Investigation of the Sources and Fate of Radioactive Discharges to Public Sewers

R&D Technical Report P288

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This report presents the final findings of research into the sources and fates of radioactive discharges to public sewers. This information will assist the Environment Agency in its regulation of the radioactive discharges from non-nuclear sites such as hospitals, universities and research centres. It will also contribute to assessing the integrated radiological impact of radioactive discharges from multiple sites.

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EXECUTIVE SUMMARY

The practise of disposing of small amounts of liquid radioactive waste from non-nuclear organisations to public sewer systems, is currently being reviewed by the Environment Agency, Scottish Environmental Protection Agency and the Industrial Pollution and Radiochemical Inspectorate. Changing sewage treatment and disposal practises, particularly the increasing use of incineration to treat sewage sludge in urban areas, have prompted this review. This study was undertaken as part of the review process to determine whether disposal into the public sewer system remains a safe disposal option.

This report describes a study to investigate the sources and fates of authorised discharges of radionuclides into the sewer system. Sewage transport, treatment, discharge and incineration systems are described and the fate of radionuclides in the systems considered. A model (SMART) was developed to predict radionuclide behaviour through these systems and calculate the resulting doses. Case studies of two large urban sewerage systems were carried out in Leeds and Central London. Leeds is served by one sewage treatment works at Knostrop, which receives disposals of ten radionuclides from six disposers. Up to seven of the radionuclides were measurable in effluents, sludges or incinerator ash at the works. Naturally occurring and fallout radionuclides were also detected. Central London is served by one large works at Beckton which receives disposals of twenty radionuclides from 65 disposers. Up to ten of the disposed radionuclides, plus naturally occurring and fallout radionuclides were detected in effluents, sludges, incinerator cake or ash at Beckton. For both sites the largest authorised disposals were of Tc-99m, C-14, I-125, I-131 and H-3. The behaviour of radionuclides during incineration was assessed using measurements and literature values.

The radiological implications of disposals in Leeds and Central London were assessed using the model SMART and by measuring dose rates. Doses were predicted for discharges at the authorised limits and for typical disposals made in 1997. All the predicted and measured doses were below the dose limit and maximum dose constraint for members of the public. Actual discharge levels are less than authorised limits and consequently the actual doses to members of the public will be less than the values quoted below. Estimated doses from disposals at maximum authorised limits in Leeds were 80 μSv y^{-1} to typical workers around the sewage works and incinerator, and up to 238 μSv y^{-1} in the sludge pressroom. The most important radionuclide was I-131. Estimated doses from disposals at authorised limits in Beckton were 210 μSv/y to typical workers and 207 μSv y^{-1} to sludge press workers. These doses were dominated by disposals of I-131 and Rb-84. Maximum estimated doses to the public from disposal of effluent and incineration of sewage sludge at Beckton were 30 μSv y^{-1} and 2 μSv y^{-1} respectively and in Leeds were 180 μSv y^{-1} and 2 μSv y^{-1} respectively. The most important radionuclides were C-14, I-125 and I-131 at both sites. Estimated doses based on typical disposals in both catchments were up to a factor of three lower at the sewage treatment works and up to an order of magnitude lower for the public. Dose rates were measured at various points in each sewage works, and were found to be below the uncertainty on the results.

The applicability of the results for Beckton and Knostrop to other sewage treatment works was addressed and simple formulae given for the important pathways.

The study also looked at the need for further data on radionuclide behaviour in incinerators and the need for discharge reduction measures. Neither can be justified on the basis of this study.

The introduction of radionuclide specific limits for the more significant radionuclides is suggested. The study concludes that the practice of disposing radionuclides into the sewer system is a safe disposal option and is acceptable in the urban situations studied.

KEYWORDS

Radionuclides, assessment, disposal, sewers.

1 INTRODUCTION

1.1 Background

Authorised discharges of small amounts of radioactive liquid wastes to sewer systems from non-nuclear organisations are permitted in the UK. Certificates of Authorisation to dispose of radioactive, liquid waste are issued by the Agencies (Environment Agency, Scottish Environmental Protection Agency and the Industrial Pollution and Radiochemical Inspectorate) under the Radioactive Substances Act 1993. This requires that the radiological risks are small and the disposal route is considered the best option. The Agencies are currently reviewing the continued acceptability of these disposal practices especially in major urban areas, given the changes in treatment and disposal practices for sewage itself. In particular, since the end of 1998, large sewage works in urban areas of the UK have been unable to dispose of sewage sludges to the North Sea. The incineration of sewage sludge is therefore likely to become increasingly important in the future. As part of their review the Environment Agency asked the National Radiological Protection Board (NRPB) to undertake this study. The project has been funded in part by the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER).

1.2 Objectives

The objective of this study is to provide the necessary information for the Environment Agency to review the acceptability of authorised discharges of small amounts of liquid radioactive wastes to sewer systems.

To meet the Environment Agency objective the study has specific aims:

- To review current and future sewage treatment practices.
- To investigate the practice of discharging small amounts of liquid radioactive waste to public sewers to determine whether it remains a safe disposal option, especially in major urban areas. A programme of monitoring and radiological assessment was undertaken at two urban sewage treatment works (STW), one in London and one in Leeds.
- To examine the practice of sewage sludge incineration, the fate of radionuclides during incineration and the radiological consequences of incinerator operation.
- To consider any dose reduction options which may be required, and to determine whether alternative practices would yield significant radiological benefit. Also, to obtain the opinion of the major disposers on these alternative practices.
- To examine the wider implications of the findings, particularly with respect to non-urban sewage treatment works.

1.3 Structure of Report

The report begins with a description of sewage collection, treatment and disposals in the UK, an indication of what changes are occurring to treatment, and what issues are driving the changes, (Chapter 2). Chapter 3 presents a review of the uses of radionuclides that are commonly disposed to sewage systems, and describes how they are likely to behave during sewage treatment. The model that was developed to assess the radiological impact of the disposal of radionuclides to sewers is described in Chapter 4.

Chapters 5 to 8 describe the case study of Beckton STW in East London. This includes a general description, the authorised discharges of radionuclides, the most important disposers, the important radionuclides and the patterns of disposal. An estimate of the resulting radiological impact is presented together with results from summer and winter sampling at the STW.

Chapters 9 to 12 describe the case study of Knostrop STW in Leeds, following the same structure as for Beckton.

The use of incinerators for sewage sludge is addressed in Chapter 13. Radionuclide behaviour in incinerators and the issues surrounding a possible tracer study to investigate this behaviour is discussed.

An analysis of the predicted and measured results for Beckton and Knostrop is presented in Chapter 14. This includes the broader implications of the study, including dose reduction options, the practicalities of decay storage and the results of consultations. Chapter 15 discusses the applicability of the findings to other STW and Chapter 16 contains a summary and the overall conclusions of the report.

Appendices A - G give detailed information on the modelling and monitoring work carried out. Appendix H contains a detailed description of all the major STW in Greater London.

2 REVIEW OF SEWAGE TREATMENT AND DISPOSAL

2.1 Introduction

This study is concerned with the behaviour and fate of radionuclides discharged into waste water. Radionuclides may enter the waste water from authorised discharges by hospitals, universities, research institutes and manufacturing, and are carried with waste water to sewage treatment plants. The waste water (sewage effluent) containing a low suspended solid content passes through some or all of four distinct stages:

- Transit in the sewer system
- Treatment of effluent and separation of solids at sewage works
- Treatment of solid matter (sewage sludges)
- Disposal of sewage sludge or treated liquid effluent.

Radionuclides present in the sewage will be subject to the same processes and the concentrations of radionuclides in the various phases will be controlled by the effect these processes have on radionuclide behaviour. The radionuclide behaviour will affect the radiological impact of each nuclide during transport, treatment and final disposal.

2.2 Overview of Sewage Treatment in the UK

Raw sewage effluent, or wastewater, is predominantly water with a low suspended solids load. Wastewater arises from dwellings, businesses and institutions such as universities and hospitals. The solids in the wastewater comprise faecal matter, paper, sanitary products, food wastes, vegetation, grit and soil. The wastewater may also contain waste chemicals such as detergents, oils and traces of organic and inorganic materials including heavy metals. Some sewer systems also carry surface water run-off, and during storms can carry very large volumes of storm water.

Sewage presents a main environmental management issue. The biochemical oxygen demand (BOD) of untreated sewage effluent prevents disposal of raw sewage directly into many environments. Disposal into the aquatic environments may lead to sustained removal of dissolved oxygen, destroying habitats for aquatic species, and resulting in the release of gases such as hydrogen sulphide, methane etc. In addition, the presence of undesirable bacteria and viruses associated with solids may present a significant health hazard to those downstream of the release point.

Wastewater treatment is aimed at reducing BOD and suspended solids. Waste water is managed by transporting it through engineered water courses to sewage works where it is treated to remove suspended solids, biochemical oxygen demand and in some cases trace pollutants. Sludges (separated solids) may be further treated by anaerobic or thermal digestion or

pasteurisation, depending on the final fate of the material. Sufficient treatment to remove BOD permits effluents to be discharged to inland or coastal waters. Sludges may be disposed to landfill sites, or may be treated and then applied to land as a soil conditioner or may be incinerated. Historically, sludges have also been disposed to coastal waters, although this ceased during 1998.

2.3 Overall Quantities of Sewage Effluent and Sludges Produced in the UK

The total water consumption (indicative of sewage effluent volumes) and total sewage sludge production in England and Wales are presented in Table 1 and Figure 1, and for Scotland and Northern Ireland in Figure 2 for the ten years between 1983 and 1993[1]. The water consumption data includes both metered and unmetered supplies. This quantity is indicative of the quantity of water disposed of in sewers, although it includes water that may leak from pipes or be used in industry and not discharged to sewer. The average water consumption in England and Wales for this period was 6.1 10^9 m^3 y^{-1} and the average production rate of sewage sludge was 9.2 10^5 t y^{-1} (dry solids). As can be seen, both water consumption and sludge production vary year by year, these small changes are probably the result of annual variations in weather (dry years and wet years), and there is no obvious evidence of a general increase in water consumption or sludge production. However, sludge production is expected to increase in the future as effluent treatment standards are increased.

2.4 Transport of Sewage Effluents in the Sewer System in the UK

2.4.1 Introduction

The waste water input into the sewer systems comprises foul water, industrial effluent (which may be pre-treated), and run-off from roads and urban areas. There are generally three kinds of sewer systems used to transport such waste water from domestic and industrial locations. These are combined sewers, separate sewers (one for foul water and other effluents, one for run-off) or a mixture of the two types.

2.4.2 Combined and separate sewer systems

In combined sewers the different types of effluents are carried together in a single sewer (70% of all sewers in the UK are this type, with a higher percentage in old towns and cities[2]). These sewers often date from the last century. Because combined sewers carry all effluents their main drawback is that they are inefficient at handling run-off during storms, and may discharge untreated wastewater into the environment.

Modern sewers are designed to carry foul water and industrial effluents separately to storm run-off. This approach allows the run-off to be discharged directly into rivers, without potentially serious effects. Sewage treatment works may be designed with much less operating headroom. Unfortunately, updating older combined systems requires large capital expenditure and

a lot of disruption. Separate systems are therefore mostly in new towns and recently built suburban areas.

In some cases, a compromise system may be installed to alleviate particular storm water problems. In this case some sewers may be combined and some separate, meaning that only a fraction of run-off is carried to sewage treatment plants.

2.4.3 Construction of sewers

A typical system consists of laterals (building connections), subsidiary sewers and trunk sewers. Sizes can range from 0.1 to 3.6 m in diameter[2], with sewer pipes serving a hospital likely to be 0.15 to 0.2 m in diameter[3]. Trunk sewers may be made of concrete or bricks, with laterals and subsidiary sewers usually being concrete, vitrified clay, asbestos cement or plastic. Sewer pipe surfaces that may corrode or dissolve are likely to have an abrasion-resistant coating, historically tar but more recently epoxy or plastic coatings have been used[2].

2.4.4 Sewer flow rates and transit times

The time taken to transfer radionuclides from the discharge point to the sewage treatment works is important for radionuclides which may decay significantly during transit. The transit time depends on the distance travelled, the reciprocal of the effluent velocity and any delay by sorption on pipe walls.

The sewer length from discharge point to works is obviously variable. However, in an urban area like London the distance may be several km, and therefore transit could take a few hours (Table 2).

Modern sewers are designed to be self cleansing (with minimal build up of sedimentary material on pipe bottoms), which requires a minimum flow rate of 0.9 m s^{-1} [(2)]. For a trunk sewer of width 2 m, with a 30 cm depth of sewage this flow rate would imply a volumetric flow rate of 4.7 10^4 m^3 d^{-1}. However, the flow rate can be lower in large diameter sewers or those built prior to 1945, and reference 3 suggests that the flow rate in London sewers may range from 0.3 to 0.75 m s^{-1}, with a mean flow of about 0.5 m s^{-1}.

The usual measure of the average volumetric flow rate of a sewer is the dry weather flow (DWF) which does not include any contribution from surface run-off. The DWF into a typical sewage works may range from around 10^4 m^3 d^{-1} to 10^7 m^3 d^{-1} in works serving large areas of London[3].

2.4.5 Storm episodes

During storms flow rates of sewer effluents can be 50 times the dry weather flow[2]. The typical hydrological conditions during storms are presented in Table 3. These include an analysis of fifteen storms over an 18-month period in urban central England[4], and data from five storms in Munich[5].

If combined sewers are being used either sewage treatment works need the capacity to handle these very large flow rates of water during storms, or the quality of the environment is compromised by allowing the discharge of untreated sewage in such conditions. The latter approach is often taken, with the justification being that the higher river flow rate dilutes the discharges to safe levels. These outflows are usually set to operate when the flow rate is 6 times the DWF; partial treatment (settlement) is given at most sewage treatment works to flows between approximately 3 times DWF and 6 times DWF. All radionuclides present in the effluent during storm events will be disposed of directly into rivers, as no treatments are provided. More recently this argument has been questioned and holding tanks have been installed at some outflows[2].

2.4.6 Solids content and sorption

Radionuclides discharged into sewers will be transported along with the raw sewage. The velocity of the radionuclides will depend on whether they are in the liquid phase or attached, "sorbed", onto either sewer structures or particles and matter within the sewage. This in turn depends on the radionuclide and its chemical form. The solids content of these effluents may be estimated from the average dry mass of solids and liquid discharged to sewers. In the UK in 1997 the average dry mass of solids produced was approximately 55 g d^{-1} per capita[6]. The average quantity of discharged water, of domestic and industrial origin, is approximately 300 l d^{-1} per capita, this is typical of most European cities[2]. The average solids content is thus about 0.02 %, or 200 g m^{-3}, (assuming an overall density of 1 t m^{-3}).

Only short lived radionuclides such as technetium-99m (6 h radioactive half-life) are likely to be affected by any increase in transport time caused by sorption and delay.

2.5 Sewage Effluent Treatment in the UK

2.5.1 Introduction

The main purpose of sewage effluent treatment is to remove suspended solids and biochemical oxygen demand from the liquid phase so that the liquid (which makes up >99% of the sewage) can be discharged into the environment. The solids occur as four main groups:

- Large inorganic and organic matter (floating items)
- Small inorganic particles (grit)
- Small organic particles (sedimentary matter)
- Very fine organic matter (often colloidal)

Sewage treatment plants use a combination of physical and biological methods to remove these materials. There are typically three main steps to the treatment process.

- Preliminary: screening and use of detritors (removal of grit and large floating solids)
- Primary: sedimentation removal of solids (removal of smaller solids)
- Secondary: biological oxidation and flocculation followed by sedimentation (removal of unsettleable solids and dissolved organic matter).

A fourth optional step (tertiary treatment) may remove dissolved chemicals such as nutrients, microorganisms and any remaining suspended matter using filter beds, reed beds and perhaps chemical treatments.

The solids removed during primary and secondary treatment are combined and subjected to pasteurisation or digestion processes (see Section 2.6).

A variety of different methods may be used in the secondary and tertiary treatment stages. These depend on the age, location and size of the STW in question. A summary has been presented in Figure 3.

Unless a storm discharge event occurs, all radioactivity disposed into a sewer will arrive at the sewage treatment plant, minus a fraction removed by radioactive decay and any adsorbed onto the sewer system structure. The activity will arrive both in solution and sorbed onto solids.

The fate of radionuclides during these processes (radioactive decay and partitioning between liquid and solid phases, expressed as removal efficiency) is considered in general terms in the following sections. The effects of sewage treatment on specific radionuclides is discussed in Section 3.4.

2.5.2 Preliminary treatment of sewage effluent

The purpose of the initial sewage treatment stage is to remove large pieces of material by screening and rapid sedimentation. These help to remove materials that could subsequently clog-up or damage process vessels and machinery.

Detritors remove grit particles greater than about 0.2 mm diameter and screens remove large pieces of material, plastics rubber etc. A substantial element of screenings is organic i.e. plastics, cotton wool buds, paper etc. The material removed at preliminary treatment is washed and disposed of, usually to landfill. These treatments can usually be performed at quite high flow rates (up to six times dry weather flow, DWF)[2]. The extraction rate of these materials is not known, however, the suspended solids load of the sewage effluent is likely to be broadly unaffected by this process.

The process is continuous therefore it is unlikely to present any significant additional delay time (although this may be accounted for by the effective distance to travel divided by the flow rate, Table 2). Radionuclides sorbed onto the grit and large solids will be extracted from the sewage at this stage. Given that the preliminary treatment is unlikely to affect significantly the suspended solid load, the effect on the overall radionuclide concentration in the sewage effluent is likely to be small.

2.5.3 Primary treatment of effluent

Following preliminary treatment, sewage passes into sedimentation tanks for primary treatment. The purpose of the treatment is to remove the remaining suspended matter by sedimentation. Settlement of particles in the sedimentation tanks takes between two and six hours[7], depending on the flow at the works at the time. Primary sedimentation removes around 90 % of the settleable solids[2], which reduces the suspended solids load in effluent to around 26 g m^{-3} (using previous assumptions). Settled solids ("primary sludge") are collected for subsequent treatment by methods such as anaerobic digestion.

Most plants have the capacity in sedimentation tanks to process flow rates up to three times the DWF. Any excess is diverted to holding tanks, which have about two hours capacity at the maximum flow rate[2], usually six times the DWF. If the capacity of the sedimentation tanks is exceeded (during storm events) then the holding tanks act as primary sedimentation tanks, with the overflow being discharged to the watercourse. In most STW the holding tanks are not part of the flow to full treatment; so following primary sedimentation the effluent would be discharged to the watercourse.

The delay time of two to six hours (Table 2) in the settling tasks is sufficient to reduce significantly the activity of radionuclides with very short half lives such as technetium-99m (half-life 6.0 hours). In addition, the radionuclides that have sorbed onto the settled solids will also be removed from the liquor with them.

2.5.4 Secondary treatment of effluents

The supernatant liquor drawn off from the sedimentation stage now undergoes a further treatment stage. The purpose of this stage is to remove the remaining biochemical oxygen demand (BOD) in the form of dissolved and very fine suspended organic matter from the supernatant liquor. A bacteria community is introduced into the supernatant and then grows, using the organic matter in the liquor as a food source; the resulting bacteria flocc can then be removed by settling. Radionuclides can become associated with the biomass and settle out. There are two principal physical requirements for secondary treatment of this type: a large area of contact with such bacteria, and the presence of air. These objectives are commonly achieved using one of two approaches:

- Grow bacteria on a suitable media (large surface area) and sprinkle water over them (percolating filter).
- Grow clumps of bacteria in the liquor (large surface area) and aerate it by bubbling (activated sludge).

These two methods are used with almost equal frequency[2]. However, the former generally requires more space and is less suited for use in urban areas[7].

Percolating filters

The bed of stones or gravel used in the percolating filter method is usually about 2 m thick[7], although there are many possible configurations. The supernatant liquor from primary treatment is sprinkled onto these beds and trickles downwards. Bacteria remove the organic matter over a few hours. Any remaining material, including clumps of detached bacteria ("humus") are removed by a secondary sedimentation stage. Some chemical dosing may be carried out during the sedimentation to increase flocculation. The time taken for the secondary sedimentation stage is likely to be similar to that used for primary sedimentation. The total transit time through the percolating filters is about 4 hours; this will affect the activity of short half-life radionuclides such as Tc-99m, by removing them via radioactive decay.

Activated sludge

Settled sewage is held in the aeration tanks containing the activated sludge for 4 to 10 hours[7] whilst the organic matter is oxidised. Activated sludge plants use either bubbles or intense stirrers for aeration and mixing. The clumps of activated sludge are then removed by a separate sedimentation stage. Some of this activated sludge is then recycled to "activate" new sludge, keeping the colonies of bacteria present. The best estimate of transit time through this stage of treatment is about 7 hours, as with percolating filters this will only affect the activity of radionuclides with short half lives.

2.5.5 Tertiary treatments of effluents

There are a number of different tertiary treatments, each designed to remove particular materials from the effluent, typically nutrients, remaining suspended matter, other chemicals and micro-organisms. Removal of nutrients may be needed where discharges are to slow flowing inland rivers, where eutrophication may be a problem; removal of micro-organisms may be required where bathing occurs in receiving waters. The main tertiary water treatment methods are:

- micro straining
- reed beds
- precipitation
- pebble bed or sand filtration[7]
- UV treatment, ozonation, or chlorination

Reed beds are particularly suited to small scale, rural plants. They are, however, becoming increasingly popular because of their high efficiency in removing heavy metals. In addition they are cheap to build and operate. Some species of reeds take up to between 50 and 100 times the concentration of metals in the water.

The main chemical treatments used for nitrogen removal are biological oxidation of ammonia and denitrification by nitrate removal in anaerobic state, often with methanol as a

carbon source. Where phosphates require removal other chemical treatments are necessary. Aluminium sulphate, lime or ferrous salts are all used for such purposes[2]. The former is added post-treatment, the latter two in the primary treatment stages. More advanced treatments include activated carbon filtration, osmosis and ion-exchange[2].

Tertiary treatment of sewage is not yet widespread in the UK, although it is expected to become more common as a result of the EC Urban Wastewater Directive, 1991 and the EC Directive on sewage sludge application to land, 1986. These directives are explained more fully in Section 2.9.

It is likely that the tertiary treatment processes are short in duration. However, there are few data available for these processes therefore it is not possible to comment on the implications for short half-life radionuclides.

2.6 Sewage Sludge Treatment in the UK

2.6.1 Introduction

Sludges removed from primary, secondary and tertiary processes are collected and usually processed on the same sewage works site. There are two central objectives for sewage sludge treatment processes: to stabilise the sludge for disposal, and to reduce the volume (raw sludge is about two to six percent solid[2]). Stabilisation processes (which reduce odour and pathogens) may include mesophilic anaerobic digestion, thermophilic aerobic digestion or composting. Reduction in volume is achieved by methods such as dewatering, gravity settling or air drying. This section describes these processes and their likely effect on radionuclides. Figure 4 describes these processes and Table 4 summarises the relative use of the different methods.

2.6.2 Initial processing

The purpose of initial processing is to remove any remaining large solid matter and to increase the solids content of the sludge. These are necessary to prepare it for subsequent treatments. Screening is the usual method of removing the solids (eg plastics, rags, glass), however, this stage is usually part of the preliminary sewage treatment. The most common method of increasing the solids content is gravity settling using large (around 20 m diameter) tanks. This can reduce the sludge volume by a factor of two to three. Other dewatering methods (described later) can also be used.

