	$1\cdot435 \times 10^{-3}$	$1\cdot714 \times 10^{-4}$
Ca^{2+}	$1\cdot854 \times 10^{-3}$	$1\cdot372 \times 10^{-4}$	$1\cdot302 \times 10^{-3}$	$1\cdot975 \times 10^{-4}$
Mg^{2+}	$5\cdot513 \times 10^{-4}$	$1\cdot234 \times 10^{-5}$	$3\cdot058 \times 10^{-4}$	$6\cdot895 \times 10^{-5}$
Fe	$4\cdot118 \times 10^{-5}$	$7\cdot162 \times 10^{-6}$	$1\cdot561 \times 10^{-5}$	$2\cdot631 \times 10^{-5}$
Al^{3+}	$1\cdot731 \times 10^{-4}$	$3\cdot706 \times 10^{-7}$	$1\cdot931 \times 10^{-6}$	$2\cdot721 \times 10^{-6}$
Zn^{2+}			$3\cdot370 \times 10^{-7}$	$4\cdot402 \times 10^{-7}$
U			$8\cdot132 \times 10^{-8}$	$1\cdot354 \times 10^{-7}$
Th			$8\cdot862 \times 10^{-8}$	$1\cdot150 \times 10^{-7}$
Pb^{2+}			$3\cdot517 \times 10^{-8}$	$6\cdot008 \times 10^{-8}$
Ni^{2+}			$3\cdot087 \times 10^{-7}$	$3\cdot409 \times 10^{-7}$
Mn^{2+}			$1\cdot000 \times 10^{-7}$	$1\cdot732 \times 10^{-7}$
Cd^{2+}			$4\cdot326 \times 10^{-8}$	$7\cdot339 \times 10^{-8}$
As			$7\cdot071 \times 10^{-8}$	$7\cdot575 \times 10^{-8}$
pH:	$6\cdot45$	-	$6\cdot80$	$0\cdot53$
Eh / mV:	$59\cdot7$	-	$96\cdot00$	$14\cdot03$

A critical review of the selected data given in Table A8 shows, that for most of the major anions and cations the analytical values have reasonable standard deviations (if available). Remarkable exceptions are the content of phosphate, cadmium, arsenic, aluminium and the heavy metals.

The overall contents of cadmium, arsenic and the heavy metals are low enough to be neglected. Iron and aluminium are mostly present in colloidal form (which renders an exact analysis difficult).

The values for the redox potential (also supported by comparatively high concentrations of ammonia) imply reducing conditions, which is normally not to be expected from water with free contact to atmosphere. A reason for this may be the leaching and subsequently degradation of organic matter from inside the trenches. As is pointed out in Section 3.3.1.c such organic matter indeed comprises a large part of the waste stored at the Drigg site. If the redox potential is really that low, it would drastically influence the chemical speciation of the redox sensitive contaminants plutonium and (to a much lesser degree) americium. However, computation of E_h from the concentrations for the redox pairs NO_3^-/NH_4^+ and NO_2^-/NH_4^+ gave mean redox potentials of 424 mV and 394 mV, respectively, indicating a less reducing environment.

In general, the aqueous phases both of the drainage channel and of the Drigg stream are dominated by chlorides and (bi)carbonates of sodium and calcium. The ionic strength is 0·0099 for the drainage channel and 0·0067 for the Drigg stream (dilution by water from the East-West stream). The pH is very close to neutral conditions. The average temperature for the two compartments corresponding to the measurements was determined to be 7·0°C.

A5   References

Murdock R.N. (1992) The transport, behaviour and fate of radionuclides in the freshwater environment. PhD thesis, University of Liverpool.

Rudge S.A. (1989) The biological transport of radionuclides in grassland and freshwater ecosystems. PhD thesis, University of Liverpool.

Appendix B   Calculation of the inventory and leachate for the Drigg trenches

B1   Radionuclide inventories of trenches

Smith *et al.* (1988) reported the individual volumes for Trenches 1-7. The total volumes of the seven trenches was $5 \cdot 5 \times 10^5$ m^3.

The inventories for ^{137}Cs, $^{239+240}$Pu and ^{241}Am in Trenches 1-6 were also reported (Smith *et al.*, 1988), i.e. 2·1 TBq ^{137}Cs; 1·6 TBq $^{239+240}$Pu; and 0·66 TBq ^{241}Am.

It was assumed in these calculations that Trench 7 received the same waste as Trench 6 and also filled by the same practice. Trench 6 had the following inventory 0·2 TBq ^{137}Cs; 0·03 TBq $^{239+240}$Pu; and 6×10^{-3} TBq ^{241}Am.

The ratio of volumes between Trenches 6 and 7 was approximately 7:10.

Therefore, the inventories for all seven trenches could be calculated as follows:

$$^{137}\text{Cs} \qquad = 2 \cdot 1 + (0 \cdot 2 \times 10/7) \qquad = 2 \cdot 4 \text{ TBq}$$

$$^{239+240}\text{Pu} \qquad = 1 \cdot 6 + (0 \cdot 03 \times 10/7) \qquad = 1 \cdot 6 \text{ TBq}$$

$$^{241}\text{Am} \qquad = 0 \cdot 66 + (6 \times 10^{-3} \times 10/7) \qquad = 0 \cdot 7 \text{ TBq}$$

$$^{234,235,238}\text{U} \qquad = 41 \cdot 3 \text{ TBq}$$

These values compare well with the ranges Smith *et al.* (1988) reported as BNFL data:

$$^{137}\text{Cs} = 0 \cdot 2 \text{ - } 20 \text{ TBq}$$

$$^{239+240}\text{Pu} = 0 \cdot 8 \text{ - } 4 \text{ TBq}$$

$$^{241}\text{Am} = 0 \cdot 03 \text{ - } 1 \text{ TBq}$$

It should be noted that the half-life of casium-137 is approximately 28 years. Consequently, over period of 40 years, the a significant amount of the casium-137 activity will have decayed. However, as a first approximation, for the purposes of these calculations, this factor was not taken into account.

B2   Radionuclides in leachates

The data used for these calculations was obtained from Murdock (1992). It was assumed that the sample taken at Sampling Point 15 was the most relevant to this study (see Figure B1).
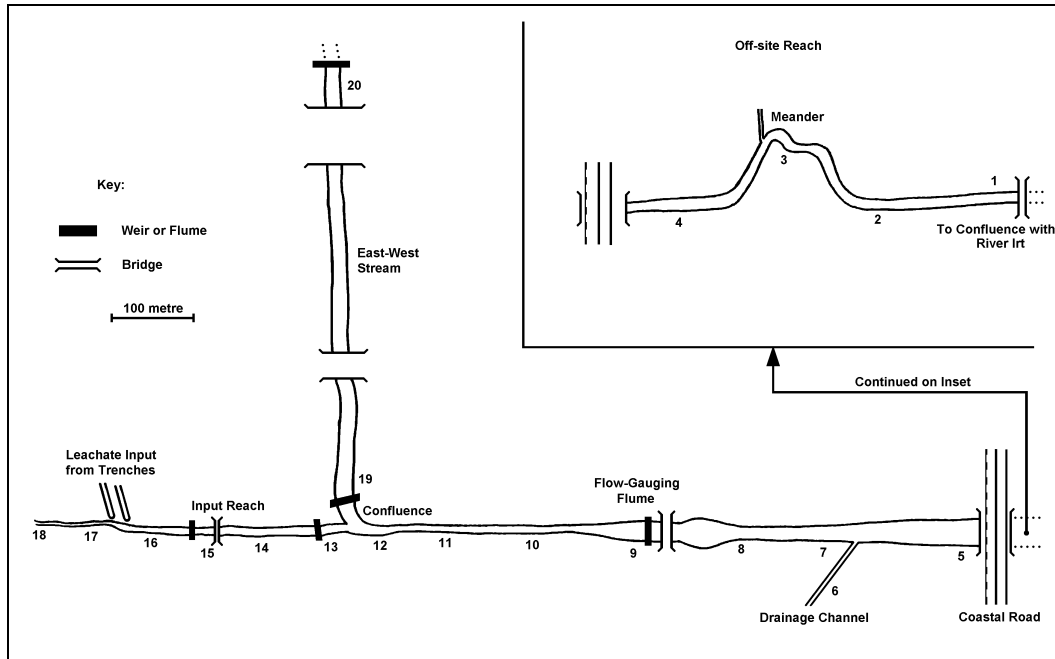
Sampling points described by Murdock (1992) are shown.

Figure B1   The Drigg stream.

The contributions from the trenches were calculated from the change in concentration caused by the introduction of the marine outlet.

The mean marine discharge data for 1990-1992 was $8 \cdot 2 \times 10^4$ m^3 a^{-1}.