It is also necessary to kill pathogens in the sludge. This is referred to as positive disinfection. Some sewage works use a pasteurisation process. Suitable treatments include heating to in excess of 70°C for half an hour, or longer (4 hours) at a lower temperature of 55°C[8]. The sludge may then be subjected to further "stabilisation" treatments or incineration. If the sludge is to be incinerated then it is unlikely to be digested as this would reduce the calorific value of the sludge.

The duration of these initial processes is variable. For simple screening and treatment to kill pathogens the duration is unlikely to exceed a few hours. Dewatering is likely to take longer. The sludge may be left in gravity settling tanks for some considerable time, perhaps a number of days. Alternatively mechanical or chemical dewatering processes may be used and these only take a few hours. The radionuclides most affected by radioactive decay during these processes are thus likely to be those having half lives of a few hours to a few days, and are technetium-99m (half-life 6.02 hours), iodine-131 (half-life 8.04 days) and phosphorus-32 (half-life 14.29 days).

Untreated sludge has a solids content of between 1.5 and 6 % of the total wet mass, the rest being water[8]. The solid fraction may be increased by a factor of 2 - 3 if the sludge has been dewatered. The radionuclides that are present are those that have sorbed onto solids and those radionuclides in the water that is present. Any radionuclides present in the water separated from the sludge may be recycled to the head of the sewage treatment plant.

2.6.3 Stabilisation treatments

Mesophilic anaerobic digestion

The most popular method of stabilisation is (mesophilic) anaerobic digestion. Here, the action of anaerobic bacteria decomposes the organic material into methane, hydrogen sulphide and hydrogen, reducing the solids content by around a third[9]. This is generally used to generate electricity or is used as fuel for incinerators *etc.* Typically the digestion process takes between 12 and 30 days. It is recommended that at 35°C it is retained for at least 12 days, or 20 days at a lower temperature of 25°C.

The digested solids undergo a further settlement stage in an open tank or lagoon before being disposed. This secondary stage lasts at least 14 days[8]. As the digestion process has converted some of the solids to gas the solids content is usually lower than before treatment, around 3.5 % of the total wet mass[8].

Thermophilic aerobic digestion

This alternative to anaerobic digestion provides a much shorter retention time of about 7 days, and requires no initial positive disinfection[8]. Both the lack of the need for disinfection and the shorter retention time make this process suitable for smaller works; however, there have been odour issues with aerobic digestion plants and they are not now used in Yorkshire. Firstly the sludge is heated to around 55°C for 4 hours[8]. The material is then held whilst aerobic bacteria, flourishing in the aerated conditions, oxidise the material. The solids content of the sludge is likely to be around 3.5 % of the total wet mass[8].

Composting

Composting is used extensively at medium sized works in Yorkshire with the product used as a synthetic soil (e.g. for tip or contaminated land reclamation). It is usually required to heat the sludge to 55°C for four hours, followed by 40°C for five days and then a period of further

maturation[8]. Composting plant may typically retain the material for between about 10 to 35 days[6]. Forced aeration or mechanical turning is usually part of the process. It may also be necessary to improve the aeration and reduce temperatures by adding filling material like straw. The optimum solids content of the compost is about 45%, although a range of 40 to 60% is usually acceptable[6]. The organic matter content should ideally exceed 70%.

Other stabilisation treatments

Stabilisation and destruction of pathogens can also be achieved by the addition of lime. The pH is raised to in excess of 12 for three hours[8], and the chemical conditions cause the temperature to rise to 70 to 80°C. The sludge may then be used directly. This treatment is uncommon in the UK. "Cold" digestion has also been practised in the past, but is very slow: untreated liquid sludge (or dewatered solid cake) must be stored for more than three months for stabilisation to occur. Untreated sludge has a solids content of about 1.5 to 6% whilst dewatered cake has a solids content of about 25%[8]. Recently, Anglian Water in the UK has encouraged the use of lagoon storage and cold digestion on farms where sludge is to be spread. In the Anglian area 70% of sludge used on farmland has gone through this treatment. However, this approach has recently been criticised by a parliamentary committee[10].

Consequences for radionuclides

As can be seen from the previous section, the duration of these processes varies from about one week to several months. Therefore radionuclides with half-lives less than 1 day will decay during these processes. The longest duration treatment is cold digestion, which is likely to require several months. The commonly used thermophilic anaerobic digestion is quicker, taking several weeks to a month. The fastest stabilisation is thermophilic digestion, taking about a week.

Other than radioactive decay, the quantity of radionuclides present may be altered by either their recycling to the head of the plant in separated liquor, or their release as gases as a result of bacterial action. All radionuclides may be recycled in liquor, but those most likely to stay in the liquid phase are hydrogen, carbon, sulphur, phosphorus, technetium and iodine. These will also be present in the liquid in the sludge. Tritium, carbon-14 and to a lesser extent sulphur-35 may be released as gases. The temperatures encountered during these processes are insufficient to cause any significant volatilization of non gaseous radionuclides.

2.6.4 Dewatering

Dewatering may not be necessary if the sludge is intended for land filling or land spreading (agriculturally, on forests or for restoration). However, there is often an economic case to dewater if substantial transport costs are involved. Dewatering may be carried out on raw or digested sludges and a range of methods are available. Physical processes such as air drying, centrifuges and filter belt presses are commonly used. Conditioning chemicals (polyelectrolytes)

may also be used. Some incinerators use recycled heat to dry sludges prior to incineration. In general, dewatering increases the solids content to around 25%[8].

Dewatering stages are likely to be short in duration, with most methods taking no more than a few hours. The stage is unlikely to result in any significant additional decay to radionuclides still present after stabilisation. Any radionuclides present in water separated from the sludge may be recycled to the head of the sewage treatment plant. Tritium may be released in water vapour.

2.7 Disposal of Sludge and Effluent

2.7.1 Introduction

Following sewage sludge treatment processes there are four main options for disposal of sludge. These comprise application to farmland (accounting for the majority of sludge), land filling, incineration and sea disposal. Proportions and some typical costs are shown in Figure 5 for the UK and EU[1]. To cope with the removal of the sea disposal route at the end of 1998, water companies are having to consider new management options for sewage sludges. Table 5 summarises the options considered by 11 regions in the UK to replace the sea disposal route. As can be seen, the majority appear to favour the incineration of sludge. Such an approach is exemplified by the commissioning (in 1998) of two new incinerators at Beckton and Crossness by Thames Water. These fluidised bed incinerators, which produce 12-14 MW of electricity, replace the dumping to sea from two of the largest sewage treatment plants in Europe.

Treated liquid effluents are disposed of to rivers or the sea. The EC urban wastewater treatment directive of 1991[11] lays down minimum standards of processing of these effluents, depending on the size and nature of the receiving water body.

2.7.2 Application of sludge to farmland

The application of sewage sludge to farmland is the most popular single disposal method (around 44% in the UK and 37% in Europe). The sludge is a rich source of phosphates, and anaerobically digested sludge has considerable quantities of ammonical nitrogen. Sludge can be applied either by spreading or by direct injection during ploughing. The typical application rates are presented in Table 6. The main drawback of sludge application to farmland is the cost of transportation of the sludge (especially relevant to urban sewage works) and the potential problems of meeting limits on heavy metals and pathogens.

Sewage sludge is generally applied to land by spreading from trailers on tractors. If applied to grassland it should not be grazed for at least three weeks[8]. In the case of arable land, it should be ploughed and cultivated as quickly as possible after application. Direct injection has become more popular as it reduces hazards to animals, odour and unsightliness. Direct injection rates should not exceed 140 m^3 ha^{-1}, and the main period for injection is August to March[8].

The application of raw sludge to land ceased at the end of 1999. Guidelines on the application of sewage sludge are provided. A Code of Practice[12] was prepared to complement the Sludge (Use in Agriculture) Regulations 1989 (SI 1989, No. 1263) covering Great Britain and the Sludge (Use in Agriculture) Regulations (Northern Ireland) 1990 (SR 1990, No. 245). These regulations enforce the provisions of the EC Directive of 1986[13]. Various limits on potentially toxic elements (PTEs, such as zinc, cadmium, chromium, lead *etc*) were established by the (then) Department of the Environment. There are a range of requirements to reduce the pathogens in the sludge. Finally, there are constraints on the duration before crop planting or harvesting can occur. The limits and constraints do not apply to land used for historic disposal of sludge or so-called "dedicated land". The latter, for which there are special provisions, is land which has no other purpose than to receive sewage sludge. No new dedicated land may be created.

Land spreading leads to the incorporation of radionuclides in the environment and in foodstuffs. These may then result in the exposure of farmers and the public. The transfer of radionuclides into foodstuffs is dependent on the rate and nature of the application of the sludge to the land and the subsequent use of the land (in particular crop type and time of harvesting relative to the application of sludge). In addition, sludge is usually only spread on land once or twice annually (in intervening times it is stockpiled centrally or on farms)[6]. There is thus a substantial period during which radionuclides can decay.

2.7.3 Disposal of sludge to landfill

The practice of sending sludges to landfill directly appears to be diminishing, only about 5% of all landfills receive sludge, which represents less than 1% of the waste disposed of via this route. In 1978 about 33% of sludge was sent to landfill, compared with 10% in 1993/1994[8]. Predictions suggest that the quantity of sludge following this route may decrease yet further[8]. Sewage sludge is considered to be a controlled waste, unless it is used on farmland, and is treated as such when landfilling. Therefore sludge can be landfilled as long as it meets the requirements of the licensing authorities and the operator. A range of conditions stipulate handling and emplacement procedures. The main problems associated with the landfilling of sewage sludge are the potential contamination of ground water, and the likely increased generation of landfill gas following its disposal. It is primarily for these reasons that sludge is usually co-disposed with municipal wastes. Normally the proportion of sludge co-disposed with municipal waste is less than 20% by weight[9]. It is also usually dewatered, so the solids content is in the range of 15 to 25%[8].

The disposal of radionuclides in landfill means that in the near future (tens of years) any radionuclides present will be retained in the waste. Most radionuclides will therefore decay in the landfill. Long-lived mobile radionuclides such as tritium and carbon-14 could possibly give rise to exposure of members of the public via landfill gas emissions and migration with ground water.

2.7.4 Incineration of sludges

Incineration is an increasingly common way of disposing of dried sludge. The fraction incinerated in the UK was 7% in the early 1990s; however, this is expected to rise substantially as sea dumping is now prohibited. Following incineration a substantial ash residue is left which must be disposed of by landfill (a typical sludge has an ash content of 25 to 30 % of dry solids[8]). However, incineration is attractive because it is a quick and cost-effective method of reducing the volume and stabilising the sludge. If the solids content of the sludge exceeds about 30% the incinerator does not require additional fuel to burn. Dewatering of sludge is achieved either by the previously described processes or the use of waste heat from the incinerator. In the latter case, a fluid bed dryer can achieve a dry solids content in excess of 90 % and a contact dryer can achieve about 50%.

There are two types of incinerators generally in use. These are the multiple hearth and fluidised bed types. The older multiple hearth design contains a series of hearths which are stacked on top of each other. The dried sludge is added at the top and slowly falls down until it reaches combustion. The remaining ash can then be collected at the bottom. In a fluidised bed furnace the sludge enters a single chamber and falls onto a bed of sand, fluidised by an air blower. The minimum operating temperature of the fluidised bed incinerator is 850°C, this will ensure complete burnout of the flue gas. Although having higher power demand, the latter furnaces have better combustion efficiencies and odour control and require less complicated plant (but more complicated control) systems. They are consequently the preferred choice in new plants[6].

An analysis of sludge ash shows that it is mostly comprised of silicon and aluminium oxides, and that heavy metals comprise less than 0.1% by mass[8]. Combustion is fairly complete as the organic and volatile matter content of the remaining ash is only about 2%.

Upon incineration radionuclides are either released to air, from where they disperse and may deposit to the ground, they may be captured in offgas scrubbers or retained in the ash. The ash is then likely to be taken to a landfill and buried, although some companies are researching more beneficial uses of incinerator ash. Off gas scrubbers may produce a slurry which may be returned to earlier parts of the system for treatment. The incineration process therefore may lead to exposure of the public by:

- Atmospheric dispersion and deposition of radionuclides
- Disposal of radionuclides in municipal tips.

The radionuclides most likely to be released into the air are tritium, carbon-14 and iodine isotopes[14], the fraction remaining in ash probably only being a few percent. The majority (>80%) of other elements such as chromium or phosphorus are likely to be retained in the ash.

2.7.5 Sea disposal of sludges

Sea dumping of sewage sludge has long been exploited in the UK. As can be seen in Figure 5, the UK was the only country in Europe to extensively use this route. The process was simple: sludge (which was usually thickened but not dewatered) was loaded at a centralised point onto ships. These proceeded to a disposal site where the sludge was dispersed into the sea. Table 5 shows the quantities (and locations where known) of some past sea disposals of sludge[15]. As can be seen, several larger cities disposed of in excess of a million (wet) tonnes of sludge per year.

In 1991, the EC Directive on Wastewater Treatment[11] specifically prohibited this practice from the end of 1998 onwards, and also laid down restrictions on the use of this route in the intervening years (most importantly that it should not grow in size). Table 5 summarises the options to replace sea disposal. In almost all cases the sludge that would have been disposed of at sea has been treated by incineration. This decision has been made on a financial basis: other options involve significant transport costs and the ready availability of either landfill capacity or agricultural land.

Radionuclides may find their way back into the food chain relatively quickly, as they may be ingested by fish or crustacea which are caught and sold as fresh seafood. It should be noted that although there may be high transfer factors from effluent to organisms, this will not necessarily give rise to significant concentrations within the fish or crustacea because of the highly diluting effect of dispersal in the sea.

2.7.6 Other routes for sludges

Application to derelict land and forests

Sludge can prove a useful fertiliser in land reclamation. It is effective in restoring the fertility and soil structure. It can also be used in forestry, with the advantage that it can be used year-round[8]. In addition, trees are not in the food chain and so there is less concern over the effects of heavy metals and pathogens. However, this application is usually restricted to commercial rather than amenity forests, because of odour and unsightliness problems. In 1991 68,000 t of dry sludge was used in land restoration (about 5.5% of all sewage sludge in the UK), and there is evidence that this route could account for all sewage sludge, although this is unlikely. Currently only about 1% of the total sewage sludge is used in forestry. The potential exists for this to rise to about 11%, although this is unlikely. As with land spreading, the main problem with exploiting these uses is logistics. This is less serious for land reclamation, as the land is often located close to urban areas, however, it is a major drawback for forestry applications.

For derelict land, heavy applications of about 100 t ha^{-1} are required[9], most commonly using sludge cake. In some colliery reclamation schemes application rates as high as 700 t ha^{-1} have been used. In forestry the maximum annual application should be 200 m^3 ha^{-1} of liquid sludge or 50 t ha^{-1} of cake sludge[8].

The application of sludge on forestry or derelict land may be year-round, therefore there is no decay time before it is spread. However, the use of this land means that the radionuclides in the sludge will not find a way directly into man's food chain for several tens of years. During this time short-lived radionuclides will have decayed significantly.

Use of sludges as a fertiliser

Caked sludge has been sold to domestic gardeners and allotment holders, but this has ceased due to tighter regulatory control of the material. Of particular concern was the potential inadvertent ingestion of sludge by infants. Sludge has also been marketed as a fertiliser for amenity areas; however, there is strong competition from other products in this section of the market. One water utility has recently developed a commercial enterprise supplying soil substitute fabricated from digested sludge.

2.7.7 Treated effluents

Treated effluents may be released into the river or sea. The levels of treatment necessary depend upon the water body that the effluent is discharged to. The main distinction is that for smaller receiving rivers, tertiary treatment may be necessary before the effluent is disposed of to prevent eutrophication. The following general rules therefore apply:

- Crude sewage and primary treated effluent (sedimentation only) can be discharged to sea, although this practice is being phased out in most areas
- Secondary treated effluent (following biological filtration) may be disposed of to some rivers or sea
- Tertiary treated effluent (following further filtration and chemical treatment as necessary) may be disposed of to most rivers or coastal waters

By the end of 2000 larger flows must have full treatment and medium flows must have full treatment by 2005, appropriate treatment must be given to other flows. By 2005 all coastal STWs (above 10 000 population equivalent in Yorkshire) will have secondary treatment and ultra violet disinfection.

The disposal of effluents to sea and river provides routes by which members of the public can be exposed to radionuclides. Downstream of the discharge point the abstraction of river water for irrigation or drinking water, irradiation by contaminated sediment and fishing provide routes of exposure to people.

2.8 Practices in Europe and North America

2.8.1 Sewage collection and treatment

Different countries have historically had differing attitudes to sewage transit systems. In Finland and Sweden less than a third of the sewer systems were combined, whilst combined systems are still widely used in Austria and Switzerland[2].

The EC Directive on wastewater treatment, issued in 1991, should have the effect of harmonising many of the waste water treatment approaches to a minimum standard. Countries that have privatised utilities (like the UK) or place higher political value on environmental issues may significantly exceed these minimum standards. However, the overall effect of the Directive is that, within Europe, urban wastewater will usually be subject to a secondary treatment and more probably a tertiary treatment.

An indication of the different secondary treatments in different European countries has been presented in Table 4. In Scandinavian countries, tertiary chemical treatments have been used widely for many years. In some warmer climates, especially parts of North America, lagoons are used instead of specific tertiary treatments. Here the secondary liquid effluent is left to be naturally digested by bacteria over a few days.

2.8.2 Sludge treatment and disposal

Figure 5 presents a summary of the fractional utilisation of different sludge disposal routes in Europe in the early 1990s. The use of different sludge treatments in European countries is presented in Table 4. The different approaches are to a large extent dictated by demographic and geographic factors. Between 4 and 76% of sludge is not stabilised, depending on the country. The most popular stabilisation method was anaerobic digestion, although some countries also had a high utilisation of aerobic digestion. Other treatments such as composting are used significantly in some countries (eg Austria) but limited in the rest of Europe [2].

In general, in the late 1990s, the main disposal route for sludge in Europe is land filling and use in land reclamation and forestry (43%), followed by application to agricultural land (37%). However, in some countries, regulations on the sewage sludge application to agricultural land are so strict they restrict the utilisation of this disposal option[6].

There are a number of legislative developments that have or may affect the sludge disposal route. Most importantly, the EC Directive on urban wastewater treatment[11] prohibits the dumping of sewage sludge at sea after the end of 1998. Another route subject to EC legislation is the agricultural use of sludge. The EC Directive on sewage sludge[13] lays down the minimum requirements for toxic substances (eg heavy metals) and a range of other requirements.

2.9 Legislation

2.9.1 EC Urban Wastewater Directive

A Directive on urban wastewater treatment was issued by the EC in 1991[11]. This Directive has had a significant impact on sewer treatment practices in the mid to late 1990s, and the key issues are summarised here.

The Directive lays down various degrees of treatment which must be provided for population centres (greater than 2,000 people for discharges into rivers and estuaries, greater than 10,000 people for discharges into the sea). An appropriate standard is defined which states that the minimum secondary treatment must leave effluents with a suspended solids load of less than 35 mg l^{-1}, 25 mg l^{-1} biological oxygen demand (BOD) or 70% removal of BOD and 125 mg l^{-1} chemical oxygen demand (COD) or 75% removal of COD.

However, to take account of the natural variation in the sensitivity of receiving waters, sensitive and less sensitive areas are defined by national authorities. In sensitive areas a tertiary treatment process must ensure that phosphates and nitrogen levels be reduced to agreed levels. In less sensitive areas (defined by appropriate studies) primary treatment alone is acceptable. However, the BOD must be reduced by at least 20% and the solids by 50%.

Finally, a requirement which was very important to the UK was that sea dumping of sewage sludge had to cease by the end of 1998. This has understandably led to a significant reorganisation of the industry in larger urban areas.

Improvements in waste water treatment, such as the introduction of tertiary treatments for effluent and finding new disposal routes for sludges are capital intensive. The introduction of these new processes has been aided by the availability of private capital in the water utilities.

2.9.2 EC directive on sewage sludge application to land

The EC issued a Directive[13] in 1986 which was required to be implemented in member states by mid 1989. It was solely concerned with agricultural applications of sludge, and had the following requirements. The use of untreated sludge is prohibited unless it is injected into the soil. Treatment itself is further defined as chemical, biological, the application of heat or of long term storage. The time of application to certain crops is also prescribed, for example there must be at least a 10 month delay before the crop is harvested, and it must not be applied where fruit or vegetable crops are grown. Other directives cover the protection of ground water. However, the sludge application directive addresses specifically the problems of heavy metals (in mg ha^{-1}) and gives requirements for sampling regimes.

Finally the directive sets out requirements for record keeping. Aspects that must be addressed are the quantities of sludge used in agriculture, the composition and properties of the sludge, the type of treatment given and the locations used.

The UK implemented this directive in the sludge (use in agriculture) regulations of 1989. These are backed with a code of practice setting out the best practice in more detail[12].

2.9.3 Recent developments

The House of Commons Environment Committee has recently considered standards of sewage treatment and sludge disposal[10] in light of the EC Wastewater Directive[11]. The main recommendations were that all sewage effluents should receive tertiary treatment to reduce nutrients and pathogens, and that the spreading of untreated sewage sludge on farmland be phased out. These recommendations exceed the requirements of the 1991 EC Wastewater Directive. Other important recommendations were made: sewage sludge should be pasteurised before application and sewer systems should be designed so that sewer overflows would only need to operate once in 20 years, ie overflows would only need to be utilised in a storm event so severe that it is likely to occur once every 20 years. Finally it was suggested that the "less sensitive areas" (otherwise known as areas of high natural dispersion) be immediately abandoned. The target date for the implementation of the recommendations is 2002. Many water companies have anticipated these more exacting standards, particularly the question of tertiary treatments. The committee noted progress on UV treatment and micro filtration; in total 35 outfalls are now UV treated.

The Environment Committee noted that the level of sludge treatment was diverse: some being applied to land untreated (70% of sludge produced by Anglian water), and in some cases almost all treated (95% anaerobically digested by Severn Trent water). Whilst supporting the continued use of sludge on land it was recommended that stabilisation and pasteurisation be introduced as a statutory minimum.

2.10 New Developments and Processes

2.10.1 Sewage effluent treatment

The recent recommendations by the government[10] suggest even stricter controls on treatment. Water companies that are anticipating more rigorous measures include Wessex Water and Welsh Water, who have both introduced programmes of tertiary treatment using either UV or membrane filtration. In many cases these will replace chlorination. Southern Water is preparing designs for Reed Bed Systems for smaller works serving populations of less than 10,000. Thames Water has introduced a phosphate removal process for final effluent using iron precipitation at several works, and is considering phosphate removal options for other works which discharge effluent to eutrophication sensitive rivers.

A recently developed secondary treatment process is submerged biological aerated filters. These consist of a packed bed of granular material which is submerged with air blown up through it, in the opposite direction to the sewage flow. The filter is cleaned by periodic backwashing, which is returned to the sewage works inlet. This process is intensive and only needs a small reactor vessel volume. In addition there is no need for a secondary sedimentation stage. The

drawbacks are that the backwash means there are periods of down time, and there is little operating experience of these types of filter.