The flow rate of leachate was taken to be the amount of rainfall passing through the trenches. This was calculated from the following: a residual rainfall rate of $0 \cdot 388$ m, a trench surface area of 10^5 m^2, an assumed cap efficiency of 90%. Water entering from the sides of the trench was taken to be $0 \cdot 16 \times 10^4$ m^3 a^{-1} (Smith *et al.*, 1988). The water passing through the trenches was assumed to be:

$$0 \cdot 16 \times 10^4 \ + \ 0 \cdot 388 \times 10^5 \times (100 - 90 \, / \, 100) = 0 \cdot 548 \times 10^4 \text{ m}^3 \text{ a}^{-1}.$$

B2.1 ^{137}Cs in leachate

From Murdock (1992)

Mean activity:	before	$= 3475$ Bq m^{-3}.
	after	$= 240$ Bq m^{-3}.

Total discharge	before	$= 8 \cdot 2 \times 10^4 \times 3 \cdot 475 \times 10^3$ Bq a^{-1}.
		$= 28 \cdot 495 \times 10^7$ Bq a^{-1}.
	after	$= (8 \cdot 2 - 0 \cdot 548) \times 10^4 \times 0 \cdot 24 \times 10^3$ Bq a^{-1}.
		$= 1 \cdot 836 \times 10^7$ Bq a^{-1}.
Activity from trenches		$= (28 \cdot 495 - 1 \cdot 836) \times 10^7$ Bq a^{-1}.
		$= 26 \cdot 659 \times 10^7$ Bq a^{-1}.
Concentration in leachate		$= (26 \cdot 659 \times 10^7) \, / \, (0 \cdot 548 \times 10^4)$ Bq m^{-3}.
		$= 48 \cdot 6$ kBq m^{-3}.

B2.2 $^{239+240}$Pu in leachate

From Murdock (1992)

Mean activity:	before	$= 620 \cdot 83$ Bq m^{-3}.
	after	$= 38 \cdot 5$ Bq m^{-3}.

Total discharge	before	$= 8 \cdot 2 \times 10^4 \times 0 \cdot 62083 \times 10^3$ Bq a^{-1}.
		$= 5 \cdot 090806 \times 10^7$ Bq a^{-1}.
	after	$= (8 \cdot 2 - 0 \cdot 548) \times 10^4 \times 0 \cdot 0385 \times 10^3$ Bq a^{-1}.
		$= 0 \cdot 2946 \times 10^7$ Bq a^{-1}.
Activity from trenches		$= (5 \cdot 090806 - 0 \cdot 2946) \times 10^7$ Bq a^{-1}.
		$= 4 \cdot 796 \times 10^7$ Bq a^{-1}.
Concentration in leachate		$= (4 \cdot 796 \times 10^7) / (0 \cdot 548 \times 10^4)$ Bq m^{-3}.
		$= 8 \cdot 6$ kBq m^{-3}.

B2.3 ^{241}Am in leachate

From Murdock (1992)

Mean activity:	before	$= 454 \cdot 5$ Bq m^{-3}.
	after	$= 20 \cdot 5$ Bq m^{-3}.

Total discharge	before	$= 8 \cdot 2 \times 10^4 \times 0 \cdot 4545 \times 10^3$ Bq a^{-1}.
		$= 3 \cdot 7269 \times 10^7$ Bq a^{-1}.
	after	$= (8 \cdot 2 - 0 \cdot 548) \times 10^4 \times 0 \cdot 0205 \times 10^3$ Bq a^{-1}.
		$= 0 \cdot 1569 \times 10^7$ Bq a^{-1}.
Activity from trenches		$= (3 \cdot 7269 - 0 \cdot 1569) \times 10^7$ Bq a^{-1}.
		$= 3 \cdot 570 \times 10^7$ Bq a^{-1}.
Concentration in leachate		$= (3 \cdot 570 \times 10^7) / (0 \cdot 548 \times 10^4)$ Bq m^{-3}.
		$= 6 \cdot 5$ kBq m^{-3}.

B2.4 ^{234,235,238}U in leachate

Assuming that the chemistry of uranium is similar to that of americium then the ratio of concentrations in the leachate should be similar to that in the Drigg stream. This americium and uranium content of the Drigg stream are known (BNFL, 1989-1991). The mean ratio of concentrations ratio for 1988-1990 is 4·33. The americium ratio in the leachate has been calculated (B2.3). Consequently, the concentration of uranium in the leachate will be:

$$6 \cdot 5 \times 10^3 \times 4 \cdot 33 = 28 \cdot 1 \times 10^3 \text{ kBq m}^{-3}.$$

The annual discharge of uranium will therefore be:

$$28 \cdot 1 \times 10^3 \times 0 \cdot 33 \times 10^4 = 9 \cdot 3 \times 10^7 \text{ Bq a}^{-1}.$$

B3   References

BNFL (1989) Annual Report on Radioactive Discharges and Monitoring of the Environment, 1988. Volume 1. BNFL, Risley.