2.10.2 Sludge treatment and disposal

Several new sludge treatment ideas are currently being investigated, amongst them gasification, wet oxidation and plasma pyrolysis. These are reviewed in reference 6. Gasification is a well known method of producing methane, extensively used in the first half of the century. Essentially dewatered sludge (> 85% dry solids) is heated with sub-stoichiometric quantities of oxygen. Carbon reacts with carbon dioxide or water to form carbon monoxide and hydrogen, which are subsequently converted to methane, ie:

$$C + CO_2 \leftrightarrow 2\,CO$$

$$C + H_2O \leftrightarrow CO + H_2$$

$$CO + 3H_2 \leftrightarrow CH_4 + H_2O$$

These processes take place in a reactor with a fixed, fluid or circulating bed. The main drawback to the process is the varying sludge quality, which means control of the gas production is difficult. However, although this method is still in the demonstration stages for sewage sludge, a full scale plant has recently been commissioned in the UK to apply the same process to municipal waste. This indicates that it may have a viable future.

In the wet oxidation process the organic component of the sludge is oxidised at high temperature (200 to 300°C) and pressures varying from 30 to 150 bar. These conditions may be reached using conventional pumps etc, however, a novel solution is to pump the sludge deep underground. The organic component of the emerging sludge is greatly reduced, and so the processed sludge can be dewatered and landfilled or recycled. A full scale plant is running in the Netherlands but has encountered operational difficulties.

Plasma pyrolysis is being investigated, chiefly in the US, where full size plants are planned to treat municipal waste. It has been adapted from processes in the metallurgic industry. Plasma torches pyrolyse the organic content of the waste at temperature of 2,000 to 4,000°C. The resulting gases (H_2 and CO_2) may be used to manufacture methanol, and the remaining slag material is in the form of a vitrified solid.

Finally there have been several experiments which use sludge in the fabrication of bricks. This has been pursued in the USA in several large projects. In the UK, Anglian Water are considering the use of sludge to produce building materials. The sludge would be mixed with clay and pulverised fuel ash and cooked. The sludge expands to make a lightweight material suitable for blocks and bricks[16].

2.11 Summary of Sewage Treatment and Disposal

The range of wastewater and sewage sludge treatment processes commonly used in the UK have been summarised. However, the situation is changing, driven by legislation aimed at better protection of the environment. The most significant change was the ending of sewage sludge dumping at sea by the end of 1998. This has primarily affected large urban areas (such as London) which previously disposed of sludges by this route. As disposal of sludge to agricultural land is not feasible for such areas, much larger quantities of sludge are likely to be incinerated. Indeed, indications show that the majority of sludge formerly disposed of to sea will now be incinerated (30 to 40% of all sludge). Other changes brought about by legislation are the increasing use of secondary and tertiary treatments. Many of these improvements have been aided by the availability of private capital in the water utilities. There is no clear picture of the likely influence of emerging novel treatments: most improvements may be made using established technology.

3 USE AND BEHAVIOUR OF RADIONUCLIDES

3.1 Introduction

This chapter describes the use and behaviour of radionuclides that are authorised under the Radioactive Substances Act 1993[17], for discharge into sewer systems. The chapter has therefore been divided into the three following sections:

- The choice of radionuclides for the study
- A discussion of the uses of the radionuclides
- A review of the behaviour of elements in sewage treatment processes

3.2 Choice of Radionuclides for Study

The radionuclides most commonly disposed of to sewers were identified and those most likely to have the greatest radiological impact were selected.

The main use of radionuclides which are discharged to sewers is tracer studies in medicine, industry and research. The radionuclides chosen for such applications are easily measured, have low radiotoxicity, short radioactive half-life, and are available in a form giving suitable chemical or biological behaviour. The ideal radionuclide is a reasonably energetic beta or gamma emitter with a radioactive half-life of a few days or weeks. There are exceptions, notably tritium and carbon-14, which have much longer half lives but weak beta emissions (lower radiotoxicity). Using information on importance of usage, eight radionuclides were identified for inclusion in the study:

- tritium, carbon-14, phosphorus-32, sulphur-35, chromium-51, technetium-99m, iodine-125 and iodine-131

The final selection of radionuclides was made from information collected on disposal authorisations for London and Leeds (radionuclides disposed and their quantities) and the overall flow of sewage effluent. This information was used to estimate the average activity concentration of each radionuclide, and these were compared with limits of detection. The study indicated that if the authorisations were used to their fullest extent, seventeen radionuclides could be present in raw effluent at levels potentially above limit of detection (either at Beckton or Knostrop). These seventeen radionuclides were selected for inclusion in the study:

- the eight radionuclides identified above, plus gallium-67, indium-111, thallium-201, rubidium-83, rubidium-84, strontium-89, yttrium-90, iodine-123 and americium-241.

3.3 Use of Radionuclides in the Non-Nuclear Industry

3.3.1 Introduction

These 17 radionuclides are used as tracers in medicine, chemical, industrial or biological research and in therapy. The use of such tracers is generally confined to four types of establishment:

- Hospitals, for example diagnostic tracers in nuclear medicine;
- Universities, for example research into the behaviour of novel chemicals,
- Research centres for industries, for example the distribution of pharmaceuticals in tissues.
- Research into agriculture, environmental sciences and geology.

Radionuclides are used in a therapeutic manner in hospitals. In particular iodine-131 and strontium-89 are used to treat cancers. A summary of the uses of the radionuclides being considered in the study is presented in Table 7.

3.3.2 Chemical forms

The radionuclides considered in this study are frequently used in medicine as biological tracers to provide images or to irradiate cancer tumours in the body. In order for the radionuclides to be effective as tracers or to accumulate in organs of the body, they may be incorporated into a wide variety of chemical compounds. Commercial organisations supplying radionuclides provide an enormous array of carrier chemicals[18]. These chemical compounds can play a very significant role in the subsequent behaviour of the radionuclides in the sewage system (see Chapter 2). Table 8 presents the compounds used in the 20 most frequent applications of nuclear medicine in 1989/90[19].

3.3.3 Usage in nuclear medicine

Reference 19 gives an indication of the frequency of use of radionuclides in nuclear medicine. However, since then both the number of treatments and the rate of use of radionuclides have increased. The UK nuclear medicine survey[20] carried out in 1993 contacted 187 of the 235 sites using radio pharmaceuticals, (accounting for about 84% of the administered activity). This showed that the number of procedures involving nuclear medicine increased from 430,000 to 490,000 between 1989 and 1993. The main area of increase is thought to be myocardial perfusion and lung ventilation studies. Of the investigations in 1993, 90.3 % were for imaging purposes, 7.1 % for non-imaging purposes and 2.6% for therapy. The average number of imaging studies carried out per department in 1993 was 2198. The total administered activities were calculated by assuming that the average activity administered for each investigation was the same as that found in the 1989 survey[19]. The most frequently used radionuclides in 1993 were Tc-99m (1.36

10^{14} Bq), followed by Kr-81m gas (1.14 10^{14} Bq), I-131 (8.27 10^{12} Bq), Xe-133 gas (3.55 10^{12} Bq) and Tl-201 (1.22 10^{12} Bq). These data are summarised in Figure 6.

A recent review of uses of radionuclides in a large hospital in Northern Ireland[21] gives further information on the pattern of use in a nuclear medicine department. Tc-99m is used frequently but in moderate quantities, whereas I-131 is used for thyroid ablation, and therefore large quantities are used relatively infrequently. These data are summarised in Table 9.

3.3.4 Properties of the selected radionuclides

The following section discusses each radionuclide in turn and includes data on half lives and emissions[22], common uses including medical uses[23], chemical forms and quantities.

a) Tritium

Tritium has a half-life of 12.35 years and emits weak beta radiation (average energy 5.7 keV). It is formed naturally in the upper atmosphere and is present at low activity concentrations in the environment. It is produced commercially in nuclear reactors. In the environment tritium is encountered as tritiated water and "organically bound" tritium (when it is associated with organic molecules). The weak beta emissions of tritium mean that it has a relatively low radiotoxicity. It is useful as biological tracer, particularly for organic compounds. However, the exchange reactions in labelling (tritium for stable hydrogen) are not efficient meaning large activities are required. In addition, the stability of tritium in molecules is very variable[24]. The widest commercial use of tritium is in light devices (gaseous tritium light devices, or GTLDs). However, it is unlikely that tritium from these sources would be discharged to sewers. In medicine tritium is used for procedures such as the measurement of total body water.

b) Carbon-14

Carbon-14 is produced naturally in the upper atmosphere by reaction of cosmic radiation with nitrogen. Carbon-14 is a 'soft' beta emitter, with average energy of 49.5 keV. It has a half-life of 5730 years, and as a result of natural production, long half-life and high mobility is found throughout the environment. Carbon-14 concentrations are frequently expressed in terms of total carbon. Man's activities, like the burning of fossil fuels and nuclear weapons tests, have recently started affecting the equilibrium concentrations in the environment. It is chiefly known for its application in determining the age of archaeological relics and fossils, but has had applications in chemistry and biological research (particularly plants).

c) Phosphorus-32

This is a relatively energetic beta-emitter (average energy 694.7 keV) with a half-life of 14.29 days. It is produced in reactors by irradiation of sulphur-32. It has been used in medicines, administered as sodium phosphate, for labelling red-blood corpuscles, nucleotides and pharmacological substances and treatment of polycythemia vera. It finds uses in industry and

research, particularly labelling plasticizing substances and reagents, and in the textile industry. It is also incorporated into inorganic fertilizers and used as a tracer.

d) Sulphur-35

The half-life of sulphur-35 is short, 87.44 days, and it decays by beta-emission with an average energy of 48.8 keV. Very small quantities are produced naturally by cosmic ray interactions. Commercial quantities of the tracer are supplied in a range of forms including the amino acid methionine. It can be used to label biochemical compounds and medicines in order to study their distribution, metabolism and mode of action. In industry it can be used to label surfactants, wetting agents, some detergents *etc.* It may also be used in various other tracer studies.

e) Chromium-51

Chromium-51 is used as a tracer because of its gamma and x-ray emissions (the highest energy is 320.1 keV, yield 0.098) and short half-life of 27.70 days. It is generally supplied as chromium chloride, sodium chromate and an EDTA complex. It may be implanted into tumours as a wire, and is widely used to study glomerular filtration (administered as EDTA), in red blood cell investigations, and blood and protein loss studies. Typical administration is 2-4 MBq for diagnosis.

f) Gallium-67

Gallium-67 decays with a half-life of 78.24 hours by electron capture, emitting several x-rays with energies up to several hundred keV. It is used in diagnosis, imaging tumours or inflammations. Typical administrations are 150 MBq as Ga^{3+} for diagnosis.

g) Rubidium-83 and 84

Rubidium-83 decays to krypton-83 (stable) with a half-life of 86.21 days by electron capture, emitting a range of photons (main energies between 520 and 550 keV).

Rubidium-84 decays to krypton-84 (stable) with a half-life of 32.77 days by electron capture and positrons (branching ratio 0.964) and to strontium-84 (stable) by beta (branching ratio 0.036). The principal emissions are photons, (yield > 0.1, 511 keV, 881 keV) and positrons, (339 keV, 756 keV).

Rubidium-83 and 84 are not widely used. It is known that they were created as a by-product in the production of krypton-81m/Rubidium-81generators, which are used for lung function tests, and residues may be washed off laboratory equipment into the sewer system. However, the route that is now more commonly used to make Kr-81m/Rb-81 generators does not produce these Rb isotopes.

h) Strontium-89

Strontium-89 decays with a half-life of 50.48 days by beta emission with an average energy of 583 keV to stable yttrium-89. It is used in treatment of bone metastases, and is administered as chloride compound.

i) Yttrium-90

Yttrium-90 decays with a half-life of 64.01 h by beta emission with an average energy of 935 keV to zirconium-90. It is administered as a colloidal silicate suspension for treatment of arthritic conditions and malignant diseases.

j) Technetium-99m

Technetium-99m is a very commonly used radionuclide which finds most of its uses in the area of clinical biology. It has a short half-life of 6.02 hours and is a gamma-ray emitter (chiefly with energy 140.5 keV and yield 0.889). Technetium-99m is eluted in the form of a pertechnate (TcO_4^-) from "technetium generators" where it is produced by the decay of Molybdenum-99. It is used in a great many chemical forms (phosphate compounds, colloids, pertechnate, human albumin, sestamibi, erythrocites) each having particular applications. Pertechnate is used for scintigraphy of the brain or thyroid, a colloidal form is used for the liver, spleen and bone marrow, labelled albumin is used to study the blood pool and sestamibi for tumour imaging. Administration ranges from 4-900 MBq.

Tables 8 and 9, and Figure 6 demonstrate the dominance of technetium-99m as a medical radionuclide. Figure 6 also illustrates the high level of krypton-81m activity that was administered. This is partly a result of a significant increase in lung scans (both perfusion and ventilation) during 1993. It should also be noted that ARSAC[23] permits relatively high activities of krypton-81m to be administered during lung ventilation scans because the low dose coefficients for krypton-81m give rise to a low effective dose equivalent. When krypton-81m activity is ignored, over 96% of all activity administered is technetium-99m, compared with about 3% for iodine-131, 0.36% for thallium-201 and 0.31% for gallium-67 (Table 9).

k) Indium-111

Alpha particle bombardment of a silver target or production in a cyclotron are the usual methods of obtaining indium-111. This radionuclide has a half-life of 2.83 days. It decays via electron capture to a stable isotope of cadmium. Its main emissions are 171.3 and 245.4 keV photons with yield 0.905 and 0.940 respectively. These properties clearly show it to be a useful radionuclide for tracer studies, and it is used in medical diagnostic scans. It is administered as organic compounds, leucocytes, non-absorbable compounds, usually 10-110 MBq for diagnoses.

l) Iodine-123

Iodine-123 has a half-life of 13.20 h, decaying via electron capture to tellurium-123. Its main emissions (yield > 0.1) are 159 and 127 keV photons and 27.5 and 27.2 keV X-rays. It is widely used in medical diagnosis, for imaging of tumours in the thyroid and renal area. It is not normally used in treatment. It is administered as iodide or organic compounds, usually 2-400 MBq.

m) Iodine-131

Iodine-131 has a half-life of 8.04 days, and is produced in nuclear reactors. It has a range of beta- and gamma- emissions of several hundred keV. Iodine accumulates in the thyroid, and administered ionic iodine-131 will accumulate there where it may be used to treat thyrotoxicosis, goitre and carcinoma. It may also be used in its ionic form as a tracer to study the thyroid and metabolism. Radio cardiographs may be taken by the inhalation of iodine labelled methyl-iodide. Various other iodine-131 labelled products may be used to study the liver and kidneys. In agriculture, iodine-131 (as potassium iodide) may be used as a tracer for water penetration. Typical administration for diagnosis is between 0.2 and 400 MBq, whilst for therapy administration is of the order 10-20 GBq.

n) Iodine-125

Iodine-125 has a half-life of 60.12 days and produces two low energy x-rays of 27 and 35 keV. Its principle advantage over iodine-131 is its longer half-life which makes it easier to store and transport. It is suitable for diagnosis and imaging. It may be applied as organic compounds, human albumin or iodide. It is not used for thyroid treatment. Typical administration is between 0.2 and 4 MBq.

o) Thallium 201

Thallium-201 has a half-life of 3.04 days and decays by electron capture. It emits a range of photons with energy up to several hundred keV. Thallium-201 is used in medical diagnosis, myocardial scans and thyroid imaging may be performed by administering thallous iron containing Tl-201 as Tl^{+}. Typical administrations are between 80 and 150 MBq.

p) Americium-241

Americium has a half-life of 432.2 y and decays by alpha particle emission. This radionuclide is used in anti-static sources or smoke detectors (sealed), and may also be used in research (unsealed).

3.4 Behaviour of Radionuclides During Sewage Treatment

3.4.1 Introduction

As described previously, the partitioning of nuclides between the solid and liquid phases, and the resulting removal of solids determines the fate of the radionuclides, the resulting concentrations and the radiological consequences. Two characteristics of the radionuclide will determine this behaviour:

- The general chemistry of the element in the sewage media
- The chemical/biological compound that the radionuclide is associated with when disposed.

The behaviour of heavy metals, radionuclides and other trace pollutants in effluent during sewage treatment are expressed in terms of a "removal efficiency" which is the activity in the effluent stream after treatment divided by the initial activity, (ignoring any radioactive decay). Removal efficiency takes into account partitioning of radionuclides onto solid phases followed by physical separation of the phases and the radionuclides. The two treatment steps where separation of effluent from solid phases occurs are during primary and secondary treatment, and it is at these points where removal to one phase or other may be significant. Past studies were reviewed and removal efficiencies for heavy metals are presented in Table 10. The distribution of tritium between the solid and liquid phases is given in Table 11 for a particular STW and removal efficiencies for the elements being considered in the project, are presented in Table 12. It was decided to use one removal efficiency to account for the removal of a radionuclide during both primary and secondary treatment. Most of the data was obtained from references 24 and 2. Other studies have either been concerned with the implications of heavy metals in sewage from industrial processes or radionuclides entering the sewage system as a result of a nuclear accident.

Where data were not available removal efficiencies were estimated by using chemical analogues and sediment Kd values to classify the nuclide into "efficiently" or "inefficiently" removed (Appendix A). A best estimate removal efficiency was then derived and used in the study.

For most elements the best estimate value is the mean of the range identified, with allowance for other data that may indicate a more specific value. Where possible the best estimates have been based on data for real, full-scale sewage processing plant. For some elements, such as iodine, the chemical form and experimental methodology have also been considered.

The best estimate values used in this study are given in Table 13 and discussed below, the two values given in Table 13 refer to the different secondary treatments occurring at Beckton (activated sludge) and Knostrop (percolating filter).

3.4.2 Removal efficiencies for specific radionuclides

a) Tritium

Few studies of the behaviour of tritium in sewage treatment processes have been conducted. However, a study of the fate of around 500 GBq y^{-1} of tritium disposed to sewers in Cambridge reports the concentrations in various environmental materials[25]. The Cambridge sewage works treats sludge by anaerobic raw sludge digestion followed by dewatering. Analysis of 13 samples of the sludge cake taken from August 1987 to March 1988 showed an overall partitioning of tritium to the solid. The range of values (in Bq kg^{-1} dry matter per Bq l^{-1} in free water) was 9 - 138, with a mean of 50. There is clear evidence of preferential partitioning of tritium into the dry matter and it is likely that the bulk of the tritium is contained within the microbial biomass. The total inventory of tritium was considered to estimate the fate of tritium in the treatment process, see Table 11. Table 13 gives the best estimates.

b) Carbon-14

As with tritium, few studies of the behaviour of carbon-14 in sewage treatment processes have been conducted. However, the study of radionuclide disposals in Cambridge[25], provides information on the behaviour of carbon-14. It was estimated that a total of 917.8 MBq of carbon-14 was discharged to drains in Cambridge in 1987, the disposal patterns being quite variable. There was some uncertainty in the natural background levels of carbon-14, due to the introduction of fossil carbon sources such as detergents. The range of baseline levels was taken to be 81% to 116% of the average modern background level. Using these considerations (respectively) it was estimated that the fraction of carbon-14 activity released to river following treatment was between 76.2 and 91.1 %. The fraction retained in sludge was estimated to be between 11.8 and 4.4 %, and the total released as $^{14}CO_2$ was estimated as between 12 and 4.5 %.

c) Phosphorus-32

It is reported that early investigations of the behaviour of phosphorus (as phosphates) showed that adsorption onto primary sludge reached a peak after 8 hours[2]. The adsorption of phosphorus-32 may be depressed where significant quantities of stable phosphorus are present. The removal efficiencies of pilot-scale percolating filter and activated sludge plants were measured and are presented in Table 12. These data show the effective removal of phosphate ions by both treatments in idealised conditions, with the activated sludge plant being more effective. A review of other data found that the overall removal efficiency of full-scale plant was lower, between 20 and 30%. Other studies could not detect phosphorus-32 in sewage in a real plant after a known discharge to sewer.

d) Sulphur-35

A study of radionuclide disposals in Cambridge[25] considered the fate of sulphur-35 in drains and sewage. It was estimated that a total of 15.54 GBq of sulphur-35 was discharged to drains in Cambridge in 1987, the disposal patterns being quite consistent. Sewage sludge cake was sampled on one week in October 1987. If the average measured activity concentration persisted for the entire year, 7.7% of the activity disposed would be expected to remain in the sludge with the remaining 92.3% being discharged in the treated effluent (supernatant). No estimate was made of the quantity of sulphur-35 released as gas.

e) Chromium-51

Reference 24 followed the fate of several GBq of chromium-51 disposed in 1987. The activity concentration in processed waters was below the limit of detection, which made it difficult to validate the partitioning of the element. However, it was estimated that about 38% of the discharged activity (correcting for decay) was retained in the sludge, equivalent to a removal efficiency of 62%. Studies of the behaviour of stable heavy metals in sewage treatment have also considered the removal efficiencies of chromium[26]. An average of removal efficiency of 78 % was found across the entire treatment process, with activated sludge plant appearing to be more effective at removing chromium (and other heavy metals).

f) Rubidium-83 and 84

No reported values were available for either rubidium-83 or 84. Removal efficiencies were estimated from possible analogues such as caesium-137 and heavy metals. The range of removal efficiencies is given in Table 12 and the selected values are given in Table 13.

g) Strontium-89 and yttrium-90

No reported values were available for strontium-89 or yttrium-90. Removal efficiencies were estimated from possible analogues. The range of removal efficiencies is given in Table 12, and the selected values are given in Table 13.

h) Technetium-99m

Because of the rapid decay of the nuclide it is difficult to establish the behaviour of technetium-99m in sewage treatment plants. The most influential factor is the transit time to, and through the sewage works. This is always significant in comparison to the half-life of technetium-99m. A study of radionuclide disposals in Cambridge[25] considered the fate of 281.6 GBq of technetium-99m disposed of annually to drains by Addenbrookes hospital in 1987. The disposal of the radionuclide appeared to be relatively consistent at around 20 GBq per month. It was confirmed that technetium-99m could be detected at the sewage works, however, it was not possible to make estimates of removal efficiency.

The sampling carried out as part of this study was used to derive removal efficiencies, as technetium-99m was above limits of detection in effluent and sludge phases during some sampling. The range of values are given in Table 12 and the selected values are in Table 13.

i) Iodine-123, 125 and -131

Reference 2 provides a detailed review of the behaviour of iodine in sewage treatment. Some results of an early study, involving activated sludge and percolating filter pilot plants have been presented in Table 12. This shows that iodine (as iodide) in the presence of carrier solution is largely unaffected by the treatments, whereas in the absence of carrier solutions, much higher removal rates of 50% to 85% have been observed. References 26 and 27 describe experimental work that suggested that the removal efficiency of both activated sludge and percolating filter plants was about 90%. It appears that the very low removal is dependent on there being significant quantities of carrier KI present. This hypothesis is suggested by other studies reviewed in this work (which suggest removal efficiencies of 15 to 98% for iodides in natural waters). Further chemical treatment appears not to remove additional iodine. A range of removal efficiencies from 2 to 20% appear appropriate for typical sewage treatment plants.

Measurements at a sewage treatment plant also show general agreement with this range. Reference 25 followed the fate of some 158.7 GBq of iodine-131 and 13.1 GBq of iodine-125 disposed of in the Cambridge area in 1987. Because of the known disposers, it was possible to co-ordinate iodine-131 monitoring with disposals. The iodine-131 activity concentration in river waters were found to be broadly in agreement with the discharged activity. However, it was estimated that about 7% of the discharged iodine-131 activity (correcting for decay) became associated with sewage sludge. There was little evidence of volatilised iodine-131 during the process. Monitoring of iodine-125, both in parallel and at a separate time, showed that about 3% of the total activity discharged was present in sewage sludge and 19% in supernatant, indicating a partitioning of 14% to sludge and 86% to water.

Although iodine is removed poorly by flocculation, one study has shown that up to 90% of iodine activity may be removed by the use of activated carbon filtration[27]. The same study estimated that standard drinking water treatments would remove about 5% of iodine activity. During incineration only about 12% of the iodine activity was reported to remain in ash.

Iodine-123 which has a half-life of 13 h, is also affected by radioactive decay during transport and treatment. However, its chemical behaviour is the same as I-125 and I-131.

j) Gallium-67, Indium-111 and Thallium-201

No reported values were available for gallium-67, indium-111 or thallium-201. Removal efficiencies were estimated from analogues and from measurements carried out under this study, if the radionuclides could be detected. The range of values is given in Table 12 and selected values are in Table 13.

k) Americium-241

No reported values were available for americium-241. Removal efficiencies were estimated from suitable heavy metal analogues. The range of values is given in Table 12 and the selected values are given in Table 13.

3.5 Regulatory Controls on Discharges of Radionuclides with Sewage

3.5.1 Introduction

Unsealed radionuclides used in medicine, research and manufacturing for diagnosis, treatment or tracer studies will eventually be excreted or disposed of and pass into the sewage system.

The extent of control placed over the practice of discharging to sewers varies from country to country, in many cases the practice may not be acknowledged. However, wherever radionuclides are administered to individuals for treatment or diagnosis, discharges of radionuclides will occur unless waste management systems are put into place (eg decay storage, absorption or ion exchange). In many cases the radiological hazard associated with the discharge is minimal and when compared with the costs and installation, the use of management systems is not normally considered a sensible use of resources.

3.5.2 UK situation

In the UK, discharges are regulated by the Agencies, under the Radioactive Substances Act[17] using a system of disposal authorisations. Requests to dispose are made by individual organisations and are accompanied by a radiological assessment. For relatively small discharges, simple, pessimistic assumptions may be used in the assessment, such as an assumption that the effluent is used by someone as their only supply of drinking water. If there is any indication that the radiological consequences would be unacceptable, then realistic assessments are required.

When granting and reviewing authorisations the Agencies consider whether there is a case for reducing discharges through the installation of waste management systems (e.g. decay storage). In a few situations in Northern Ireland decay storage is employed by major ablation treatment hospitals.

3.5.3 International situation

The international situation with regard to permitting discharges of radionuclide is generally unclear. Few countries appear to acknowledge that radionuclides are discharged to sewers even from patient treatments. The Financial Times[28] reported in 1998 that *'the UK is the only country in the EU to dispose of low level radioactive waste in the sewers'*. This statement is untrue. The majority of EU countries utilise radionuclides in treatment and diagnosis, and as a result discharges of radionuclides to sewers in wastes excreted by patients is inevitable. Even

if decay store methods were available and utilised 100% to retain sewage for the required several months to reduce discharges of radionuclides such as iodine-131, radionuclides such as carbon-14 and tritium which have long half lives would be totally unaffected, whilst radionuclides such as iodine-125 and strontium-89 would only be partly affected. In Italy, for example, studies have been carried out into the fate of radionuclides discharged into sewage systems from nuclear medicine departments[29]. Various radionuclides including technetium-99m, iodine-131, gallium-67, thallium-201 and iodine-125 were all found in the sewage wastes, arising from excretion of activity by patients undergoing therapy. In addition, studies were carried out in Sweden of activity concentrations of various radionuclides from radiopharmaceutical releases to sewer systems[30]. These two studies show that rather than the UK being the only country to dispose of low level radioactive waste, it is more likely that the UK is the only country to acknowledge and authorise the practice.

In the US, activity concentration limits for effluents and for releases to sewers[31] for a range of radionuclides have been set. The levels are related to a dose criteria of 0.5 mSv y^{-1} and assume an oral intake of effluent of 7.3 10^6 ml.

In Canada, discharges of radionuclides into sewers are permitted under local regulations, which set permissible concentrations for each radionuclide in effluent. The permissible concentration of iodine-131 is 370 Bq l^{-1}, which is expected to be reduced to around 200 Bq l^{-1}.

4 SEWER ASSESSMENT MODEL

4.1 Introduction

A computer model was developed in this study to assess the fate and impact of radionuclides discharged into public sewers (Sewer Model for the Assessment of Radionuclide Transport, SMART). SMART models the 4 main stages of sewage collection, treatment and disposal: transport of the sewage to the treatment works, treatment of sewage at the works, treatment of solids and sludge at the works and final disposal of treated sewage effluent and sludge. The model calculates effective doses to workers during the first three stages and effective doses to members of the public from the last stage. For convenience "effective dose" has been abbreviated to "dose" for the remainder of the report. The exposure pathways considered by SMART are external irradiation, where the geometry of the source is considered; inhalation of sprayed or re-suspended material; and inadvertent ingestion. Skin contact has been considered unlikely since the nature of the material being handled means that protective equipment is likely to be worn at all times.

The transport stage is divided into two, the collection of sewage from premises in small diameter pipes of radius 0.2 m and the transport of effluent gathered from a number of premises in larger man accessible trunk pipes. Exposure of workers unblocking the small pipes and sewer gangs spending time in cleaning and repairing man accessible pipes is considered.

SMART considers all four steps in the treatment of effluent described in Chapter 2: preliminary treatment (grit removal and screening), primary treatment (suspended solid sedimentation), secondary treatment (aerobic digestion and settlement of bacteria) and tertiary treatment (removal of dissolved chemicals such as phosphates and heavy metals). Exposure of workers to the separated solids, sludges and treated effluents is considered at each of these steps.

In the treatment of solids and sludge stage SMART assumes that the solids removed from the effluents are treated separately. The grit and floating material arising from preliminary treatment are assumed to be disposed of without any treatment. Sludges arising from primary and secondary treatment are joined and treated in one of two ways (depending on final disposal route). The first treatment option considered in the model (for disposal of sludge to sea, landfill and farmland) contains 2 main stages, initial sludge processing and sludge stabilisation. There may be a third optional stage involving dewatering. The second treatment option (for sludge incineration) considers two main stages, initial sludge processing (settling) and dewatering and drying. Exposure of workers to the sludges at each treatment stage and during incineration and ash handling is also considered.

The final stage in the model is disposal of treated effluent and sludge. Disposal of treated effluent normally involves discharge into inland or coastal water bodies. Exposure of the public from these disposal routes is considered in the model. The four sludge disposal routes: disposal offshore, disposal to landfill, application to farmland and disposal by incineration (with residual

ash disposed to landfill) are considered by the model even though offshore disposal has now ceased. Exposure of the public from all these disposal routes is considered.

The sewage treatment part of the model is summarised in Figure 7. A full description of the treatment and collection of effluent part of the model is given in Appendix B. The models for assessing the radiological impact of disposal of sludges and effluents are described in Appendix C. The database is given in Appendix D.

SMART was customised for the two case studies: Beckton STW in London and Knostrop STW in Leeds. It can also be used to model other STW by setting appropriate input data.

4.2 Comparison with Other Sewer Models

The sewer model (SMART) developed for this project has been compared with three other models: a simple sewer model developed at NRPB to calculate Generalised Derived Constraints for discharges to public sewer systems; the model used by Electrowatt Engineering Services in 1993 to study the implications of releases of radionuclides to sewer systems; and a preliminary model for radionuclide migration in urban environment and drainage systems developed for the EC MOIRA project, mainly for caesium-137.

4.2.1 Generalised Derived Constraints model for sewers

Generalised Derived Constraints (GDCs) for sewers[32] are estimates of the amount of activity discharged to the sewers, which, if not exceeded mean that it is very unlikely that sewer workers or members of the public would receive an effective dose above the maximum dose constraint of 0.3 mSv y^{-1}. A simple sewer model was developed to evaluate GDCs for sewer discharges. The sewage collection and treatment system was assumed to serve a small rural community (population: 500), and was characterised by low volumetric effluent flow. The model assumes instant transport of effluent and radionuclides to the works, where effluent was treated to remove suspended solids and biochemical oxygen demand through primary and secondary treatment. Treated effluents were then assumed to be discharged to a river of low volumetric flow, and treated sludges applied to nearby farmland.

Discharges of radionuclides to sewers were assumed to continue for 50 years under the same conditions, and the resulting build up in the river and farmland environments were modelled. The GDC model does not explicitly consider partitioning of radionuclides, assuming that either 100% of the activity is associated with the sludges or 100% with the effluent. The GDC model is therefore simpler and more cautious than SMART as it does not model radionuclide transport in pipes, makes simplifying cautious assumptions about radionuclide behaviour at the works, and considers only one disposal route for effluent and one for sludge.

Doses for GDCs were calculated for three separate exposure groups. Sewer workers were considered to spend a working year at the sewage works, and were exposed to sludges where radionuclides had been concentrated. The second group were members of the public who are exposed to river water that has received treated effluent. The third group were assumed to live

on land treated repeatedly with sewage sludge and to consume animal products produced from the treated land. Foods consumed were assumed to have been produced within either the river catchment or on treated agricultural land, and intakes were assumed to be at critical group levels. The SMART model is more complete, considering exposure during transport, at the various stages of treatment and from all the possible disposal routes. The effluent disposal to river and sludge to land modelled by SMART is the same as in the GDC model. SMART considers two sets of individual doses to a hypothetical farmer. One set assumed that the farmer produced all his own food and consumed them all at critical group rates. The other set assumed that the farmer consumed all foods at mean consumer rates with cereals not being consumed at all. This approach was taken for modelling reasons and it differs from standard NRPB methodology. The doses to farmers given later in the report are those that result from the assumption of critical group intake rates.

Doses to workers in the GDC model were calculated on the basis of 2000 h y^{-1} spent at the works, with 1000 h y^{-1} above sludge tanks, an inhalation rate of 0.83 m^3 h^{-1}, an ambient dust loading factor of 1 10^{-7} kg m^{-3} and an inadvertent ingestion rate of 5 10^{-6} kg h^{-1}. SMART considers workers spending 1000 h y^{-1} treating sewage or sludge, a working adult inhalation rate of 1.18 m^3 h^{-1}, an inadvertent ingestion rate of 1.25 10^{-6} kg h^{-1} and dust loadings of 1 10^{-7} kg m^{-3} for ambient conditions and 1 10^{-5} kg m^{-3} for dusty environments. Characteristics of the sewage effluent, sludge and treatments assumed in the GDC model are summarised in Table 14. The nuclides considered included Sr-89 and Sr-90, Ru-106, I-125 and I-129, Cs-134 and Cs-137, and several isotopes of plutonium, americium and curium.

The GDC calculations were based on the assumption that the doses incurred by workers at the sewage treatment works were higher than those incurred by sewer workers in large pipes. This assumption was confirmed by using the GDC parameters in SMART which give doses to workers at the sewage treatment works that were an order of magnitude higher than the doses to workers in large pipes. This is to be expected given the higher effective occupancies at the works, and the media to which workers are exposed at the works, which includes exposure to separated dewatered sludges where the activity concentration is significantly enhanced.

In conclusion, the model developed for the purpose of calculating GDCs is much simpler than the one described in this project and does not consider all disposal routes for effluent and sludge. The parameter values were chosen to be conservative whereas SMART aims to be more realistic. The GDC model was only designed to make general estimates, applicable to small, rural works, for which it is appropriate to look at the practice of spreading of sludge on agricultural land.

4.2.2 Electrowatt Engineering Services study

The Electrowatt Engineering Services (Electrowatt) study presents a fairly complex and comprehensive time-dependent model[9] of the transport of radionuclides through a sewer system (pipes). This model takes account of the sedimentation and re-suspension of solids in the pipes,

depending on the velocities of the water and associated solids, the slope of the pipes and the additional water from side channels. The purpose of the model is to calculate the concentration of suspended solids in water, the concentration of contaminant in the solution (dissolved) and the concentration of contaminants on suspended solids at any point of the system and any time. Activity concentrations on suspended solids and in solution in the raw sewage are estimated using K_d values. The Electrowatt model for transport of radionuclides in pipes is more complicated than SMART.

The Electrowatt study considers the following nuclides: H-3, C-14, Co-57, Co-60, Zn-65 and I-125. Calculations carried out in the study using various K_d values (Table 15) indicate that equilibrium is reached within a few hours of discharge, and equilibrium is assumed from the arrival of sewage at the works. The SMART model does not consider partitioning during transport, assuming that the solids and liquids travel at the same velocity and arrive at the works together. The Electrowatt model considers the effects of sewage and sludge treatments by recalculating the activity concentrations in the final sludge with a new suspended solids content. At the end of treatment, the activity concentrations in sludge for 1 MBq d^{-1} discharged are given. The model was used to assess doses arising at the Maple Lodge sewage works.

SMART was also used to model Maple Lodge with the same discharge rate of 1 MBq d^{-1}, a throughput of sewage effluent for Maple Lodge, (1.3 10^5 m^3 d^{-1}), and with removal efficiencies (to sludge) of 20% for tritium and 10% for iodine-125 and carbon-14. Activity concentrations in sludge (20% solid) are given in Table 16. The SMART results are within an order of magnitude of the Electrowatt results for H-3, a factor of 2 for carbon-14 and in good agreement for iodine-125. The differences are most likely to be caused by differences in the modelling of the uptake of tritium and carbon-14 onto solid in the two models.

Exposure of workers in pipes and at the works are not considered in the Electrowatt study. Public doses were calculated for the following sludge disposal options: sea disposal, agricultural land fertiliser, garden composting, landfill, incineration and land reclamation. Electrowatt note that sea disposal gives 'very much smaller radiation doses' than some other options, but the only option that they give results for and discuss further is the fertilization of farmland. For farmland treatment it was assumed that 100 t ha^{-1} is ploughed in land, in a single application, immediately after sludge production. The pathways considered are external exposure, ingestion of soil, inhalation of re-suspended activity, ingestion of contaminated plants and animal products. These pathways are similar to those considered by SMART. The overall maximum annual effective doses to members of the public from application to land using the Electrowatt and the SMART model are presented in Table 17, for a release rate of 1 MBq d^{-1}. Agreement between the two models is reasonable, SMART predicted doses from iodine-125 and tritium are ten and five times lower than the Electrowatt model respectively, whilst doses from carbon-14 are approximately the same. The main differences are expected to arise because of the modelling of radionuclide transfer into plants and animal products. The Electrowatt study has adopted equilibrium concentration ratios, whilst SMART uses dynamic transfer models.

4.2.3 MOIRA model

The model presented by the Spanish team is part of the EC MOIRA[33] project, which is aimed at evaluating the consequences of radioactive contamination of aquatic ecosystems, and the possible countermeasures. The model developed for MOIRA simulates the movement of radioactive material deposited on urban surfaces and released to drainage systems. It contains two sub-models: one describing the radionuclide behaviour in urban systems, and the other in the drainage system.

The urban part of the model describes urban decontamination after the deposition of radioactive material and will not be further discussed here.

The sewer component of the MOIRA model refers only to Cs-137, with some consideration of I-131 and Sr-90. The suspended solids load in incoming sewage is considered to be 160 mg/l (typical values for Spain being 150 to 400 mg/l), and the organic load is 235 mg/l (150 to 400 mg/l). These values are similar to the assumptions in SMART of 500 mg/l (150 mg/l after primary treatment).

The water treatments considered in MOIRA are very similar to those in SMART, preliminary treatment, primary sedimentation, biological digestion and secondary sedimentation. Typical sludge treatment data are shown in Table 18. Sludge treatments are very similar to those in SMART as summarised in Table 19. The disposal options considered by MOIRA are the recycling of sludge as fertiliser on agricultural land and the disposal to landfill, either as de-watered sludge or incinerated ash. MOIRA notes that a considerable proportion of the activity entering the treatment is retained in the sludge, thus making the application of sludge to farmland a potentially harmful solution.

The influence of sewage treatment on radionuclide activity concentrations is considered in terms of radioactive decay, which is significant for radionuclides with short half lives such as I-123, I-131 and Tc-99m, adsorption onto sewer walls and the efficiency of treatments at removing specific nuclides. Processes that were estimated to have little impact on the activity concentration of Cs-137 were discarded in order to simplify the model. From various sources, iodine removal efficiency from sewage effluent after treatment is quoted as ranging from 15 to 97 %, this wide range encompasses the values used in SMART.

In an attempt to overcome the problem created by data being so limited and variable, the MOIRA model considered use of K_ds, taking account of the chemical and ionic characteristics of radionuclides. Expected average K_d values are presented as 1 10^3 l/kg for Sr, 10 l/kg for I and 1 10^3 l/kg for Cs in fresh waters, although it is noted that these expected values can vary because of the varying composition of water and sewage, as well as because of the influence of treatments. From a consideration of K_d values they concluded that strontium has a relatively low affinity for the solids phase, iodine interacts weakly with the solids phase, and caesium has a strong interaction with the solids phase with a very wide range of K_ds (2000-4000 l/kg).

Overall, the SMART and MOIRA models were found to be very similar in their analysis of sewer systems and sewage treatments. The SMART model, however, appears to be more flexible, can consider tertiary treatment and is able to address the main disposal routes for treated sludge that are available. SMART considers different nuclides to those given in MOIRA. No doses were estimated in this preliminary MOIRA report, making it difficult to compare the models any further.

5 DESCRIPTION OF THE BECKTON SEWAGE SYSTEM IN GREATER LONDON

5.1 Introduction

A total of 12 sewage treatment works (STW) of varying size serve 9.6 million population equivalent (p.e.)* in the Greater London area, their catchments are shown in Figure 8. A description of the general characteristics of the sewage collection, treatment and disposal systems of the major sewage works in London can be found in Appendix H. The important characteristics of these works are summarised in Tables 20 and 21 and Figures 9 - 11. Tables 20 and 21, and Figure 9 illustrate that sewage collection and treatment in London is dominated by three works, Beckton, Crossness and Mogden. Together these serve 2/3 of the population, account for 3/4 of the total sewage processing capacity and produce 2/3 of the sewage sludge. Of the three, Beckton is the largest, serving a third of the population of Greater London (3 million p.e.) and producing 27% of the sewage sludge. In fact it is the largest STW in the UK and among the largest in Europe.

Figure 10 shows the suspended solids in effluent discharged from a selection of works. This is a measure of the effectiveness of treatment at these works. There is no correlation between the size of the catchment and the suspended solids concentration, and the size of the works and the quality of the final effluent. Figure 11 and Table 21 show the quantity and disposal routes of sludges produced. There are two main options for sludge disposal, it is generally anaerobically digested and then either incinerated or sent to land. For the sewage treatment works near the centre of London there is limited access to farmland, leaving incineration as the preferred option for disposal.

Beckton has a number of research establishments and hospitals within its catchment using radionuclides for medical diagnosis and treatment. Beckton STW is also fully urban and has recently switched from disposing of sewage sludge to the North Sea to incineration of sludges. It was therefore chosen as the most appropriate works for more detailed study.

5.2 Sewage Collection

The run-off and effluents from central London are transferred to low level trunk sewers which run principally from west to east. Pumping stations assist the flow in these pipes. The principal pumping stations are at Abbey Mills, Greenwich and Western. Smaller outstations are controlled from these main pumping stations remotely.

Abbey Mills pumping station serves the 135 km^2 of the low-lying districts north of central London, pumping sewage to the Beckton STW. This area includes Camden, Ealing, Islington Hackney, Hammersmith, Kensington and Chelsea, Newham, Tower Hamlets, Waltham Forest

* Population equivalents include notional population to account for industrial discharges.

and the cities of London and Westminster. It has a capacity of 3.4 10^6 m^3 d^{-1}, and in storms excess flow is discharged directly into the Thames. Four trunk sewers, ranging in size from 2.7 to 3.5 m diameter, gravitate the sewage to the station. The pumping station raises the effluent by 12 m and it then travels by gravity the remaining 6 km to Beckton STW.

5.3 Sewage Treatment

5.3.1 Sewage Treatment at Beckton

The sewage collects at the Abbey Mills pumping station and then enters Beckton, where it passes through sedimentation tanks and activated sludge plant. The present consented flow rate of Beckton is 2.7 10^6 m^3 d^{-1}, which is considerably larger than the consented flow in 1978. The final effluent is discharged to the Thames tideway at about 90 % of the influent flow rate. The peak outfall flow rate is in the early afternoon. Typical suspended solids loads in out flowing sewage are about 17 mg l^{-1}, with a maximum of 130 mg l^{-1}.

Historically the sewage sludges were thickened to about 4.5 % solids and mixed with surplus activated sludge (about 6 % solids). These sludges were then processed by anaerobic digestion for about 3 weeks. The sludges were then passed to lagoons for secondary digestion (a tertiary treatment) and then disposed of in coastal waters at Barrow Deeps, in the outer Thames Estuary. On average there were four sailings per day, each carrying about 2200 wet tonnes with about 2 % solids. However, these sludge conditioning processes were replaced by incineration in 1998.

For incineration, the sewage sludge is thickened to about 32 %, this allows autothermic operation of the incinerator. Normally, the time between the start of the sewage treatment process and incineration is 2-4 days. The incinerator is expected to handle 7.0 10^4 tonnes of dry solid (tds) y^{-1} of sludge. Around 20% of the sludge remains as ash, which will be sent to a landfill at Beddington.

6 RADIONUCLIDE DISCHARGES IN LONDON

6.1 Introduction

An application to dispose of radioactive waste must be made by the disposer to the Agencies. The Agencies then decide whether or not to issue an authorisation. Under Section 18 of RSA 1993[17] the relevant water company should be consulted by the Agencies if the disposal is likely to require that special precautions be taken by the water company. However, the water companies do not have any direct control over what radioactive material may be discharged into their sewers.

The actual disposers within the catchment of Beckton were identified and the total discharges to Beckton were assessed, using publicly available sources of information on radionuclide discharges to sewers. A detailed analysis of the discharges to this works (identifying the most important radionuclides and premises) was carried out.

6.2 Sources of Information on Radioactive Disposals to Sewers

6.2.1 Information available from authorisations

Regulatory permission to dispose of radioactive aqueous waste to sewer from a specified premises is contained within a certificate of authorisation issued to a user of radioactive material who accumulates and disposes of radioactive waste. The Agencies use an authorisation to regulate all accumulations and disposals of radioactive waste from the users' specified premises. The authorisation certificates are generally held on public registers (although there is no legal requirement to do so) and are therefore accessible to the public. A certificate authorising the disposal of aqueous waste contains the following information on discharges: the aqueous disposal route, and the monthly disposal limits (in Bq) for specified radionuclides or groups of radionuclides. The disposal limits are set by the Agencies after receipt of an application for an authorisation from the user (including a radiological dose assessment to the most likely exposed individuals from the disposal). The radionuclides disposed of to sewers are almost always low radiotoxicity radionuclides that emit beta particles and/or gamma-rays. As well as specified radionuclides there is usually a group category of "any other radionuclide except an alpha emitter (in total)". Very occasionally a category of "alpha emitters except natural uranium and natural thorium (in total)" may be specified.