BNFL (1990) Annual Report on Radioactive Discharges and Monitoring of the Environment, 1989. Volume 1. BNFL, Risley.

BNFL (1991) Annual Report on Radioactive Discharges and Monitoring of the Environment, 1990. Volume 1. BNFL, Risley.

Murdock R.N. (1992) The Transport, Behaviour and fate of Radionuclides in the Freshwater Environment. PhD thesis, Liverpool.

Smith G.M., Fearn H.S., Smith K.R., Davis J.P. and Klos R. (1988) Assessment of the Radiological Impact of Disposal of Solid Radioactive waste at Drigg. National Radiological Protection Board - M148, Oxon.

Appendix C   Calculations of the volumes of waste and residual activity fractions after remediation at the Drigg site

C1   Dimensions of site

The total radionuclide inventory for Drigg Trenches 1-7 may be calculated as the sum of the radionuclide inventories given in Appendix B. This gives a value of 46 TBq.

The Drigg stream, which represents the major discharge of water from the Drigg site, contains typically 40 g m^{-3} of suspended solid (Murdock, 1992). The flow rate of the Drigg stream is of the order of 1×10^6 m^3 a^{-1}.

C2   Volume of waste and residual activity arising from remediation process

The remediation technologies falls into distinct categories: those which treat solutions and those which treat solids.

Water treatment will be an on-going process. For the purposes of these calculations they are assumed to last 500 years.

Using data from Zeevaert and Bousher (1998) in TD3+4 the specific performance data for remediation techniques which are appropriate to the Drigg site may be estimated. These are summarised in Table C1. This data may be used to estimate the volume of waste generated through remediation of the site and the fraction of activity remaining on-site after remediation. These are summarised in Table C2. The calculations used to arrive at these values are described in the following sections.

C2.1   Filtration

A process from treating liquid discharges.

It is assumed that the remediation process will remove contaminated suspended solids from the Drigg stream water. Consequently, if filtration is 100% efficient at removing solids, the process will remove $40 \times 1 \times 10^6 = 4 \times 10^7$ g a$^{-1} = 4 \times 10^4$ kg a^{-1}.

Assume that the density of the removed sediment is 1600 kg m^{-3}. Then the volume of waste will be $4 \times 10^4/1600 = 25$ m^3 a^{-1}.

If the process continues for 500 years then,

$$\text{Volume of waste} \qquad = \qquad 25 \times 500 \qquad = \qquad 12500 \text{ m}^3$$

The decontamination factor is radionuclide-specific for this technique. However, a value of 100 is used for most of the important radionuclides present at the site. This factor is used for the purposes of this calculation.

Consequently,

$$\text{Residual activity fraction} \qquad = \qquad 1 / 100 = \qquad 0{\cdot}01$$

C2.2   Chemical solubilisation

This form of remediation will be applied to the solid waste.

The extraction and processing of the contaminated waste is readily calculated. However, there is little information concerning the fate of the solubilised radionuclides. In all probability it would be treated further to reduce the volume. Here it is assumed that the liquid waste will be stored in containers off-site. Any additional treatment will thus occur off-site.

A further problem is that there is no defined concentration of radionuclides in the liquid arising from the separation. For the purposes of this calculation an arbitrary concentration of 1 GBq m^{-3} has been assumed.

If the decontamination factor is 10, then there will be $46 \times 10^{12} \times (9/10) = 4 \cdot 1 \times 10^{13}$ Bq in the liquid waste.

Consequently,

Volume of waste	=	$4 \cdot 1 \times 10^{13} / 1 \times 10^9$	=	$4 \cdot 1 \times 10^4$ m^3
Residual activity fraction	=	$1 / 10$	=	$0 \cdot 1$

C2.3   Ion exchange

This is a process for treating liquids. It involves two steps. A filtration step to remove solid material followed by treatment using an ion exchange column. The solid waste produced by the filtration step will be very similar to that described for remediation by filtration (see Section C2.1). The same calculation is followed here. It is reasonable to assume that this solid would have to be disposed of.

The ion exchange step is assumed to use a column which can be regenerated after its capacity has been reached. This gives rise to a liquid waste. However, the volume of waste produced by this part of the process will be negligible in comparison to the solid waste (produced using filtration) and is ignored. Therefore, only the volume of solid waste is considered by this calculation.

Consequently,

Volume of waste   =   12500 m^3 (see Section C2.1)

The decontamination factor is radionuclide-specific for this technique. However, a value of 5 is used for most of the important radionuclides present at the site. This factor is used for the purposes of this calculation.