6.2.2 Other publicly available information

The applications for authorisations contain more detailed information than the authorisations themselves. Like the certificate of authorisation the application is generally held on Public Registers, but there is no legal requirement to do this. In the application the Agencies request the details that ensure the effect of any disposal of aqueous waste will not cause an unacceptable risk through the route of disposal. The total expected monthly volume of radioactive

effluent, the total volume of all effluent from the organisation and the chemical and physical nature of the wastes must all be included in the application. The receiving STW and location of its effluent discharge are also stated. The actual disposals must be assessed, although in practical terms the amount of activity disposed of may only be an estimate. An assessment of the potential doses to the most exposed individuals from the proposed disposal is also required as part of the application.

Conditions in the authorisation require that records must be kept of the activities of radionuclides disposed of to sewer and by other routes, and that an annual summary of radioactive waste disposals must be sent to the relevant Agency by the 31st January of the following year. These enable a comparison to be made between the authorised limit and the actual disposal.

6.3 Identification of Disposers in Beckton Catchment.

The details of the premises authorised to accumulate and dispose of radioactive material in Greater London were collected with the assistance of the Environment Agency. About 130 authorised premises were located. The geographical position of each disposer was identified and the STW at which their effluent is treated was identified. This was achieved by using sewer catchment maps supplied by Thames Water Utilities Ltd[34].

Figure 12 presents the total number of premises authorised to discharge (non-organic) aqueous radioactive waste to each sewage works, as at spring 1998. As may be seen, 65 of the authorised premises lie within the catchment of Beckton STW, with around 20 in both the Crossness and Mogden catchments. The remaining authorised premises are located in the catchment of the remaining STWs.

Copies of the certificates of authorisation to dispose to sewer were obtained from a public register for 52 of the 65 Beckton STW premises. The 13 authorisations that were not available were old authorisations, i.e. they had either lapsed or been revoked. Riverside STW also disposes of its sludge to Beckton, so the relevant authorisations for this catchment were also obtained. In addition, the actual recorded discharges made during 1997 were available for 24 of the 65 premises. Copies of the application for authorisation to dispose of radionuclides to sewers (which contain an assessment of the potential doses) were available for 12 premises.

6.4 Detailed Analysis of Discharges to Beckton

Disposals of radionuclides to Beckton (including the contribution from sludges from the Riverside works) given in the authorisations were analysed to establish:

- The range of monthly disposal limits and the most frequently specified radionuclides;
- The extent to which authorised disposals were used;
- The most important radionuclides authorised for disposal in terms of potential impact;

- The most important disposers in terms of the potential impact of their authorised discharges;
- A source term for the sewer assessment model SMART.

6.4.1 Analysis of the limits and radionuclides

Figure 13 presents the distribution with activity of the 243 individual monthly discharge limits given in the current certificates for all authorised premises in the Beckton and Riverside catchments. As may be seen, for most radionuclides (or groups of radionuclides) the cumulative percentage appears to indicate a general lognormal spread, with 5th percentile in the region of 6 10^6 Bq month^{-1}, 95th percentile at about 8 10^9 Bq month^{-1} and median around 3 10^8 Bq month^{-1}. It can also be seen that authorised limits for I-125 and I-131 are generally lower than for other radionuclides. Limits for Tc-99m are generally very much higher, with all but two of the 27 limits in the Beckton and Riverside catchments for greater than 10^{10} Bq month^{-1}.

Figure 14 shows the relative frequency with which authorised limits are issued for various nuclides in the Beckton and Riverside catchments. Unsurprisingly the limits most often given in certificates are for "other beta-gamma" nuclides (48 instances out of 56 certificates). The next most frequently given are limits for I-125 and I-125/I-131 (33 instances), C-14 or H-3/C-14 (28 instances) and Tc-99m (27 instances). Amongst the disposers in Central London only one organisation holds an authorisation to dispose Rb-83/84 to sewers. However, the process that generated these Rb isotopes is no longer used, so the authorisation is not currently in use. It is not likely that monitoring at the sewage works will detect these radionuclides.

6.4.2 Ranking of radionuclides and disposers

An indication of the most important radionuclides and disposers was obtained by using the authorised monthly limits on activity discharged, and the relative radiotoxicity of the radionuclides. Hence, indicative estimates of the dose from ingestion for each radionuclide and each premises were calculated, using the following assumptions:

- A person obtains all their drinking water from the undiluted effluent of the works. It has been assumed that a person drinks 600 l y^{-1} of the contaminated effluent[35], and the throughput of the works is 1.08 10^6 m^3 d^{-1}. The ingestion dose coefficients[36] used are presented in Table 22
- All premises dispose of the maximum authorised quantity of radionuclides
- Where authorisation certificates specified either "H-3 or C-14" or "I-125 or I-131", the radionuclide with the highest dose coefficient was selected (C-14 and I-125 respectively).
- In order to account for the "other beta-gamma" category it has been assumed that this is entirely I-131.

The monthly authorised limits for each radionuclide from all premises in the Beckton STW catchment are given in Table 22. The radionuclides have been ranked using the method above. As can be seen, a number of radionuclides not usually considered as significant may be important, notably Rb-83/84, Sr-89, I-123 and Y-90.

The monthly authorised disposal limits for the 20 premises with the most potentially significant discharges are presented in Table 23. It may be seen that the majority are hospitals. Of the other authorised disposers the highest ranked is the Imperial College of Science and Technology (15th), followed by EISAI London Research Laboratories Ltd, located at University College London (18th). These top 20 premises account for 95.4 % of the total radiological impact from the discharges, estimated using the total indicative ingested dose.

An external dose scoping calculation has also been carried out using disposals of iodine and "other beta-gamma" from all premises in the catchment. Iodine was used because it is responsible for the majority of the external dose and the "other beta-gamma" radionuclides have been considered to be all iodine-131. With these assumptions the 20 premises described in Table 23 account for 95.7% of the total external indicative dose.

The indicative doses for each of the 52 premises holding certificates of authorisation in the Beckton catchment and the additional 3 premises in the Riverside catchment are presented in Figure 15 to show how they are distributed. The distribution of indicative doses from each disposer is broadly lognormal, ranging between 1 10^{-14} Sv y^{-1} and 1 10^{-4} Sv y^{-1}, with an average of 1.1 10^{-6} Sv y^{-1}. It is notable that the range is very wide, since only a few premises dominate the overall radiological impact of the disposals.

6.4.3 Extent of use of the authorisation

The extent to which authorised disposal limits were used was investigated, by comparing the actual discharges reported during 1997 with the disposal authorisations. In total there were 56 actual disposals which corresponded to authorised disposal limits for Beckton and Riverside. The modal usage was around 10% of the authorised limit. The mean value was 14% with a maximum of 76%. It was difficult to distinguish trends by nuclide, as there were rarely sufficient observations. However, it was noted that typically disposals of Tc-99m were closer to the authorised limit, with a mean of 33% of the limit.

6.4.4 Establishing a source term for authorised disposals

Disposal authorisations do not always specify radionuclide specific limits for discharges. In particular, authorisations do not normally specify limits for C-14 and iodine-131 as individual radionuclides, they are usually grouped with other radionuclides. As a result assumptions have been made in the interpretation of grouped limits in authorisations.

Table 23 shows that authorisations generally include H-3 and C-14 together. The exact division between H-3 and C-14 is not known, therefore in the assessment it is conservatively assumed that the H-3/C-14 authorised disposals are all C-14. Obviously it is desirable to establish

the relative proportions of H-3 and C-14 disposed within the grouped authorisations. However, as the majority of C-14 and H-3 disposers do not occur in the top twenty establishments, a large number of authorised establishments would need to be contacted to get a reasonable picture.

The radio-iodine authorisations may be given as individual radionuclides, I-125/I-131 or 'iodines'. The disposal authorisations therefore required interpretation for the radiological assessment. Where I-125/I-131 or 'iodines' are included in the authorisation, the assessment assumes I-125, giving a total I-125 for the assessment of 1.6 10^{11} Bq/month.

Where 'other β/γ' are specified in the authorisations, I-131 has been assumed to be disposed in the assessment, (total authorised I-131 = 8.3 10^{10} Bq/month, authorised 'other β/γ' = 1.6 10^{11} Bq/month, therefore total I-131 for assessment = 2.4 10^{11} Bq/month).

The assumptions made for discharges of 'other β/γ' category will over-estimate the quantity of I-131 disposed to sewers. The over-estimate occurs because establishments which already have an iodine authorisation would not then dispose additional quantities of iodine under 'other β/γ ('other β/γ in top twenty which already have an I-131 authorisation = 9 10^{10} Bq/month). However, the assumption of I-125 disposals for any grouped iodine authorisation is likely to overestimate the I-125 disposals under this authorisation but underestimate the I-131 disposals in the iodine groups. In view of the similar levels of assumed disposals for these two nuclides, the assumptions are considered to be conservative but not overly so. The source term used to assess doses from authorised disposals is given in Table D20 in Appendix D.

6.4.5 Establishing a source term for typical disposals

The typical disposals during 1997 have been received from the top twenty establishments and the majority of the disposing establishments hold nuclide specific disposal records. The 'other β/γ' specifically described included disposals of P-32, S-35, Ga-67, Tl-201, In-111 and Cr-51. Where the radionuclides disposed were not named specifically the same assumptions were made as for authorised disposals.

Typical radio-iodine disposals of establishments in the top twenty consist almost entirely of I-131 (99%), the majority of the iodine disposals occurring from hospitals. There are three iodine authorisations under which I-131 could be disposed; authorisations for I-125/I-131, iodines or I-131. Typical disposals of I-131 are typically around 20 % of the total of all these authorisations. For assessment purposes 30% of the maximum authorised disposal limit of the specified I-131 authorisation was assumed. However, due to the large activity and infrequent nature of ablation therapies, disposals in some months are likely to be much closer to the monthly authorisations.

The typical disposals of I-125 are around 1% of authorised levels or the quantity assumed for the assessment, indicating a large over estimate of the I-125 disposals. The typical disposals of H-3 and C-14 from the top twenty establishments is around 40% of their H-3/C-14 authorisations. The typical disposals consist of 85% H-3 and 15% C-14.

Consideration of typical discharges however can only provide a snapshot of a particular year and suggest trends. Those authorised to dispose are allowed to do so up to the limits set, although the typical disposals examined suggest that in 1997 at least, disposals in Central London are generally much less (eg 20-30% occur for I-131). It should be noted that in 1997 the incinerator had not been commissioned and sea disposal was used for sludges, so the incinerator doses are hypothetical. The source term used to assess doses from typical disposals is given in Table D21 in Appendix D.

7 PREDICTED DOSES FOR BECKTON

7.1 Introduction

The sewer model (SMART) was set up to represent Beckton STW and used to provide an assessment of the doses from disposals of radionuclides to sewers in the Beckton catchment. Doses to workers and members of the public were calculated using the authorised discharge limits and typical discharges (based on 1997 disposals), summed over the top twenty disposers (see Chapter 6). Table 24 shows the authorised and typical discharges used for the assessment. It was assumed that the discharges were diluted in the average dry weather flow in the catchment. Wet weather flow and storm events were considered separately.

Four groups of workers were considered: workers dealing with a blocked pipe on the premises of one disposer, workers cleaning and repairing man accessible pipes, workers at the Beckton sewage treatment works, and workers at the incinerator plant. In addition doses to the public arising from disposal of effluent to the Thames tideway and from disposal of sludges offshore or from incineration of sludge were also calculated. The assessments were performed using the best estimate information on radionuclide behaviour during sewage treatment. In all cases, doses were calculated to members of the critical group, representing the most highly exposed group.

Input data specific to the Beckton Sewage Treatment Works case study are presented in Appendix D, Tables D17 to D23.

The calculated doses were compared with the annual dose limit for members of the public of 1000 μSv/y[37], the maximum public dose constraint of 300 μSv/y[37] and the threshold for optimisation of 20 μSv y^{-1} [38].

The annual dose limit for members of the public[37] applies to the sum of all the doses incurred as a result of exposure to man made sources of radiation, (other than medical exposure). The dose constraint is a restriction on the annual dose to an individual from a single source, such that when aggregated with doses from all sources (excluding natural radiation and medical procedures) the dose limit is not likely to be exceeded. The dose constraint does not replace the requirement on operators to optimise their use of sources and management of practices so that exposures are kept as low as reasonably achievable. Dose constraints lower than the maximum may be set by appropriate government departments, where such doses are readily achievable. The threshold for optimisation[38] is a lower bound introduced by the government, such that below this level regulators should not seek to secure further reductions in the exposure of members of the public, provided that they are satisfied that the operator is using the best practicable means to limit discharges. The risk to an individual receiving a dose of 20 μSv y^{-1} over his or her lifetime is broadly equivalent to an annual risk of death of 10^{-6} (one in a million).

7.2 Exposure of Workers from Blocked Pipes

Table 25 presents the doses received by workers repairing a small pipe, serving one premises only. To illustrate the doses that might be incurred the assumptions about activity concentrations within the pipe were based on disposal authorisations for one of the larger disposers in the catchment. The workers are considered to be unblocking a pipe that serves toilets used by patients who have received large doses of I-131 during thyroid treatment. The blockage is assumed to comprise 20% of one day's excretion of I-131 from one patient (assuming total excretion over 2 days), plus radionuclides present from one day's continuous use in tracer or laboratory work. For an administration of 10 GBq of I-131, the amount of activity present in the blockage would be 1GBq. For the other radionuclides disposed, a fraction of 1 day's discharge was assumed to be in the pipe. For Tc-99m, 80GBq may be disposed per month. This is disposed of fairly continuously throughout the month, (since laboratories are unlikely to work at weekends one month has been assumed to be 20 working days) so during 1 day ~ 4GBq of Tc-99m is disposed. Assuming a 10 hour working day and 2 hour's build up in the pipe there would be ~0.80GBq of Tc-99m in the pipe. The same argument implies that there would also be ~0.09GBq of 'other' radionuclides in the pipe.

The activity was assumed to be a point source for external dose and diluted in 2 m^3 of liquid for internal exposure. A delay time of 2 hours between the time of discharge and when the repair work is being carried out is assumed, since it would take some time for somebody to be called in to carry out the repair. It has been assumed that workers spend 2 hours fixing blocked pipes, obviously this will depend on how serious the blockage is and the nature of the items blocking the pipe. Two hours would be conservative if the pipe was blocked only in one small area but if there were any structural damage to the pipe or the drain had collapsed, fixing it could take much longer. The doses are presented as a dose per event because small pipes could become blocked at any number of times during a year and may not always be fixed by the same worker.

The dose is estimated to be 108 μSv each time a pipe is unblocked, arising from external exposure; estimates of doses arising from internal exposure are negligible compared with external exposure. It should be noted that this exposure pathway assumes that the worker unblocking the pipe is unaware of the radioactivity present. Within hospitals however, small-bore sewer pipes from active drains are labelled as possibly carrying radioactive materials and carry details of who to contact in case of blockage or leakage. When a blockage occurs, the local RPA would be involved in assessing the quantity of activity present and planning a scheme of work for remedial actions, prior to the work being undertaken. Hospital RPAs have suggested that for such remedial actions the aim would be to keep the dose below ~100-200 μSv, similar in scale to the dose calculated by the SMART model.

7.3 Exposure of Workers in Man Accessible Pipes

Table 24 presents doses to workers in large pipes, based on the models and data in Appendices B and D. Two cases were considered: disposals at the authorised limit and disposals at typical levels. Maximum individual doses are 12 μSv y^{-1} and more typical doses are 2 μSv y^{-1}; both are below the annual dose limit for members of the public of 1000 μSv y^{-1}, and below the threshold for optimisation[38] of 20 μSv y^{-1}. External dose makes the most significant contribution to the total dose; contributions from inhalation and ingestion are insignificant. The workers are assumed to spend 1600 h y^{-1} exposed to the discharges from all sites within the catchment diluted in the total dry weather flow to the sewage works (1.08 10^6 m^3d^{-1}).

7.4 Exposure of Workers at the Sewage Treatment Works

The individual doses to workers at the sewage treatment works are presented in Table 26. The time spent by workers, and the partitioning of the radionuclides at each stage of the treatment process and the resulting activity concentration affects the doses. The data are given in Appendix D.

The estimated individual dose at the works based on typical disposals is 19 μSv y^{-1} and the maximum individual dose, based on authorisations is 210 μSv y^{-1}. Both are summed over all the stages of sewage treatment including incineration. The total dose is dominated by the external dose, the contribution from inhalation and ingestion is very small. The external dose occurs mainly during sludge treatment at the incinerator, and the nuclides responsible for most of the dose are I-131 and Rb-84, contributing 12 and 6 μSv y^{-1} respectively to the typical dose and 42 and 160μSv y^{-1} to the dose at authorised discharge levels. Given that discharges of Rb-84 are no longer thought to occur, the typical dose may be nearer to 13 μSv y^{-1} and the maximum dose only about 50 μSv y^{-1}.

Some workers at the STW spend all of their time in close proximity to de-watered sludge. For these workers the dose can be up to 45 μSv y^{-1} from typical disposals and 207 μSv y^{-1} from maximum authorised disposals. I-131 and Rb-84 contribute 42 and 2 μSv y^{-1} respectively to the typical dose and 153 and 49μSv y^{-1} respectively to the dose at authorised discharge levels.

The estimated doses are below the dose limit for members of the public of 1000 μSv y^{-1}, the maximum public dose constraint of 300 μSv y^{-1}, and average individual doses in the UK from natural radiation[39] of 2200 μSv y^{-1}.

7.5 Exposure of Members of the Public

Table 26 also presents the estimated individual doses to the public resulting from disposals of effluents and sludges. For effluent disposals to the Thames Estuary the estimated dose from typical disposals is 0.6 μSv y^{-1} and from maximum possible authorised disposals is 31 μSv y^{-1}; they are both the same for discharges over 1 and 50 years. The majority of the dose arises from carbon-14, iodine-131 and phosphorus-32. The most highly exposed group is assumed to

consume aquatic foods caught locally at 97.5th percentile rates, spend high amounts of time on the banks of the estuary and to be exposed to seaspray. The dose to the majority of the population in the vicinity of the Beckton outfall will be lower than these by a factor of 3 or more, depending on how much locally caught fresh aquatic food is consumed.

Estimated doses from the atmospheric release of radionuclides during sludge incineration are all below 2 μSv y^{-1} for disposal over 1 and 50 years. The most highly exposed group is the farmer occupying, working and living off produce from a farm around 13 km from Beckton. Doses to residents living near (1.5-2 km) the incinerator, consuming some foods produced in their gardens at critical rates are between 0.07 and 0.5 μSv y^{-1} depending on whether the typical or total possible authorised disposals are considered. Average members of the population living within 5 km of the incinerator, eating some locally produced foods will receive doses of less than 1 μSv y^{-1}.

Estimated doses to members of the public from disposal of sludge ash to landfill are 0.04 μSv y^{-1} for one year and 2 μSv y^{-1} for 50 years of authorised disposals; for typical disposals the estimated doses are 2 10^{-4} μSv y^{-1} for one year and 0.01 μSv y^{-1} for 50 years of disposals. In both these cases, the dominant nuclide is C-14 via leaching of the ash and migration through the environment and into foods and drinking water. Due to the time taken for leaching and migration (several years) all other nuclides have decayed to insignificant levels. Intrusion into the landfill site was not considered.

Other routes of disposal or re-use of sludge ash have not been assessed because they are generally at an experimental stage. The main alternative to landfill disposal is the inclusion of sludge ash in building materials. This is unlikely to prove a significant source of exposure because building materials are only likely to give rise to external doses and the radionuclides responsible for the majority of the external dose are Tc-99m and I-131. Due to their relatively short half lives the activity of these nuclides will have decayed to insignificant levels in the time taken to manufacture building materials and then build a house.

All the estimated doses from the disposal of effluents and sludges are well below the dose limit for members of the public of 1000 μSv y^{-1}, and the maximum public dose constraint of 300 μSv y^{-1}. Doses due to incineration and disposal of ash are below 0.2% of the dose limit for members of the public and they are well below 20 μSv y^{-1} (the threshold for optimisation[38]) and 10 μSv y^{-1}, the level considered to be trivial by IAEA[40]. The doses from 1 year and 50 years of disposals are similar because the dominant radionuclides have a short radioactive half-life and hence do not build up in the environment. In the landfill case the dominant nuclides are H-3 and C-14 which have longer half lives and hence can build up over a 50 year disposal period.

Table 27 presents the collective doses to the UK population truncated at 500 y for 1 year and 50 years disposal of sludges and effluents. For the maximum authorised disposals, the collective dose per year of disposal of effluent is 0.1 manSv and the collective dose from incineration of sludge is 0.76 manSv, (for atmospheric release) and 9.2 10^{-3} manSv (for the disposal of ash to landfill). For typical disposals, the doses are 1.5 10^{-3}, 8.4 10^{-2} and

4.2 10^{-5} manSv respectively. The radionuclides responsible for most of the collective dose are C-14, I-131 and I-125. The collective doses meet the IAEA's criteria for a trivial collective dose[40] as they are all below 1manSv y^{-1}. Collective doses to the UK population truncated at 500 y from 50 years of disposal show the same pattern as for one year's release but are 50 times higher.

7.6 Wet Weather Flow

The total flow of sewage to Beckton STW can rise to 2.70 10^6 m^3d^{-1} depending on the weather. SMART was used to estimate the effect this may have on all of the doses discussed so far. It was found that the estimated doses were largely unaffected. The inhalation and ingestion component of doses to workers in large pipes and at the works were slightly lower. However, the majority of the dose to workers arises from external dose, and the majority of the external exposure arises from radionuclides in sludges and sewage cake, and is controlled by the activity concentration in the solids. Total flow was assumed not to affect the activity concentration in the sludges and cake because the total amount of suspended matter carried by the increased flow would largely be unchanged compared with dry weather flow.

Predicted doses to the public are not affected by increased flow through the works. This is because the doses are directly related to the route of discharge and the total quantity of radionuclide discharged into the environment. These are independent of concentration and are therefore largely unaffected by changes in flow through the works.

7.7 Storm Events

In times of very severe weather conditions when the STW is in danger of flooding, the flow to the STW may be discharged directly to the receiving watercourse without undergoing treatment. Exposures of members of the public from this kind of event have been modelled by assuming that all of the activity is discharged into the receiving water body.

The only doses that are considered are those to the public from the disposal of effluent. There will be no doses to workers since the STW is effectively bypassed and it is assumed that there will be no workers in large pipes during a storm. The dose received by members of the critical group as a result of a storm event lasting a few hours at authorised levels of disposal has been estimated to be 0.02 μSv. The collective dose is 5.3 10^{-5} manSv. Assuming 10 such events a year, the total annual dose would be of the order 0.2 μSv y^{-1}. This dose is below the level considered by IAEA as trivial.

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8 MONITORING AT BECKTON SEWAGE TREATMENT WORKS

8.1 Introduction

The aim of the monitoring programme was to determine whether the concentrations of radionuclides in the effluents arriving at the sewage treatment works and in the sludges and treated effluents were detectable. If measurable values could be obtained, then the results would also assist in the validation of the computer model, SMART, used to assess the radiological impact of disposals to Beckton STW (Chapters 4 and 7). It was therefore important to carry out the monitoring programme at times when radionuclides were passing through the treatment plant.

Initially the disposal authorisations were inspected to determine which radionuclides were likely to be present and which could be above detection limits. The latter was determined by estimating the activity concentrations in effluents and sludges from the discharge limits given in the authorisation and assuming continuous discharge into the volumetric flow of effluent passing through Beckton. Detection limits for sludges and liquids were estimated based on instrumental performance and likely sample sizes and configurations. A comparison of the estimated detection limits and the possible activity concentrations in effluents and sludges indicated that I-123, I-125, I-131, P-32, H-3, C-14, Rb-83, Rb-84, Tc-99m, S-67, Ga-67 and In-111 could be present at activity concentrations above the limits of detection. Measurements of activity concentrations in the samples taken therefore concentrated on these 12 radionuclides.

The observed activity concentrations at any given time will be influenced by two main factors: the rate of flow of effluent through the works and the amounts of activity discharged at a particular time. Values in winter could therefore be lower than in summer, because of the increased input from storm-water. To investigate this possibility, sampling programmes were mounted in August 1998 and January 1999. Values will also be lower at times when discharges are reduced, eg at weekends, early in the week, and during holidays. The sampling was planned to avoid times when discharges reaching the works were expected to be low. The temporal patterns of disposal were therefore established before sampling occurred.

8.2 Temporal Patterns of Disposal

It is important to note that radionuclides may be used and disposed on either a continuous or intermittent basis (the latter during therapy). In the latter case, all of the authorised monthly allocation may be utilised within a few days, and so the resulting discharges are likely to arrive at the STW in a relatively short pulse which in turn could make detection and interpretation difficult. If, for example, sampling took place while a pulse of activity was passing through the STW, then the results could be much higher than those expected on the basis of continuous discharges (Section 8.1). Conversely, if sampling took place during the time periods when disposals were not being made, then the radionuclides may not be detected at all. Consequently,

it was important to establish the temporal patterns of disposal for those facilities that contributed most to the throughput of radionuclides at the Beckton STW.

The main disposers of radionuclides to the Beckton STW were identified and a significant number were contacted to determine whether there were any temporal patterns to their disposals. From this investigation, it was established that the majority of disposals, including those of ^{99m}Tc, occurred fairly continuously during normal working hours. However, some of the hospitals contacted undertake thyroid ablation therapy. As a result, a significant part of one month's disposal authorisation of isotopes of iodine is used and discharged within a short time period. During treatment, each patient is administered with between 3 and 10 GBq of ^{131}I. In general, the treatments are administered in the afternoon, and so the first disposal to sewer should occur a few hours later as the iodine is excreted. Fifty percent of administered iodine is excreted within 24 hours, and about 90% is excreted within 48 hours of intake. The frequency that hospitals undertake thyroid treatments varies from 1 to 2 treatments per week to 3 to 4 treatments per year. The individual disposal authorisations give an indication of how frequently these treatments will be carried out in each hospital.

8.3 Sampling Strategy

Since the majority of the radionuclides are disposed gradually during laboratory hours, sampling was carried out over a 24 hour period in the middle of a working week: the summer programme took place on Tuesday and Wednesday, 11 and 12 August 1998 and the winter programme on Wednesday and Thursday, 28 and 29 January 1999. Sampling also allowed for transit time from the disposer to the works during a working day, and through the works.

8.3.1 Sampling at the sewage works

The monitoring of influent to Beckton was planned to cover a 24 hour period. In practice the sampling occurred over 10 hours only and outside the times when discharges were likely to arrive at the works.

- **August sampling**

To coincide with the periods covered by the bulked samples of raw sewage, samples of sludge were collected from the sludge output pumps of the primary settlement tanks at 1500h and 2000h on 11 August. A sample of the activated sludge was taken early on the morning of 12 August.

- **January sampling**

It was decided not to sample the raw sewage input in January because of problems in obtaining representative samples; the input of activity could in any case be more conveniently estimated from measurements of sludge and liquid from the primary settlement tanks.

Samples of the liquid and sludge were taken from the output of the primary settlement tanks at 10:45 on 27 January. Similarly, samples of the liquid and return activated sludge were taken from the secondary settlement tanks at 16:00 on 27 January. The outfall was continuously sampled from 11:30 on 27 January to 00:30 on the following day. These samples were bulked to cover the periods 11:30 – 15:30, 15:30 – 19:30 and 19:30 – 00:30. After measurement of the gamma ray emitting isotopes, the samples were bulked again to give a single composite sample for radiochemical analysis.

8.3.2 Sampling at the incinerator

Incinerator inputs (feedstock) comprise sewage sludge cake, whilst outputs are residual ash, off gases, and off gas scrubber wastes. The feedstock for the incinerator is prepared by dewatering sludges from 2-5% solid to up to 30% solid using batch mode filter presses. The pressroom contains 8 presses, each one being made up of 158, 2 m square press plates. At the end of each filtering cycle, each plate contains a filter cake about 70 mm thick. The resultant "cake" is then burnt at 750 – 800 °C in multiple fluid bed incinerators. The residual ash is continually removed by suction, passed through precipitators and then deposited in hoppers. The hoppers are emptied directly into trucks.

The off-gases from the incinerator pass through scrubbers in the stack. Most of the liquids from these are recycled around the scrubbing system, but a proportion is discharged back to the sewage works.

- **August sampling**

A sample of the 'cake' from the filter press and the ash was taken on 12 August. Unfortunately, the incinerator had been shut down on the previous day and so the cake was at least one day 'older' than normal. The only sampling point that can be used for the ash was at the outlet of the hopper. Consequently, it was not possible to ensure that the ash collected derived from the batch of sludge sampled earlier. Therefore the ash is likely to be from various batches.

- **January sampling**

A sample of the 'cake' from the filter press and the residual ash was taken in the manner described for August. The sampling date was 28 January. At the same time samples were taken from the liquors in the waste gas scrubbers.

8.4 Sample Preparation and Analysis

For gamma-ray spectrometry, reliable results can only be obtained if the sample remains homogeneous during the entire counting period, which may be up to 16 hours. This is a particularly important factor in the present study because sludges contain a high moisture content and liquids often contain a high proportion of suspended solids, while the concentrations of radionuclides in the liquid and solid phases may be very different. For this reason, all liquid samples were filtered on site and the residue and filter paper discarded. If necessary, on return to the laboratory sludge samples were concentrated to a stable paste by centrifugation. The incinerator cake and one sludge sample were sufficiently solid to be counted as collected. All samples were then packed into standard containers and gamma-ray emitting radionuclides determined using high-purity Ge detectors housed in a purpose built facility and appropriately calibrated. These measurements took place at the Board's Chilton laboratory. Separate subsamples were sent to the Board's Glasgow laboratory for the radiochemical determination of ^{3}H, ^{14}C, ^{35}S, ^{32}P and ^{125}I. All these analyses are within the scope of the relevant laboratory's formal United Kingdom Accreditation Service (UKAS) accreditation.

8.5 Results and Discussion

The results for August and January are shown in Tables 28 and 29 respectively. Data are presented for radionuclides that are authorised for disposal to sewers and for any other radionuclides that were measurable using high-resolution gamma ray spectrometry. In practice, radionuclides of natural origin such as ^{7}Be, ^{40}K and members of the uranium and thorium decay chains such as ^{235}U, ^{214}Bi and ^{226}Ra and ^{228}Ac were detectable. In addition, ^{137}Cs could also be detected. This radionuclide is widespread in the environment because of fallout from the testing of nuclear weapons in the atmosphere and from deposition after the Chernobyl accident.

For the purposes of the subsequent assessment, the results for sludge have been expressed in terms of wet mass, ie as sampled. The dry matter content of each sample is given in the relevant table.

It should be noted that apparently similar samples may have different limit of detection values reported for the same isotope. This is due to different measurement conditions such as delay from sampling to measurement and counting times and should not be considered as indicative of different actual concentrations in the samples.

Materials with radionuclide activity concentrations below 0.4 Bq g^{-1} are exempt from control under the Radioactive Substances Act 1993 (RSA93)[17] by the Radioactive Substances (Substances of Low Activity) Exemption Order 1986[41], and so 0.4 Bq g^{-1} is often used as the effective definition of a radioactive material. At Beckton, three radionuclides had activity concentrations above 0.4 Bq g^{-1} in filter cake entering the incinerator and in residual ash. These concentrations arise partly because of the concentrating effects of volume reduction during sludge dewatering. Of the radionuclides involved, Be-7 and K-40 are of natural origin while only I-131

results from authorised discharges into the sewerage system. Since the sludge contains I-131 in consequence of the disposal of radioactive waste, it is not considered radioactive material under the definition given in the Act. The means of control is via the authorisation of the primary disposer and there is no legal basis for the authorisation of the incineration of sewage sludge.

Data on the throughput of the STW were provided by the operators; only information on total flows was available. The flow through the STW was 998,000 m^3 on 12 August 1998 and 1,157,000 m^3 on 28 January 1999. The amount of sludge sent for incineration each day is about 800 tonnes, which corresponds to only about 0.06% of the total raw effluent throughput, but equates to the solid content of the influent (0.05-0.1%). Of the radionuclides of interest, only ^{131}I was detectable in both the liquid and solid phases in the primary and secondary settlement tanks in January 1999 (Table 29). On the basis of the observed activity concentrations, almost all of the activity entering the STW remained in the liquid phase. Thus the values in raw sewage can be directly compared with those in liquid from the primary settlement tank. Activity concentrations in raw sewage in August were up to around 6 times lower than the values in liquid from the primary settlement tank in January. This is probably due to differences in discharges since the flow rates differed by only about 15%. This is an expected result given that ^{131}I is discharged intermittently by hospitals (Section 8.2).

Under normal circumstances, the time taken for material to pass through the STW is about 12 h. A comparison between activity concentrations in the input and the outfall was therefore only possible for the samples taken in the afternoon and evening of 11 August and the morning of 12 August respectively. For ^{14}C, ^{35}S and ^{131}I, there was reasonable agreement between the activity concentrations in the input and output streams; the agreement was also reasonable for ^{99m}Tc ($T_{1/2}$ = 6h) when radioactive decay was taken into account. In contrast, activity concentrations of ^{32}P differed by an order of magnitude, and the difference could not be accounted for by the values observed in the sludge. However, the activity concentrations in the input in the morning of 11 August were around 20 times lower than the values observed later in the day, and so it is possible that the differences observed for ^{32}P reflect rapid changes in activity concentrations in the input. Overall therefore the results indicate that, for the radionuclides for which data were available, most of the activity discharged into the sewerage system can be accounted for by monitoring at the STW.

8.6 Estimation of Radionuclides Lost in Offgases

The amounts of radionuclides lost in offgases is of particular relevance to potentially volatile radionuclides such as ^{131}I. The measurement information was used to estimate the amount of some radionuclides lost in off gases from the amounts introduced in sewage cake feedstock and the activity recovered from residual ash. To enable the sludge cake and residual ash to be related to each other it was necessary to know the ash content of the sludge cake. The ash content was measured by ashing two samples of the dried incinerator sludge from Beckton in a muffle furnace. In both cases, the ash content was about 22% of the mass of the original dried sludge.

The amount of activity in off gases was then calculated in the following way. One gram of the August incinerator cake contained 417 mBq ^{131}I (Table 28). This mass of wet cake corresponds to 0.283 g of dry matter, which in turn should produce 0.283 * 0.22 = 0.063g of ash. This mass of ash would contain about 38 mBq of ^{131}I, based on the measured activity concentration in ash of 608 $mBq.g^{-1}$ (Table 28). These results suggest that, of the original 417 mBq in each g of the incinerator input cake, 38 mBq was recovered from the ash, and the remaining 380 mBq (90%) was lost in the off gases. Similar calculations were not possible for all of the radionuclides of interest, either because the activity concentrations in either the sludge or the ash were below detection limits, or because the short half-life of the radionuclide made such calculations unjustified, or because the high counting uncertainties associated with the data made such calculations unwarranted. Where estimates were possible, the corresponding data for both the August and January samplings are shown in Table 30. As might be expected, the results indicate that for ^{14}C all of the activity was volatilized. However, results for less volatile radionuclides such as ^{235}U and ^{51}Cr were less conclusive: in the case of ^{235}U, the results indicated that around 30% of the activity was volatilized, whereas for ^{51}Cr the amount of activity measured in the ash was around 50% greater than was estimated to arise from the values observed in the sludge. These discrepancies may well be due to sampling difficulties, since it is not possible to ensure that samples of ash and feedstock were comparable (Section 8.3). The results for iodine should therefore be taken as broad indications, although in the case of ^{14}C it would be surprising if any activity remained in the ash. This also indicates that the use of tracers within an incinerator is unlikely to produce reliable estimates of volatility. If more rigorous information on volatility is required, then small-scale controlled experiments would be more appropriate.

The stack scrubber may remove some of the volatilized radionuclides from the off gases and reduce the discharges to atmosphere. The stack scubbers produce waste liquors, and during the January sampling, samples were taken from both tanks that hold the scrubber liquors. The largest measured concentration of ^{131}I in these was 70 $Bq\ kg^{-1}$. The scrubber liquor is recycled, and so the total activity in the scrubber at a given time may reflect throughput over several days. Consequently, estimates of the proportion of ^{131}I trapped each day from the incinerator off-gas will be cautious if they are based on the activity concentrations observed in the liquor. The average daily throughput of the plant is about 800 tonnes of cake, with an observed activity concentration of about 330 $Bq\ kg^{-1}$ (Table 29). If the activity concentration of iodine is sustained, about 270 MBq of ^{131}I passes into the incineration plant each day. One of the two scrubber liquors contained 70 $Bq\ kg^{-1}$ of ^{131}I, the other contained only 1 $Bq\ kg^{-1}$. The plant is estimated to contain about 10 m^3 of useable liquor in total and so even at the higher concentration, the total ^{131}I content of the scrubber liquor would be about 0.7 MBq. This represents 0.25% of the daily plant input of iodine-131. Since most of the ^{131}I entering the plant is expected to be volatilized, the scrubbers are very unlikely to affect the amount of activity discharged to atmosphere.

9 DESCRIPTION OF THE LEEDS SEWAGE COLLECTION, TREATMENT AND DISPOSAL SYSTEM

The assessment model SMART described in Chapter 4 was set up to model the Leeds sewage collection and treatment system and assess the associated radiological impact.

9.1 Sewage Collection in Leeds

Leeds was chosen for study because it is considered representative of urban areas in the UK. It is served by one STW at Knostrop located in the east of the city. The majority of the sewage collection system is a combined system, which carries foul water from domestic premises, trade effluent and run off from roads and urban areas, although newer developments in the city have separate systems installed. The catchment area for Knostrop is divided into 'low' and 'high' levels, the former serving the more industrial side of Leeds to the south of the river Aire, the latter serving the more residential areas to the north. A schematic map of Leeds showing the location of the STW is given as Figure 16. The low level part of the catchment extends west to include Pudsey, Horsforth and Drighlington, the boundary follows the line of the M62 from Drighlington until it meets the M1 and then approximately follows the line of the M1 north as far as the river Aire. The high level catchment, north of the river Aire extends east to include the towns and districts of Colton, Manston and Shadwell. The northern extent of the catchment boundary includes Alwoodley and Cookridge to the North West. The total area of the catchment is estimated to be approximately 800 km^2. Inputs of raw effluent to the high level and low level catchments correspond to a population equivalent of 877,000. All of the authorised disposers to Knostrop are located in the high-level catchment area.

9.2 Sewage Treatment at Knostrop

This section has been based on information supplied by Yorkshire Water Services.

The high and low level works are treated separately at Knostrop. After treatment the high and low level final effluents are combined before being discharged to Wyke Beck. Since all of the disposers of interest are in the high level catchment the description will mainly be confined to the high level works. At Knostrop the average daily flow rate for the low-level sewage works in 1996/97 was 131 000 m^3 d^{-1}, with a dry weather flow of 106 000 m^3 d^{-1}. For the high level works the average daily flow rate of effluent through the STW was 97 600 m^3 d^{-1} with a dry weather effluent flow rate of 79 000 m^3 d^{-1}. The maximum effluent flow rate through the STW is 205 000 m^3 d^{-1}. The mean level of suspended solids in the high level crude sewage is 444 mg l^{-1}. The range is 67 to 1690 mg l^{-1}; one reason for the lower bound is the high volumes of rainfall being picked up by the combined sections of the sewage systems.

Knostrop is a large site with office and ancillary workers based on site. Yorkshire Water Services import and treat trade wastes prior to controlled discharge to the works. They also

operate a clinical waste incinerator on site. The number of workers involved in actually treating sewage comprises 5 teams of 3 shift operators: two shift men being employed in the sewage sludge incinerator and power plant, with one man monitoring and operating the sewage works. Five day men do tasks on the works and operate satellite works and pumping stations and there is a technician based mainly in the incinerator. Additional support comes from a manager who is responsible for Knostrop and other sites. There are area science and technical staff as well as contract maintenance staff.

The high level works treats sewage consisting mainly of domestic waste as well as waste from hospitals and light industry. Raw sewage is pumped to preliminary treatment, where screens and grit traps remove large floating items and heavy solids. Effluent flows up to 205 000 $m^3 d^{-1}$ are passed on to primary treatment. The residence time in primary sedimentation tanks is between 2 and 6 hours, depending on the incoming flow. During this time light, mainly organic solids settle to the bottom of the tank and are scraped into a central hopper. The residence time of solids in the hopper is about 1 day, depending on the level of sludge retained in the tanks. Flows above 205 000 $m^3 d^{-1}$ are passed to storm tanks for settlement for 2 to 6 hours and storm overflows from the high level works can be discharged to Wyke Beck if necessary.

Primary tank effluent is pumped to either the high-level secondary treatment tanks or the low level tanks. Seventy five to 100% of the flow is passed through biological filters at the high level works. The microorganisms are encouraged to form colonies in beds, about 2m deep, filled with clinker or irregularly shaped stones, this bed then forms the biological filter. The primary effluent is sprayed over the beds by moving distributors. Retention time on the biological filters is about 30 minutes.

Secondary effluent emerges from the biological filtration carrying humus sludge produced as a result of biological activity, this is then settled out in humus tanks. The retention time in the humus tanks is between 2 and 6 hours. Settled humus solids are either mixed with primary sludge or returned to the inlet where the sewers combine before preliminary treatment.

Up to 25% of the high level primary effluent is treated in the lower level works. Treatment is broadly the same as that described above.

The high and low level effluents emerging from secondary treatment are combined before being discharged to Wyke Beck about 600m from the river Aire and about 2kms east of the lowlevel storm outfall. The time taken for effluents to pass through the high level works is between 7 and 18 hours.

9.3 Disposal of Effluent

Knostrop high level works has consent to discharge fully treated effluent to the river Aire at a maximum rate of 205 000 $m^3 d^{-1}$, provided that the discharge contains no more than 70 mg l^{-1} suspended solids. Subject to the same conditions fully treated sewage from the activated sludge plant can be discharged to Wyke Beck.

The combined maximum flow of low and high level is 405 000 m^3 d^{-1}. The Urban Waste Water Consent applies at Knostrop at 25 mg l^{-1} BOD or 70% removal, 125 mg l^{-1} COD or 75% removal and a 15 mg l^{-1} ammonia consent also applies.

Settled storm effluent from the low-level works may be discharged to the river Aire during storm events when the storm tanks are full, the high level works may discharge to Wyke Beck. The suspended particles in storm water overflows must be less than 20mm in any dimension.

9.4 Disposal of Sludges

The first stage of sludge treatment is the storage of primary, storm tank and humus sludges. Initially the dry solids content of the sludge is approximately 5%. The solid content of the sludges is increased to about 22% by dewatering to a cake and squeezing on belt presses. In times of incinerator shutdown this material can be taken directly to the works landfill site. Normally the sludge cake is passed to a pre-drier directly in line with the incinerator to get the sludge to about 30% dry solids which allows auto-thermic operation of the incinerator. The incinerator has a stack height of 40.0 metres, it is designed to have a throughput of 3.3 tonnes dry solid (tds)/hr and it operates at a minimum temperature of 850°C. The temperature of the exit gases varies from 950°C in the combustion chamber to about 210°C at the electrostatic precipitator to 95°C up the stack. Ash is produced at a rate of about 1.0 tonne/h, the ash contains on average 30 - 35% of the original dry solid weight of the sludge and the collected ash is disposed in the on site licensed landfill at Knostrop.

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10 RADIONUCLIDE DISCHARGES TO SEWERS IN LEEDS

10.1 Introduction

This chapter describes the authorised radionuclide discharges into the catchment of Knostrop high level works. These were required to enable the radiological implications of radionuclide discharges to Knostrop STW in Leeds to be assessed, as well as providing an input to the development of the sampling strategy at the works.

10.2 Sources of Information on Disposal of Radioactivity to Sewers

The main sources of information on disposals of radionuclides to the Knostrop works were the Certificates of Authorisation issued by the Environment Agency.

Establishments in the Leeds area holding authorisations for disposal were identified using access listings provided by the public register office of the Environment Agency in Leeds. This listing was used in conjunction with the catchment map for Knostrop sewage treatment works to identify the disposers who were within the catchment of the works. Copies of the relevant Certificates of Authorisation were obtained, and the establishments which held authorisations to dispose aqueous wastes to sewers were then identified. There are six authorised disposers in the Knostrop catchment area, three hospitals, two universities and the NRPB Northern Centre.

10.3 Most Important Radionuclides and Premises

The disposers to Knostrop STW were ranked in order of importance using a simple indicator of dose for the total disposals from each premises. In Chapter 6, the disposers to Beckton STW were ranked by estimating ingestion doses, since it was assumed that this was the dominant pathway for exposure. However, external doses to sewer workers could be equally if not more important than ingestion doses. In order to account for this, estimates of both external and ingested doses were made for the disposals to Knostrop. It was assumed that each establishment used its authorisation to the fullest extent and that the radionuclides were disposed continuously into the total effluent flow of 205 000 m^3 d^{-1} at the sewage works. A decay time of 12 hours for each radionuclide, between being disposed and reaching the sewage works and dilution into the total effluent flow per month was also assumed. As described in Chapter 6 authorisations for "other beta/gamma" emitting radionuclides are considered 100% I-131, authorisations for any combination of "iodines" are assumed to be I-125 and joint authorisations for C-14 and H-3 and considered 100% C-14.

The indicative ingested dose was estimated using the method described in Section 6.4.3. Indicative external dose rates were calculated using the external dose factor for a point source. The doses for each radionuclide were summed for each establishment and used to establish the ranking of the most important disposers and radionuclides.

The six disposers identified in the catchment of the Knostrop STW are ranked as shown in Table 31. The radionuclides and quantities specified in the authorisations are also shown in this table.

The estimates of typical disposals in 1997 were also obtained for each of the disposers to Knostrop. From these, typical disposals of each radionuclide were estimated. Some disposers gave nuclide specific typical disposals but the majority gave the same groupings as the authorisations, in which case the same assumptions were made as to which nuclides would represent the groups. Table 32 shows the typical and authorised disposals of each radionuclide (Bq/day) that were used in the assessment. Tables D27 and D28 in Appendix D also show the monthly authorised and typical disposals used in the assessment.