Consequently,

Residual activity fraction   = $1 / 5$ = $0 \cdot 2$

C2.4   Biosorption

Biosorption is used to remove radionuclides from liquid wastes.

It would appear reasonable that, like ion exchange, a filtration step would be prerequisite for this process. Also the concentrations of radionuclides are extremely low. Any process which concentrates these will produce very small quantities of waste. Consequently, this is taken to be negligible in comparison to the solid waste produced during the filtration stage. The volume of solid waste produced by the filtration step will be very similar to that described for remediation by filtration (see Section C2.1). Hence, the calculation of the cost for remediation and waste disposal will be very similar.

Consequently,

Volume of waste   =   12500 m^3 (see Section C2.1)

A common decontamination factor of 10 is used for all radionuclides.

Consequently,

Residual activity fraction   = $1 / 5$ = $0 \cdot 1$

C2.5   Capping

This is appropriate for remediating solid material such as that contained in the trenches. The calculations are very straightforward.

No waste is created by the process and all activity remains on-site.

Consequently,

Volume of waste	=	0
Residual activity fraction	=	1

C2.6   Subsurface barrier

This is appropriate for remediating solid materials, such as that contained in the trenches.

No waste is generated by this type of remediation as the activity remains on-site.

Consequently,

Volume of waste	=	0
Residual activity fraction	=	1

C2.7   Physical immobilisation (*ex-situ*)

This remediation technique is applied to the treatment of solid materials.

No waste is generated by this approach and all activity remains on the site.

Therefore,

Volume of waste	=	0
Residual activity fraction	=	1

C2.8   Physical immobilisation (i*n-situ*)

This remediation approach is for treating solid materials.

No waste is generated by this approach and all activity remains on the site.

Therefore,

Volume of waste	=	0
Residual activity fraction	=	1

C2.9   Chemical immobilisation (e*x-situ*)

This techniques is applied to the treatment of solids.

No waste is generated by this approach and all activity remains on the site.

Therefore,

Volume of waste	=	0
Residual activity fraction	=	1

C2.10   Chemical immobilisation (i*n-situ*)

This techniques is applicable to treating solid materials.

No waste is generated by this approach and all activity remains on the site.

Therefore,

Volume of waste   =   0

Residual activity fraction   =   1

C3   Reference

Murdock R.N. (1992) The transport, behaviour and fate of radionuclides in the freshwater environment. PhD thesis, University of Liverpool.

Zeevaert T. and Bousher A. (1998) Restoration techniques: characteristics and performances. RESTRAT Technical Deliverable TD3+4.

Table C1   The performance of remediation techniques appropriate to the Drigg site.

Remediation Technique	Unit[†]	Value
Physical separation (*ex-situ*) Filtration	DF	7 (Cs) 100 (U, Am, Pu)

Chemical Separation (*ex-situ*) (including solvent extraction)
    Chemical solubilisation
 	
    Ion exchange (liquids)
	DF
	
	DF
	DF
		10
		
		33 (Cs)
		5 (U, Am, Pu)

Biological Separation Biosorption (liquids)	DF	10

Containment
    Capping
 	
    Subsurface barriers
	k
	
	k
		1×10^{-10} m s^{-1}
		
		1×10^{-9} m s^{-1}

Physical Immobilisation	MRF	10
Chemical Immobilisation	MRF	10

[†] *DF = decontamination factor; k = permeability coefficient; MRF = mobility reduction factor.*

Table C2   **The volume of waste produced and fraction of activity remaining after remediation of the Drigg site.**

Remediation strategy	Volume of waste removed from site m^3	Fraction of activity remaining after remediation
Filtration	0	1
Chemical Solubilisation	12500	0.01
Ion Exchange	41000	0.1
Biosorption	12500	0.2
Capping	12500	0.1
Subsurface Barrier	0	1
Physical Immobilisation (*ex-situ*)	0	1
Physical Immobilisation (*in-situ*)	0	1
Chemical Immobilisation (*ex-situ*)	0	1
Chemical Immobilisation (*in-situ*)	0	1

Appendix D   Cost calculations for the Drigg site

D1   Dimensions of site

The volume of waste in Drigg Trenches 1-7 is 5.5×10^5 m^{-3} and the surface area is 1×10^5 m^{-2}. The estimated length of the trench perimeter is 1·8 km. The total radionuclide content of the trenches is 46 TBq.

The Drigg stream, which represents the major discharge of water from the Drigg site, contains typically 40 g m^{-3} of suspended solid (Murdock, 1992). The flow rate of the Drigg stream is of the order of 1×10^6 m^3 a^{-1}.