10.4 Analysis of Disposers

The majority of the radionuclide disposals to Knostrop come from hospitals. Using ingested doses, all three hospitals are responsible for a similar percentage of the dose rate. When considering ingested or external dose rate Cookridge Hospital is ranked 1, followed by the General Infirmary and St James University Hospital. However, the ranking order is sensitive to the delay time assumed. Limited disposals from Seacroft University Hospital, currently part of the same NHS Trust as St James, are made under the Hospitals Exemption Order. The other significant disposer is the University of Leeds. Disposals to Knostrop consist mainly of six radionuclides, these are Tc-99m, I-123, I-125, I-131, H-3 and C-14. The disposal authorisations also include "other" radionuclides. The nuclides included under this heading vary depending on the establishment, but specifically exclude alpha emitters. Details of "other" radionuclides are also given in Table 31.

11 PREDICTED DOSES FOR KNOSTROP

11.1 Introduction

As in Chapter 7 the sewer model (SMART) has been used to provide an assessment of the doses from disposals of radionuclides to sewers in the Knostrop catchment. Doses based on authorised and typical discharges were assessed, both were assumed to be diluted in the average dry weather flow in the catchment. Table 32 shows the authorised and typical discharges used in the assessment. Doses were calculated to four groups of workers involved with sewage: workers dealing with a blocked pipe at the premises of one establishment, workers cleaning and repairing man accessible pipes, workers at the Knostrop STW and workers at the incineration plant. Doses to members of the public exposed to discharges of treated effluent to the River Aire and to off-gases from incineration of sewage sludge were also calculated. The effect of wet weather flow and storm events on all the exposure scenarios was considered.

For workers at Knostrop STW the assessment considers the best estimate values for partitioning of activity between sewage sludge and effluent during treatment. Sludge disposal is normally by incineration and ash disposal to landfill, however as described in Chapter 10, when the incinerator is not in use, disposal of sludge is to landfill site. An estimate of these doses is also included. Input data specific to the Knostrop STW case study are presented in Appendix D, Tables D24 to D30, best estimate transfer coefficients are also presented in Appendix D.

11.2 Exposure of Workers from Blocked Pipes

Table 33 presents the doses received by workers repairing a small pipe, serving one premises only. The premises chosen to illustrate the doses that may be incurred was the highest ranking disposer to Knostrop. The workers are considered to be unblocking a pipe that serves toilets used by patients who have received large doses of I-131 during thyroid treatment. As described in Chapter 7 the blockage is assumed to comprise 20% of one day's excretion of I-131 from one patient (total excretion over 2 days), plus radionuclides present from one day's continuous use in tracer or laboratory work. For an administration of 10 GBq of I-131, the amount of activity present in the blockage would be 1 GBq. For the other radionuclides disposed, a fraction of 1 day's discharge was assumed to be in the pipe. For Tc-99m, 40 GBq may be disposed per month. This is disposed of fairly continuously throughout the month, so during 1 day ~1.33 GBq is disposed. Assuming a 10 hour working day and 2 hour's build up in the pipe there would be ~0.27 GBq of Tc-99m in the pipe. On the same basis there would also be ~0.01 GBq of 'other' radionuclides in the pipe. For calculation of external dose, the activity is assumed to approximate to a point source. For internal exposure calculations, it has been assumed that the activity is diluted in 2 m^3 of waste liquid. Again, a delay time of 2 hours between the time of discharge and when the repair work is being carried out is assumed, and it has been assumed that workers spend 2 hours fixing blocked pipes. The doses are presented as a dose per event because

small pipes could become blocked at any number of times during a year and may not always be fixed by the same worker.

The dose is estimated to be about 90 μSv each time a pipe is unblocked. This dose is below the annual dose limit for members of the public of 1 mSv y^{-1}, and the maximum public dose constraint of 300 μSv y^{-1} It is also well below the annual average dose from natural background radiation[39] of 2.2 mSv y^{-1}

This exposure pathway assumes that the worker unblocking the pipe is unaware of the radioactivity present. Within hospitals however, small-bore sewer pipes from active drains are labelled as possibly carrying radioactive materials and carry detail of who to contact in case of blockage or leakage. In reality when a blockage occurs, the local RPA would be involved in assessing the quantity of activity present and planning a scheme of work for remedial actions, prior to the work being undertaken. Hospitals RPAs have suggested that for such remedial actions, an aim would be to keep the dose below around 100-200 μSv, similar in scale to the dose calculated by the SMART model.

11.3 Exposure of Workers in Man Accessible Pipes

Table 32 presents doses to workers in large pipes. The calculation assumes the workers spend 1600 h y^{-1} maintaining pipes and exposed to the discharges from all sites within the catchment diluted in the total dry weather flow to the catchment, 7.9 10^4 m^3d^{-1}. The dose based on typical discharges is 0.4 μSv y^{-1} and on the authorised limits is 2.7μSv y^{-1}. Both are below the annual dose limit for members of the public of 1 mSv y^{-1}, below the threshold for optimisation[38] of 20 μSv y^{-1}, and are at a level generally regarded as trivial.

11.4 Exposure of Workers at the Sewage Treatment Works

Individual doses to workers at the sewage treatment works are presented. Two scenarios for sludge disposal at Knostrop are considered:

- Incineration of sludge where best estimate removal coefficients of radionuclides during sewage treatment, and best estimate volatilizing fractions of radionuclides in the incinerator are used, resulting in radionuclide concentrations in both incinerator flue gases (releasing to atmosphere) and radionuclides in the ash disposed to landfill.
- Sludge taken directly to landfill, where best estimate removal coefficients during sewage treatment only are used. This represents the occasional disposal of sewage sludge directly to landfill due to incinerator shutdown. Incinerator shutdowns can last for a few weeks, due to the major engineering required, but they are infrequent. However, for modelling purposes it was conservatively assumed that a years' worth of sludge is taken to landfill.

Incineration of sludge was estimated to lead to doses to sewer worker of 25 μSv y^{-1} when typical disposals are assessed and maximum doses of 78 μSv y^{-1} when discharges at authorised limit are considered. The dominant pathway for the workers at the STW is external exposure, based on the assumption that workers spend long periods of time in close proximity to treatment tanks. The occupancy times for workers at all stages of the treatment process are given in Appendix D, Table D16.

As at Beckton, some workers may spend all of their time in the sludge pressroom in close proximity to sludge being de-watered prior to incineration. Doses to these workers are 79 μSv y^{-1} from typical disposals and up to 238μSv y^{-1} from disposals at the authorised limit. These doses come predominately from external exposure to I-131 which is responsible for 75 and 221 μSv y^{-1} for typical and authorised disposals respectively.

The best estimate case with all sludge going to a landfill site results in an estimated dose of between 5 and 16 μSv/y, depending on whether typical or authorised limit discharges are used. These are also dominated by the external dose. The estimated doses are below the dose limit for members of the public of 1 mSv y^{-1}.

11.5 Exposures of Members of the Public

Table 34 also presents individual doses to members of public with critical group habits resulting from disposals of effluents sludges and incinerator ash and 50 years of disposals has been assumed.

Using typical discharges the estimated doses to members of the public are 19 μSv y^{-1} from effluent disposed to the River Aire, 0.03 μSv y^{-1} from activity discharged via the incinerator stack and 6.3 10^{-4} μSv y^{-1} following the disposal of incinerator ash to landfill. The estimated doses to members of the public for the hypothetical disposal of all sludge to landfill are about 0.06 μSv y^{-1}. When discharges are at the authorised limit the doses to members of the public become 180 μSv y^{-1} from disposal of effluent, 2.2 μSv y^{-1} from incinerator releases to atmosphere and 1.5 μSv y^{-1} from the disposal of incinerator ash to landfill. The doses to the public from the disposal of all sludge to landfill would be 150 • Sv y^{-1}.

The doses from 1 year and 50 years of disposals of effluent and incinerator releases are similar either because the dominant radionuclides have short radioactive half lives and do not build up in the environment or because of dispersal in the river system. This is only true of typical disposals where the doses are dominated by I-131; doses from disposals at the maximum authorised limit are dominated by C-14, which is long lived. The estimated dose from the disposal of sludge and incinerator ash to landfill is due almost entirely to C-14. Carbon-14 and H-3 have relatively long half-lives and hence the 50y disposal gives higher doses than 1y disposal. All the doses are below the dose limit for members of the public of 1 mSv y^{-1}, and the maximum dose constraint of 300 μSv y^{-1}. Doses from the incinerator are all well below 20 μSv y^{-1} (the threshold for optimisation[38]) and 10 μSv y^{-1}, the level considered as trivial by IAEA[40].

The estimated doses from effluent discharges at the authorised limit require further attention. The majority of the dose arises from C-14, however the authorised limits are given for C-14/H-3 combined. The assessment conservatively assumed that this was all carbon-14. In reality, much of the disposal will be H-3. These considerations are explored further in the discussion chapter.

The collective doses to the UK population, truncated at 500 y, are presented in Table 35. The collective doses from typical discharges and 1 year's disposal are 0.7 manSv from effluent disposed to the River Aire, 0.04 manSv from atmospheric releases from the incinerator and 2.5 10^{-6} manSv following the incinerator ash disposal to landfill. The disposal of all sludge to landfill gives rise to a collective dose of 2.5 10^{-4} manSv. The collective dose from discharges at the authorised limit would be 3.3 manSv from effluent disposed to the River Aire, 0.4 manSv from atmospheric releases from the incinerator and 0.006 manSv following the disposal of incinerator ash to landfill. The dose resulting from the disposal of all sludge to landfill would be 0.62 manSv. Collective doses to the UK population truncated at 500 y from 50 years of disposal show the same pattern as for 1 year's release but are 50 times higher.

11.6 Wet Weather Flow

The total flow of sewage to Knostrop STW can rise to 2.05 10^5 m^3d^{-1} depending on the weather. SMART was used to estimate the effect this may have on all of the doses discussed so far in this chapter. The inhalation and ingestion doses to workers in large pipes and at the works are slightly lower. However, the total dose to workers is dominated by the external dose, which is strongly influenced by the activity concentration in separated and dewatered sludges. The quantities of sludges produced are not significantly affected by wet weather flow conditions, so the activity concentrations in sludges are unaffected. The doses to the public also remain unchanged.

11.7 Storm Events

In times of very severe weather conditions when the STW is in danger of flooding, the excess flow to the STW may be discharged directly to the receiving watercourse without undergoing treatment. The radiological impact of these events, are modelled assuming that all of the activity remains in the effluent and is discharged directly into the nearby water courses. Delay time is considered to be near zero.

The only doses that are considered are those to the public from the disposal of effluent. There will be no doses to workers since the STW is effectively bypassed and it is assumed that there will be no workers in large pipes during a storm. The dose received by an individual with critical group habits as a result of a storm event lasting a few hours at authorised levels of disposals has been estimated to be 0.08 μSv and the collective dose is 1.40 10^{-3} manSv. If ten

such events occurred per year, the annual dose would be less than 1 μSv. This dose can be considered negligible.

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12 MONITORING AT KNOSTROP SEWAGE TREATMENT WORKS

12.1 Introduction

The general approach to this part of the study was very similar to that adopted for the Beckton STW (Chapter 8). Monitoring of the radionuclide concentrations in the effluents arriving at the Knostrop sewage works and in the sludge and treated effluents was intended to support the radiological assessment, in terms of both the resultant doses and the validation of the model SMART (Chapter 4). The potential influence of changes in flow rate were assessed by mounting sampling programmes in September (late summer) and November 1998 (early winter) to reflect both summer and winter conditions. The disposals given in the authorisations were inspected. If disposals occurred at the levels specified in the authorisations on a continuous basis into the volume of effluent passing through Knostrop then six radionuclides (^{123}I, ^{125}I, ^{131}I, ^{99m}Tc, ^{3}H and ^{14}C) may be present at activity concentrations above the limits of detection. These radionuclides were therefore chosen for the case study.

12.2 Temporal Patterns of Disposal

As was noted in Chapter 8, information in temporal patterns of radionuclide disposal is essential in the planning of a sampling strategy. This is because the disposals of radionuclides may not be continuous and may cease at times, during holidays or over weekends. Large disposals of radionuclides used in therapy may occur over a few days in a month. Therefore it is important to know what the likely patterns of disposal were immediately before and during the sampling, so that peaks of discharge can be identified and times when no discharges are occurring are avoided. The situation at Knostrop STW is very different from that at Beckton (Chapter 8). Most of the activity received by the Knostrop STW originates from a single facility, which is a hospital carrying out thyroid ablation therapy once every 1 - 2 weeks, using ^{131}I. Treatment is usually carried out on Tuesdays. Other radionuclides are discharged from various other facilities in the Knostrop catchment, generally on a continuous basis, but the amounts are relatively small.

12.3 Sampling Strategy

Both sampling runs were carried out over a 24-hour period in the middle of a working week. The works were sampled on Tuesday/Wednesday, 15/16 September 1998 and Wednesday/ Thursday, 18/19 November 1998. Sampling was timed to cover approximately discharges to drains during a working day, but it was not known if ablation therapy was occurring around the time of either sampling trip. Following the experience at Beckton, no attempts were made to sample raw input sewage. The intention was to collect liquid and sludge from both the primary and secondary settlement tanks along with a continuous 24 sampling from the plant outfall. Unfortunately during both visits, no sludge was being pumped from the primary settlement tank

and so it was not possible to take a sample. As the sludge from the secondary settlement is pumped back to the input, the operators deliberately maintain a low suspended solid content. This meant that despite taking large samples at the works only a small amount of suspended solids was obtained. Consequently, it was only possible to perform a limited number of radiochemical analyses on this sample.

12.3.1 Sampling at the STW

a) September

A sample of the liquid outfall from the primary settlement tank was taken at 11:45 on the 15 September. Samples of both the liquid and sludge were taken from the secondary settlement tank at 16:15 on the same day. The works outfall was sampled continuously from 16:00 on the 15 September to 08:30 on the 16. Unfortunately during the morning of the 16 the auto sampler failed and so a single sample was taken at 12:15. The auto sampler was then re-set to sample from 12:15 to 16:00 on the 16. After measurement of the gamma ray emitting isotopes, the samples were bulked to a single composite sample for radiochemical analysis.

b) November

A sample of the liquid outfall from the primary settlement tank was taken at 11:30 on 18 November. Samples of both the liquid and sludge were taken from the secondary settlement at 16:00 on the same day. The works outfall was sampled continuously from 12:00 on 19 November to 10:00 on the following day. The sampler was changed at 16:00 to give one sample from 12:00 to 16:00 on 19 November and one from 12:00 on 19 to 10:00 on 20. After measurement of the gamma ray emitting isotopes the samples were bulked to a single composite sample for radiochemical analysis.

12.3.2 Sampling at the incinerator

The inputs to the incinerator are sewage cake feedstock, and the outputs are off gases, residual ash and offgas scrubber liquors. The feedstock for the incinerator is prepared by dewatering sewage sludges using continuous belt presses. The pressroom contains 5 presses each is approximately 2 m high and 3 m long. A continuous belt about 1.5 m wide repeatedly passes through the press at different levels. A layer of sludge about 100 mm thick is spread on to the top belt and water is pulled through the porous belt as the sludge is transferred through the press. The water content of the sludge is reduced from around 95% to about 75% during this process. The resultant "cake" may then be air dried to around 30% solid before incineration at about 800 °C in a fluid bed incinerator.

a) September

A sample of both the input liquid sludge before thickening and the ash was taken on 16 September. No cake was taken during this sampling, because of lack of access.

b) November

Samples were taken of sludge during thickening from the belt thickener and ash on 19 November. The possibility of sampling the waste gas scrubber liquids was investigated. However, there was no suitable sampling point, and so the influence of the stack scrubbers on discharges of radionuclides to atmosphere could not be studied.

12.4 Sample Preparation and Analysis

The preparation of samples for gamma-ray spectrometry followed the approach described in Chapter 8: all liquid samples were filtered on site or at the laboratories of NRPB's Northern Centre at Leeds. Residues and filter papers were discarded. Most sludge samples were subsequently concentrated to a stable paste by centrifugation. The sample from the belt thickener was sufficiently solid to be counted as collected. Gamma-ray emitting radionuclides were then determined using hyperpure Ge detectors at the Leeds laboratory. This laboratory does not hold formal UKAS accreditation but operates under comparable schemes of QA. Some of the radionuclides of interest are short-lived, and so it was necessary to use this facility to minimise losses in overall analytical sensitivity due to radioactive decay. Crosschecks were however performed with the Chilton facility to confirm the validity of the measurements. Separate subsamples were sent to the Board's Glasgow laboratory for the radiochemical determination of ^{3}H, ^{14}C, ^{35}S, ^{32}P and ^{125}I. The analyses at Glasgow were inside the scope of that laboratory's formal UKAS accreditation.

12.5 Results and Discussion

The results for September and November are shown in Tables 36 and 37 respectively. The average daily flow through the works on 15 and 16 September was 75,000 m^3, the corresponding value on 18 and 19 November being 99,000m^3. These flows are about an order of magnitude lower than those through Beckton (Chapter 8). As in the case of the Beckton STW, data are presented for radionuclides which are authorised for disposal to sewers and also for those radionuclides that were measurable using high resolution gamma ray spectrometry. In practice, radionuclides other than ^{131}I were generally only detectable in incinerator ash.

As there was insufficient sample to obtain a dry matter content, the results for sludge from the secondary tank are given in mBq kg^{-1} as sampled; the incinerator sludge results have been expressed in terms of the original wet state using the dry matter content given in the tables.

For the reasons given in Chapter 8, apparently similar samples may have different limit of detection values reported for the same isotope.

As discussed in Chapter 8 materials with radionuclide activity concentrations below 0.4 Bq g^{-1} are exempt from control under the Radioactive Substances Act 1993 (RSA93)[17] by the radioactive substances (substances of low activity) exemption order[41] therefore 0.4 Bq g^{-1} is often used as the definition of a radioactive material. The only samples containing radionuclides at concentrations higher than 400 mBq g^{-1} (0.4 Bq g^{-1}) are ^{7}Be (560) and ^{40}K (460) in the incinerator ash from the November sampling. The ash from the September collection contained 353 mBq.g^{-1} of ^{7}Be which when calculated uncertainties are considered would not differ significantly from the 400 mBq.g^{-1}. Both ^{40}K and ^{7}Be are naturally occurring. The difference in daily flows through the STW between September and November was about 30%, which is not sufficient to account for the differences of a factor of 5 in the concentrations of ^{131}I in liquid from the primary settlement tanks. As in the case of Beckton, this was an expected result because of the intermittent discharges of ^{131}I into the sewerage system. The September sampling results for ^{131}I suggest that thyroid ablation may have occurred around the time of sampling, whilst the November results suggest that no ablation had occurred around the time of sampling.

As in the case of Beckton, the time taken for material to pass through the plant is about 12 h. The radionuclides of interest are similar to those studied at Beckton, and so it would be reasonable to assume that most of the activity remains in the liquid phase. The samples of liquid taken from the primary settlement tank on the morning of 15 September should therefore be comparable with samples from the outfall taken over the following night. Although data are only available for ^{131}I, the two results were in reasonable agreement. This observation supports the conclusion reached for the Beckton STW, ie that most of the activity passes through the plant.

Estimates have been made of the amount of each radionuclide volatilised during the incineration process, using the approach described in Chapter 8. In the case of Knostrop the ash content of the dried sludge was 43% in September and 35% in November. The results for those radionuclides for which such calculations are warranted are shown in Table 38. As in the case of the Beckton incinerator, the results indicate that most of the ^{131}I and ^{14}C was volatilised, whereas more non-volatile elements tended to remain more in the ash. However, the issues concerning the comparability of the ash and cake samples noted in Chapter 8 also apply here.

13 INCINERATION OF SLUDGE

This chapter examines the practice of sewage sludge incineration with respect to the authorised disposals of radionuclides to sewers. Specifically included is a review of current and proposed use of sludge incinerators, a review of the behaviour of radionuclides within incinerators, the radiological consequences of the operation of incinerators, the sensitivity of assumptions made in the radiological assessments and a consideration of the feasibility and usefulness of a tracer study to improve the knowledge of incinerator behaviour.

13.1 Current and Proposed Use of Sewage Sludge Incinerators

Sewage sludge produced during sewage treatment has been disposed of in a number of ways, including application to agricultural land, disposal to landfill, disposal to sea and incineration. The routes by which sewage sludge are disposed depend strongly on the logistics of the operations and the content of the sludge (ie potentially toxic element concentrations). In rural situations suitable quantities of agricultural land may be available nearby, whilst in urban situations, agricultural land maybe a number of miles away, and may require a large continuous tankering operation. Historically, the larger more urbanised sewage works on the eastern side of the UK have used sea disposal. Sewage works are frequently located on the edge of rivers or estuaries, which allowed barges easy access for pick up of the sludges. Now this option is no longer available, alternatives have been put in place.

A literature review of the current and proposed use of sewage sludge incinerators and visits to Beckton and Knostrop sewage treatment works have provided the following information:

- In 1995 the disposal routes for sewage sludge in the UK were 44% to agricultural land, 30% sea disposal, 8% to landfill site and 7% to incinerators.
- The removal of the sea disposal route (end of 1998) has lead a number of water companies to consider alternative sewage sludge management options. Of the 14 regional water companies in the UK, the majority appear to favour incineration. Incineration is particularly important for STW within urban areas where the costs of transporting sludge to agricultural land (or landfill) are high. Incineration is also attractive because it is a quick and cost-effective way of reducing volume and stabilising the sludge.
- It can be expected that incineration will replace most of the sea disposal percentage, giving a total percentage of 37%.

13.1.1 Current incineration methods

- There are generally two types of incinerator, multiple hearth and fluidised bed. The multiple hearth design has a series of hearths stacked on top of each other. The sludge is added at the top and slowly falls down until it reaches combustion. The ash is collected at the bottom. Multiple hearth incinerators are an old design and in the UK they have largely been replaced by fluidised bed incinerators. In these incinerators the sludge enters a single chamber and falls onto a bed of sand, fluidised by an air blower. This design has a greater demand for power, but has better combustion efficiencies and odour control.
- If the dry solids content of the incinerator sludge is around 30%, an incinerator does not require additional fuel to burn. However, untreated sludge has a solids content of 1.5 - 6% (by weight), and the solid fraction may be increased by dewatering. Dewatering can either be achieved by air drying, centrifuges, filter belt presses, or the use of waste heat from the incinerator.
- Following incineration the ash residue can contain 20 - 40% of the original dry solid weight of the sludge, depending on the temperature and efficiency of the incinerator. Incinerator ash is normally removed to landfill.
- Analysis of sludge ash shows it mostly containing silicon and aluminium oxides, with heavy metals comprising less than 0.1% by mass. Combustion is fairly complete as only around 2% of organic and volatile matter remain.
- In the Greater London sewage system, the large urban sewage works at Beckton and Crossness have recently commissioned sewage sludge incinerators. At Beckton the sludge is thickened to around 32% of dry solids and incinerated. Normally the time between the start of sewage treatment process and incineration is 2-4 days. The incinerator is expected to handle 70,000 tonnes dry solid (tds) per year. Around 20% of the sludge solids remain as ash which is sent to landfill at Beddington, this has been confirmed by recent measurements of the ash content in this study. The Riverside STW sends its sludge to Beckton for incineration. The incinerator at Crossness is expected to handle 50,000 tds y^{-1}.
- The sewage sludge from the Knostrop STW in Leeds is also incinerated. Following dewatering and squeezing on belt presses (to 22% dry solids) the sludge cake is passed to a pre-dryer directly in line with the incinerator to increase the dry solids content to 30%. The incinerator is designed to handle 3.3 tds h^{-1} and Yorkshire Water suggest around 30-35% of sludge remains as ash (approximately 1 tonne h^{-1} of ash produced), however recent measurements indicated that 44% of solid content remains as ash. Collected ash is disposed to the on-site landfill at Knostrop.
- The Beckton and Knostrop incinerators both have flue gas cleaning systems to reduce dust, acid gases, metals and other pollutants.