D2   Cost of remediation process

The remediation technologies falls into distinct categories: those which treat solutions and those which treat solids.

Water treatment will be an on-going process. For the purposes of these calculations they are assumed to last 500 years.

Using the specific performance data given in Appendix C and the costs of remediation techniques given by Zeevaert and Bousher (1998) in TD3+4 it is possible to determine the remediation, waste disposal and monitoring costs which are specific to the Drigg site. The data from these sources are summarised in Tables D1 and D2. The costs arising from remediation of the site and disposing of the waste are summarised in Table D3. The calculations used to arrive at these values are described in the following sections.

Table D3 also gives estimates for the monitoring costs after remediation. For the purposes of these calculation it has been assumed that monitoring costs are linearly dependent upon the fraction of activity remaining on the site after remediation. In addition, the cost of monitoring the site, if no remediation has taken place, is assumed to be 7.5×10^7 EUR.

D2.1   Filtration

A process from treating liquid discharges.

After 500 years then the total volume of disposable waste will be 12500 m^3. (see Table D2) The total volume of liquid to be remediated will be $500 \times 1 \times 10^6 = 5 \times 10^8$ m^3.

The fraction of activity remaining on-site is 0·01 (see Table D2)

Processing costs are 0·75 EUR m^{-3} (liquid) (see Table D1).

Transport and disposal costs of waste is 2500 EUR m^{-3} (liquid) (see Table D1).

Consequently,

Remediation cost	=	$0.75 \times 5 \times 10^8$	=	3.8×10^8 EUR
Waste disposal cost	=	12500×2500	=	3.1×10^7 EUR
Monitoring cost	=	$0.01 \times 7.5 \times 10^7$	=	7.5×10^5 EUR

D2.2   Chemical solubilisation

This form of remediation will be applied to the solid waste.

Excavation, transport and treatment costs are 550 EUR m^{-3} (see Table D1).

There will be 4.1×10^4 m^3 of waste solution (see Table D2).

The fraction of activity remaining on-site will be 0·1 (see Table D2).

Disposal costs of the waste solution will be 2500 EUR m^{-3} (assumed to be the same as that for solid waste).

Therefore,

Remediation cost	=	$550 \times 5.5 \times 10^5$	=	3.0×10^8 EUR
Waste disposal cost	=	$2500 \times 4.1 \times 10^4$	=	1.0×10^8 EUR
Monitoring cost	=	$0.1 \times 7.5 \times 10^7$	=	7.5×10^6 EUR

D2.3 Ion exchange

This is a process for treating liquids. It involves two steps. A filtration step to remove solid material followed by treatment using an ion exchange column. The solid waste produced by the filtration step will be very similar to that described for remediation by filtration (see Section D2.1). The same calculation is followed here. It is reasonable to assume that this solid would have to be disposed of.

The ion exchange step is assumed to use a column which can be regenerated after its capacity has been reached. This gives rise to a liquid waste. However, the volume of waste produced by this part of the process will be negligible in comparison to the solid waste (produced using filtration) and is ignored. Therefore, only the volume of solid waste is considered by this calculation.

The volume of liquid to be treated will be 5×10^8 m^3 and the total volume of waste produced by the process will be 12500 m^3 (see Section D2.1).

The fraction of activity remaining after remediation will be 0·2 (see Table D2).

Remediation costs are 2 EUR m^{-3} (liquid) (see Table D1).

Disposal costs of the wastes will be 2500 EUR m^{-3} (see Table D1).

Consequently,

Remediation cost	=	$2 \times 5 \times 10^8$	=	1.0×10^9 EUR
Waste disposal cost	=	12500×2500	=	3.1×10^7 EUR
Monitoring cost	=	$0.2 \times 7.5 \times 10^7$	=	1.5×10^7 EUR

D2.4 Biosorption

Biosorption is used to remove radionuclides from liquid wastes.

It would appear reasonable that, like ion exchange, a filtration step would be prerequisite for this process. Also the concentrations of radionuclides are extremely low. Any process which concentrates these will produce very small quantities of waste. Consequently, this is taken to be negligible in comparison to the solid waste produced during the filtration stage. The volume of solid waste produced by the filtration step will be very similar to that described for remediation by filtration (see Section D2.1). Hence, the calculation of the cost for remediation and waste disposal will be very similar.

Consequently, the volume of liquid to be treated will be 5×10^8 m^3 and the total volume of waste produced by the process will be 12500 m^3 (see Table D2).

The fraction of activity remaining after remediation will be 0·1 (see Table D2).

Remediation costs are 2·5 EUR m^{-3} (liquid) (see Table D1).

Disposal costs of the waste solid will be 2500 EUR m^{-3} (solid) (see Table D1).

Therefore,

Remediation cost	=	$2\cdot5 \times 5 \times 10^8$	=	$1\cdot3 \times 10^9$ EUR
Waste disposal cost	=	12500×2500	=	$3\cdot1 \times 10^7$ EUR
Monitoring cost	=	$0\cdot1 \times 7\cdot5 \times 10^7$	=	$7\cdot5 \times 10^6$ EUR

D2.5   Capping

This is appropriate for remediating solid material such as that contained in the trenches.

Remediation costs are 35 EUR m^{-2} (see Table D1).

No waste is created by the process. Therefore, there are no disposal costs. However, the fraction of residual activity will 1 (see Table D2).

Consequently,

Remediation cost	=	$35 \times 1 \times 10^5$	=	$3\cdot5 \times 10^6$ EUR
Waste disposal cost	=	0 EUR		
Monitoring cost	=	$1 \times 7\cdot5 \times 10^7$	=	$7\cdot5 \times 10^7$ EUR

D2.6   Subsurface barrier

This is appropriate for remediating solid materials, such as that contained in the trenches.

A number of assumptions have to be made.

- A grout curtain represents an effective barrier for this waste.

- The perimeter is estimated to be 1·8 km (see Section D1).

- The barriers are used on all sides of the trench.

The typical depth of the grout curtain is taken to be about twice the typical depth of the trenches. The depth is calculated from the volume of the trenches divided by the surface area, i.e. $5\cdot5 \times 10^5/1 \times 10^5 \approx 5$ m. Consequently, the grout curtain depth will be 10 m.

No waste is generated by this type of remediation. However, the fraction of residual activity will 1 (see Table D2).

Remediation costs are 350 EUR m^{-2} (barrier) (see Table D1). The surface area of the barrier is calculated from its length of the trench perimeter and the depth of the barrier.

No waste is created by the process. Therefore, there are no disposal costs.

Consequently,

Remediation cost	=	$350 \times 1800 \times 10$	=	$6\cdot3 \times 10^6$ EUR
Waste disposal cost	=	0 EUR		
Monitoring cost	=	$1 \times 7\cdot5 \times 10^7$	=	$7\cdot5 \times 10^7$ EUR

D2.7   Physical immobilisation (*ex-situ*)

This remediation technique is applied to the treatment of solid materials.

The calculation is straightforward as the volume of waste is known.

Immobilisation costs are 35 EUR m^{-3} (see Table D1). The excavation and transport costs are 65 EUR m^{-3} (see Table D1).

No waste is generated by this approach. However, the fraction of residual activity will 1 (see Table D2).

Therefore,

Remediation cost	=	$(35 + 65) \times 5 \cdot 5 \times 10^5$	=	$5 \cdot 5 \times 10^7$ EUR
Waste disposal cost	=	0 EUR		
Monitoring cost	=	$1 \times 7 \cdot 5 \times 10^7$	=	$7 \cdot 5 \times 10^7$ EUR

D2.8   Physical immobilisation (i*n-situ*)

This remediation approach is for treating solid materials.

The calculation is straightforward as the volume of waste is known.

Immobilisation costs are 350 EUR m^{-3} (see Table D1). No waste is generated by this approach. However, the fraction of residual activity will 1 (see Table D2).

Therefore,

Remediation cost	=	$350 \times 5 \cdot 5 \times 10^5$	=	$1 \cdot 9 \times 10^8$ EUR
Waste disposal cost	=	0 EUR		
Monitoring cost	=	$1 \times 7 \cdot 5 \times 10^7$	=	$7 \cdot 5 \times 10^7$ EUR

D2.9   Chemical immobilisation (e*x-situ*)

This techniques is applied to the treatment of solids.

The calculation is straightforward and very similar to that for *ex-situ* physical immobilisation (see Section D2.7).

Immobilisation costs are 180 EUR m^{-3}. The excavation and transport costs are 65 EUR m^{-3} (see Table D1).

No waste is generated by this approach. However, the fraction of residual activity will 1 (see Table D2).

Consequently,

Remediation cost	=	$(180 + 65) \times 5 \cdot 5 \times 10^5$	=	$1 \cdot 3 \times 10^8$ EUR
Waste disposal cost	=	0 EUR		
Monitoring cost	=	$1 \times 7 \cdot 5 \times 10^7$	=	$7 \cdot 5 \times 10^7$ EUR

D2.10   Chemical immobilisation (i*n-situ*)

This techniques is applicable to treating solid materials.

The calculation is straightforward and very similar to that for *in-situ* physical immobilisation (see Section D2.8).

Immobilisation costs are 100 EUR m^{-3} (see Table D1).

No waste is generated by this approach. However, the fraction of residual activity will 1 (see Table D2).

Therefore,

Remediation cost	=	$100 \times 5 \cdot 5 \times 10^5$	=	$5 \cdot 5 \times 10^7$ EUR
Waste disposal cost	=	0 EUR		

Monitoring cost $\quad=\quad 1 \times 7 \cdot 5 \times 10^7 \quad=\quad 7 \cdot 5 \times 10^7$ EUR

D3   Reference

Murdock R.N. (1992) The transport, behaviour and fate of radionuclides in the freshwater environment. PhD thesis, University of Liverpool.

Zeevaert T. and Bousher A. (1998) Restoration techniques: characteristics and performances. RESTRAT Technical Deliverable TD3+4.

Table D1   The costs of remediation techniques appropriate to the Drigg site.

Remediation Technique	Cost
Physical Separation (*ex-situ*) Filtration (liquids)	0·75 EUR m^{-3} (liquid)

Chemical Separation (*ex-situ*)
        Chemical solubilisation (including solvent extraction)
            Separation costs
            Cost of excavation and transport (prior to
            solubilisation)
                    NB This does not include the cost of disposal
 	
        Ion exchange (liquids)
	
	550 EUR m^{-3}
	65 EUR m^{-3}
	
	2 EUR m^{-3}
	(liquid)

Biological Separation Biosorption (liquids)	2·5 EUR m^{-3} (liquid)

Containment
        Capping
 	
        Subsurface Barriers
 	
                Slurry walls
 	
                Grout curtains
	35 EUR m^{-2}
	(surf. area)
	
	550 EUR m^{-2}
	(barrier)
	350 EUR m^{-2}
	(barrier)

Physical Immobilisation
        Ex-situ
            Cost of excavation and transport (prior to
            immobilisation)
 	
        In-situ
	35 EUR m^{-3}
	65 EUR m^{-3}
	
	350 EUR m^{-3}

Chemical Immobilisation
        Ex-situ
            Cost of excavation and transport (prior to
            immobilisation)
 	
        In-situ
	180 EUR m^{-3}
	65 EUR m^{-3}
	
	100 EUR m^{-3}

Table D2 **The volume of waste produced and fraction of activity remaining after remediation of the Drigg site.**

Remediation strategy	Volume of waste removed from site m³	Fraction of activity remaining after remediation
Filtration	0	1
Chemical Solubilisation	12500	0.01
Ion Exchange	41000	0.1
Biosorption	12500	0.2
Capping	12500	0.1
Subsurface Barrier	0	1
Physical Immobilisation (*ex-situ*)	0	1
Physical Immobilisation (*in-situ*)	0	1
Chemical Immobilisation (*ex-situ*)	0	1
Chemical Immobilisation (*in-situ*)	0	1

Data from Appendix C

Table D3 **The cost of applying remediation technologies to the Drigg site.**

Remediation strategy	Monetary costs of remediation [EUR]		
	Remediation (incl. labour)	Waste disposal (incl. transport)	Monitoring costs
Filtration	$3 \cdot 8 \times 10^8$	$3 \cdot 1 \times 10^7$	$7 \cdot 5 \times 10^5$
Chemical Solubilisation	$3 \cdot 0 \times 10^8$	$1 \cdot 0 \times 10^8$	$7 \cdot 5 \times 10^6$
Ion Exchange	$1 \cdot 0 \times 10^8$	$3 \cdot 1 \times 10^7$	$1 \cdot 5 \times 10^7$
Biosorption	$1 \cdot 9 \times 10^9$	$3 \cdot 1 \times 10^7$	$7 \cdot 5 \times 10^6$
Capping	$3 \cdot 5 \times 10^6$	0	$7 \cdot 5 \times 10^7$
Subsurface Barrier	$6 \cdot 3 \times 10^6$	0	$7 \cdot 5 \times 10^7$
Physical Immobilisation (*ex-situ*)	$5 \cdot 5 \times 10^7$	0	$7 \cdot 5 \times 10^7$
Physical Immobilisation (*in-situ*)	$1 \cdot 9 \times 10^8$	0	$7 \cdot 5 \times 10^7$
Chemical Immobilisation (*ex-situ*)	$1 \cdot 3 \times 10^8$	0	$7 \cdot 5 \times 10^7$
Chemical Immobilisation (*in-situ*)	$5 \cdot 5 \times 10^7$	0	$7 \cdot 5 \times 10^7$