(i) At Beckton the flue gases, following cooling in a steam boiler from 850°C to 200°C, are cleaned in three stages[42].

In the first stage gases are cooled to approximately 150°C in a reactor tower, where recycled water from the scrubbing system (third stage) which has been treated with lime is sprayed into the gas stream. The hot gases evaporate the water and the lime reacts with the acid gases. Some ash and lime particulate collects as dry powder at the base of the tower.

In the second stage the particulate matter in the flue gases is removed by bag filter. The ash is transported pneumatically to storage silos outside the building.

In the third stage, the gases are passed through a two-stage wet scrubbing system to remove volatile metals and any remaining acid gases. Gases are cooled to saturation temperature and washed under acidic conditions to remove chlorides, fluorides and mercury. A second scrubbing system provides alkaline conditions (lime) to promote removal of the remaining gases, particularly SO_2.

About one third of the daily ash made is collected from the boiler and reactor tower, the remainder is collected from the bag filters. Sludge from the wet scrubber is settled, dewatered and mixed with ash for off-site disposal.

Gases leaving the scrubbing system (at 75-80°C) pass through a demister system. The gases are cooled by indirect contact with high rate water sprays to reduce moisture content, the effluent being returned to the STW. The gases are finally reheated to minimise the risk of visible plume formation when released to atmosphere.

(ii) At Knostrop the system gases from the combustion chamber (950°C) are cooled and passed through an electrostatic precipitator (210°C) and released to stack where temperatures are around 95°C.

13.2 Behaviour of Radionuclides in Incinerators

Information sources on the behaviour of radionuclides in incinerators are limited, the details obtained from References 25, 14 and 43 are discussed in this section:

- Upon incineration of sewage sludge, radionuclides will either be released to air (where they disperse), are retained in the ash or by-products of flue gas cleaning systems, or partition between the two. The incineration process may lead therefore to exposure to the public through atmospheric dispersion and deposition and landfilling of radionuclides in municipal tips.
- Radionuclides preferentially released to air are likely to be H-3, C-14 and iodines (the fraction remaining in the ash is likely to be a few %).

- The other radionuclides such as chromium or phosphorus are likely to be, mainly, retained in ash.

As discussed in Chapters 8 and 13, samples of incinerator feedstock and incinerator ash were collected at both Beckton and Knostrop STW. The aim was (as far as possible) to coordinate the timing of sampling, so that incinerator feedstock and incinerator ash samples were from the same 'batch' of sewage sludge. The waste liquor from the Beckton stack scrubber was also sampled. Sampling of by-products from the flue gas cleaning system at Knostrop was not possible. From measurements of radionuclide concentrations in these samples, the transfer of nuclides to ash can be estimated. The release of radionuclides to air can then be inferred. The byproducts from the stack scrubber at Beckton suggested less than 0.25% of iodine is removed by the flue gas cleaning system, a similar quantity is likely for the Knostrop incinerator.

13.2.1 Incinerator removal coefficients

As discussed in Chapters 8 and 13 (and given by Tables 30 and 38), estimates of the incinerator removal coefficients or volatilised fractions have be made for radionuclides that were measured in both incinerator feedstock and ash above limits of detection (LOD). Table 39 compares these estimates with volatilised fractions taken from available literature and those used in the model SMART.

13.3 Doses from Sewage Incinerators

As described in Chapters 7 and 12, the model SMART has been used to calculate the doses to sewer workers (including work in the incineration plant) and members of the public, as a result of authorised disposals of radionuclides to sewers in the Beckton and Knostrop STW catchments. The calculated doses for Beckton have been summarised in Table 26 and for Knostrop in Table 34. The estimated doses illustrate that:

- Workers at the incinerator are likely to be the most exposed persons at a STW. Using typical discharges doses of around 18 μSv/y have been estimated at the Beckton STW and around 23 μSv/y at Knostrop STW.
- The largest dose to members of the public is as a result of effluent discharges. Public doses directly related to the incineration process (incinerator releases to atmosphere and incinerator ash disposal to landfill) are at a level generally regarded as trivial.
- The worker doses are dominated by the external exposure from iodines and Rb-84. The public doses associated with incinerators are dominated by C-14 and iodines.

13.3.1 Sensitivity of estimated doses

The estimated doses have shown that a limited number of nuclides are important when considering the radiological consequences of sludge incineration. Iodines and C-14 are dominant, Cr-51, Tc-99m, P-32 and H-3 only make small contributions. To test the robustness of data used for the calculation of doses associated with incinerators a brief study has been made of the sensitivity of the doses estimated to changes in volatilised fraction. The sensitivity study considers the possible variation of volatilised fraction for each of the dominant nuclides separately while the parameters associated with the other radionuclides remained as best estimates:

Iodines

Table 39 indicates the volatilising fraction for iodine appears to be well known, ie there is good agreement between measurements and literature. The sensitivity analysis has therefore only considered limited variations of +/- 5% around the best estimate volatilised fraction of 0.95. At 0.9 the worker dose at the Knostrop incinerator from iodines increases from 64 to 96 μSv/y, increasing the total dose at the incinerator from 73 to 101 μSv/y. At 1.0 the public dose from atmospheric release at Knostrop increases from 2.2 to 2.3 μSv/y

H-3, C-14

The volatilised fraction for these elements is well known, although the best estimate volatilised fraction is almost 100% release to air.

At 80% released to air, the dose to a member of the public from disposals of ash to landfill at Beckton increases from 2 to 44 μSv/y.

Others

The only other nuclides of interest for doses from incineration (and the best estimates of volatilised fraction) are Cr-51 (0.01), Tc-99m (0.95) and P-32 (0.01). Reducing all these fractions to zero, maximises the doses to sewer workers from these radionuclides increasing the total dose from 73 to 97 μSv/y at the Knostrop incinerator (no increase occurs at Beckton). If the volatilising fraction for Cr, Tc and P are all increased to 1.0 then the dose to members of the public from atmospheric releases do not change significantly.

This shows that the calculated doses are not sensitive to those nuclides for which limited information exists on volatilised fractions.

13.4 Tracer Study of Incinerators

The aim of a tracer study for an incinerator would be to establish more accurately the volatilised fractions of various nuclides present in sewage sludge. Attention should focus on those radionuclides which contribute significantly to dose. The study would need to consider each of the process streams associated with an incinerator, the sludge feedstock, the flue gases and the residual ash. Such a study could either attempt to measure existing radionuclide concentrations in the various process streams, or 'spike' the feedstock if existing levels are below possible limits of detection.

Samples of incinerator feedstock and ash taken at Beckton and Knostrop STW have measureable concentrations of I-131, In-111, Ga-67, Cr-51 and C-14. However the concentrations primarily in the feedstock are subject to high uncertainties for a lot of these nuclides as concentrations are close to limits of detection. The measurement of radionuclides in the flue gases is likely to be difficult.

The typical sampling efficiency for particulate in flue gas sampling is 0.01 - 0.06% of the total flow in the flue with a one-hour sampling period. This obviously would be lower for vapour phase species such as iodine which needs to be adsorbed from the flue gas rather than trapped as a particulate. A simple charcoal bed (the usual method used for absorbing iodine) would not work effectively at the temperatures found in the flue gas. Consequently, adequate sampling of iodine species in the flue gas would not be straightforward.

As further problem is likely to be the limits of detection. An approximate calculation for I-131 passing through the stack has been considered for Beckton:

Incinerator throughput		$7\ 10^4$ tonnes dry solid / year
	ie	8 tonnes per hour
I-131 in cake, measured at		417 Bq/kg @ 28% dry solid
	ie	1489 Bq/kg of dry solids
So at 8 tonnes per hour		$1.19\ 10^7$ Bq I-131 in incinerator load
Assuming 90% combustion		$1\ 10^7$ Bq I-131 in flue gas
Assuming 0.01% collected		1000 Bq / h

At this level the iodine concentrations would be measurable by gamma ray spectrometry, however 0.01% may be an optimistic estimate of the collection efficiency.

The above calculation has illustrated that it may be possible to measure the I-131 content of the flue gas without adding any tracer, but would useful additional information be obtained from what was gleaned from recent measurements? If an experiment was performed using the I-131 already present in sewage sludge, the estimate of the volatilised fraction would still be dependent on the uncertainty of the measurements of radionuclide concentrations in the cake. In a tracer experiment there are still likely to be considerable uncertainties in the flue gas measurement, and indeed to achieve a 1 hour sample period, a suitable tracer would need to be mixed with at least 8 tonnes of incinerator feedstock. In both types of experiment the estimate of the collection efficiency for vapour phase species would be critical.

13.5 Incinerators: Summary and Conclusions

- The use of sewage sludge incinerators is on the increase, particularly for urban areas which used to dispose to sea but now have to use alternative disposal routes.
- Measurements of radionuclide concentration in incinerator feedstock and incinerator ash have been made at the Beckton and Knostrop STWs, and volatilised fractions estimated. The volatilised fraction has been considered as the partitioning fraction between incinerator ash and incinerator gases discharged.
- Limited information exists on radionuclide transfer within incinerators for comparison with estimates based on measured quantities, although the information found appears to agree well with measurements made by NRPB at Beckton and Knostrop STWs.
- Estimates of the dose received by sewer workers, using best estimates of radionuclide transfer within the sewer works have values of 23 and 18 μSv/y, well below the public dose constraint of 300 μSv/y. Around 95% of this dose is as a result of work within the incinerator due to the increases in concentration of radionuclides during sludge drying and ash production.
- Estimated doses received by members of the public associated with the use of sewage incinerators (incinerator atmospheric discharges and incinerator ash disposed to landfill) are small in comparison with doses calculated for the disposal of liquid effluent, and are generally to be regarded as trivial.
- The dominant nuclides to the dose pathways associated with incinerators are I-131 (contributing to both external exposure for STW workers and atmospheric discharges to the public) and C-14 (incinerator ash disposal to landfill). An examination of the uncertainty in the assumed value of volatilised fraction for these nuclides shows that the assessments are robust (ie estimates of dose would change only slightly if likely variations are considered).
- The main aim of a tracer study would be to establish more accurately the volatilised fraction, particularly for dominant radionuclides.

- A tracer study would probably be feasible, although measurements in hot flue gases would provide a significant technical challenge. However an examination of all process streams would be expensive, and is unlikely to provide useful additional information compared with the current measurements of volatilised fraction of dominant nuclides.
- Overall, considering the dominant nuclides, worker and public doses, recent measurements and the likely accuracy of further practical analyses, a tracer study of an incinerator appears not to be warranted.
- A more sensible and cost effective action would be to examine the external exposure of sewer workers. The use of strategically positioned TLDs could confirm (or otherwise) the levels of dose. Due to the generally constant radionuclide throughput a TLD remaining in position for only a month or two, should give a realistic estimate of doses. TLDs have been placed at both Beckton and Knostrop STW, the results are described in Appendix G.

14 REVIEW OF RESULTS AND DISPOSAL PRACTICES AT BECKTON AND KNOSTROP

14.1 Comparison of Beckton and Knostrop Sewage Treatment Works

The main features of the two sewage treatment works (STW) are summarised in Table 40. The main difference is the size of the works and the quantity of radionuclides received. Both sewage works serve large urban areas. The catchment of Beckton STW covers parts of Central London, whilst Knostrop STW serves all of Leeds, and is divided into high and low level works. Beckton sewage works receives up to 17 times more effluent than the Knostrop high level and produces 7 times more sludge by volume. Primary treatment at each sewage works is the same, whilst secondary treatment differs. Secondary treatment at Knostrop uses a mixture of biological filtration and activated sludge, whilst Beckton uses activated sludge only. Neither Beckton nor Knostrop employ tertiary treatment at present. Beckton discharges treated effluents into the Tidal Thames, whilst Knostrop disposes to the freshwater section of the River Aire. Since the end of 1998, both sewage works have disposed of sludge by incineration.

Beckton receives radioactive disposals from 65 disposers, the total reported activity disposed in 1997 was around 3.5 10^{11} Bq. Knostrop receives radioactive disposals from six organisations, (all to the high level works) the total disposal was around 2 10^{11} Bq in 1997. The main disposers of radioactivity to both STWs were hospitals. The majority of the activity disposed was Tc-99m, (more than 50% in both cases) followed by I-131.

14.2 Estimated Doses from Authorised Disposals to Sewers in Leeds and London

The model SMART was used to estimate effective doses to workers and the general public from authorised and typical disposals to sewers in London and Leeds. The results are discussed in Chapters 7 and 11 and given in Tables 24 to 27 and Tables 32 to 35 respectively. A summary is given in Table 40. The dominant radionuclides for both sites are C-14, Tc-99m, I-125 and I-131. Rb-84 has also been important for Beckton in London.

It should be remembered that for the disposals based on the authorised limits, assumptions had to be made about the exact mix of radionuclides disposed, and the disposals of I-131 in particular were thought to be overestimated because it was used to represent other $\beta\gamma$ emitting radionuclides. Typical disposals are based on returns made for 1997, which is the most recent year for which information is available. These discharges were assumed to be typical of any year, with the exception of discharges of Rb-84 to Beckton which are thought to have ceased at the start of 1998.

It is unlikely that Rb-84 discharges occurred after the incinerator at Beckton was commissioned. Therefore, two sets of estimated doses for Beckton are given, one set including Rb-84 discharges and one set without.

All of the predicted doses at both sites were below the public dose limit and dose constraint of 1 mSv/y and 0.3 mSv/y respectively. Since the doses associated with the sewage treatment works result from multiple disposals, they should only be compared with the dose limit of 1 mSv y^{-1}. The maximum dose constraint and thresholds for optimisation only apply to single sources i.e. individual disposers. In most cases the doses resulting from all disposals have been below the threshold for optimisation so there is no need to consider individual disposers. However where the doses are above the threshold for optimisation individual disposers must be considered since it is possible that one disposer is responsible for all of the dose. The situations where the dose is above the threshold for optimisation are discussed further in Section 14.5.

14.2.1 Doses based on authorised disposal limits

In the Knostrop catchment, the predicted effective doses in excess of 20μSv y^{-1} were to workers unblocking smaller pipes around a hospital, to workers at the sewage works incinerator and in the sludge pressroom, and to members of the public as a result of the discharge of treated effluent into the River Aire. Exposure of workers was dominated by external exposure to I-131, and exposure of members of the public by C-14 ingested in fish. In the Beckton catchment, the predicted doses in excess of 20μSv y^{-1} were to workers unblocking pipes, to typical workers at the sewage works, workers at the incinerator and in the sludge pressroom and to members of the public as a result of the discharge of treated effluent to the Tidal Thames. Exposure of workers dealing with blocked pipes was dominated by I-131 and exposure of workers at the STW was dominated by Rb-84 and I-131. External exposure dominated these exposure pathways. Doses to members of the public arose mainly from C-14 in marine foods.

All other predicted doses were below 20 μSv y^{-1}.

14.2.2 Doses based on typical discharges (1997)

In the Knostrop and Beckton catchments, all the predicted doses based on typical discharges were lower than those based on the authorised limits. At Knostrop, the doses predicted to be in excess of 20μSv y^{-1} were to workers unblocking small pipes around a hospital and to workers at the sewage works incinerator and sludge pressroom. At Beckton, all the predicted doses were below 20 μSv y^{-1} except for the workers spending all day in the sludge pressroom, and workers unblocking small pipes.

14.3 Analysis of Predicted Doses

The results show that there are three dominant exposure situations and four dominant radionuclides. The dominant exposure situations are workers unblocking a pipe, workers at the sewage treatment works in the sludge pressroom and incinerator plant, and public exposure to discharged liquid effluent.

Doses to workers unblocking pipes on a disposers premises are predicted to be around 100 μSv per two hour event assuming one day's worth of discharges plus 1GBq of I-131 from thyroid treatment. The estimated dose is dominated by the external gamma emitter I-131 and is expected to be an overestimate because this activity and the expected doses should be managed and under the control of the site RPA. Sewer worker gangs working near to the site could be exposed to radionuclides discharged from the site. However, these radionuclides would be diluted into greater volumes of effluent (at least a factor of 10) and so doses per unblocking event should be below 10 μSv.

Doses at the sewage works and around the incinerator are important for two main reasons. Firstly at the sewage works the discharges of all the disposers are combined. Secondly, processing of the effluents tends to concentrate radionuclides associated with solids. The solids are separated off and their volume is reduced by several orders of magnitude by dewatering. During incineration, the solid volume is reduced by a further order of magnitude. Thus, the activity concentration of radionuclides that transfer to solids are gradually increased. If radionuclides are retained during incineration, further concentration in ash will occur. Radionuclide concentrations in incoming effluent and treated effluent remain low and these media are relatively unimportant contributors to the doses at the works.

For the radionuclides considered, the main exposure pathway at the works is external irradiation. In the Knostrop catchment, I-131 is important, whilst at Beckton both I-131 and Rb-84 are important. For these radionuclides doses from external exposure are several orders of magnitude greater than those from internal exposure from inadvertent ingestion and inhalation pathways.

At Beckton the workers are exposed to iodine-131 in the dewatered sludge and exposed to rubidium-84 in sludge ash, following incineration. Most iodine-131 is lost to atmosphere on incineration so there is little exposure to it in sludge ash, whereas rubidium-84 remains in the ash and becomes more concentrated. At maximum authorised levels of disposal the doses from iodine-131 and rubidium-84 roughly balance, hence the doses to sludge press workers and typical workers are similar. Typical disposals of rubidium-84 at Beckton are far lower, and it can be seen that the doses to typical workers are less than half of the doses to sludge press workers. At Knostrop, no rubidium-84 is disposed of at all and the doses to workers are dominated by iodine-131, the doses at Knostrop are therefore dependent on the worker's exposure to dewatered sludge.

Doses to the public from C-14 are important because of the relatively high disposal rate of the radionuclide. About 70% of the C-14 is expected to remain in the effluent, which then enters the fresh water stretches of the River Aire (Knostrop) or the tidal Thames (Beckton). Carbon-14 is predicted to be readily transferred into freshwater fish and marine foods, which may give rise to significant doses from consumption of aquatic foods. However, the predicted doses may be an overestimate because of the conservatism in the modelling especially the assumed transfer of C-14 into aquatic foods and the quantities of fish and marine foods assumed to be caught in the vicinity of the outfalls and eaten.

Doses to the public from incineration off-gases are low (below 20μSv y^{-1}) because many radionuclides only partially transfer to the solid at the sewage works, and then only a fraction of some radionuclides in the sludges are actually discharged with the exhaust gases, the remainder are associated with the ash. The high stack heights employed at Knostrop and Beckton disperse the waste gases effectively. Once released, most of the radionuclides do not build up in the environment because they have short radioactive half-lives or are very mobile.

14.3.1 Predicted and measured doses

Doses were estimated at the Knostrop and Beckton sewage works by placing Thermoluminescent Detectors (TLDs) at various points in the parts of the works where radionuclides were at their most concentrated. The results are presented in more detail in Appendix G. The areas selected were where sewage sludge had been separated off and partially dewatered to form sludge cake (ready for incineration) and where the ash was collected after incineration. Controls were also placed away from sewage sludge cake or ash. The TLDs were left in place for between 2 and 6 months from the end of 1998 to early 1999. The measured doses were compared with the controls. The TLDs have an inherent variability of +-25% for a dose of 250 μSv. At the levels of total dose recorded by TLDs at both sites, the additional dose needs to be in excess of approximately 100 μSv to be quantifiable.

No doses were measured, at either works that were above the estimate of uncertainty on the results. Therefore it is possible to conclude that excess dose was always less than 100 μSv. This was in agreement with model predictions for both works based on typical discharges and authorised discharges and shows that the model is not underestimating doses.

14.3.2 Comparison of measured and predicted activity concentrations

The measured activity concentrations in the various sewage phases were compared with those predicted by the model. A full comparison of the measured and predicted values is given in Appendix F. In general, the model tends to predict higher activity concentrations for most of the radionuclides in most of the phases than were actually measured.

In particular, the model overestimated the C-14 concentrations in the effluent. However, the predicted and measured concentrations of I-131 in sludge, sludge cake and ash were similar indicating the model is not overestimating those.

The measurements confirm that the process of separating off and dewatering sewage sludges during treatment concentrates some radionuclides. The results also confirm that other radionuclides that are below detection levels in other media become detectable in residual ash. Measurements also confirm that some radionuclides remain in the ash after incineration of sewage sludge and are further concentrated as a result.

Rubidium-84 was not detected in any of the samples taken at Beckton. This confirmed that discharges of Rb-84 had ceased.

An additional study at Worcester Park sewage works showed that a discrete set of I-131 discharges can be traced through a sewage system. However, the monitoring carried out did not account for all the I-131 introduced into the system. Either sampling did not commence soon enough after the administration of the I-131 to the patient or significant amounts (50%) of the administered I-131 was retained in the patient and would be discharged after 48h. The latter may have implications for patients who are sent either home or to smaller local hospitals in rural areas 48 h after thyroid ablation treatment.

14.3.3 Comparison of radiological impacts at Beckton and Knostrop

The features of the two sewage works are summarised in Table 40. Beckton receives a factor of 6 or so more total activity from the radioactive discharges than Knostrop high level. However, Beckton works processes an order of magnitude more effluent and 7-8 times more sludge and incinerator ash into which the extra activity may be diluted. Therefore the radiological implications of the discharges to Beckton can be expected to be broadly similar to those from discharges to Knostrop. A comparison (using typical discharges) shows that SMART predicts that doses to typical workers at Knostrop and Beckton will be similar (annual doses of 25 and 19 μSv respectively).

Doses to members of the public from authorised disposals at about 1 km from the incinerator at Beckton are twice those at Knostrop. The predicted doses are expected to be related to the amounts of comparable radionuclides disposed into each sewage works catchment, given that the stack heights are approximately the same. At both sites I-125 and I-131 dominate doses. The amount of I-125 and I-131 disposed to Beckton are approximately four times those disposed to Knostrop.

Doses to members of the public from effluent discharges from Knostrop are higher by around a factor of 30 than those from Beckton. The dominant radionuclides in the doses from effluent are I-131, C-14 and P-32. Discharges of I-131 and P-32 at Beckton are about five times greater than at Knostrop, and discharges of C-14 are 20 times greater at Beckton. Only one food (freshwater fish) is assumed to be consumed from the River Aire, whilst three foods (fish, crustacea and mollusca) are assumed to be consumed from the Thames. It was assumed that the exposed group at Knostrop would eat 20 kg/y of freshwater fish, whilst at Beckton 20 kg/y of fish and 5 kg/y of both crustacea and mollusca were thought to be consumed. The different amounts of aquatic foods consumed around the two works may influence the predicted doses, however it is clear that the main influence is the dilution of the radionuclides in the Tidal Thames compared with that in the River Aire.

It can be seen that there are two main factors influencing the radiological impact of disposal to urban sewers using incinerators, the size of the sewage works (in terms of total sludge and ash production) and the size and composition of the source term.

14.4 Implications of Disposals at Authorised Limits

At Knostrop the estimated doses to members of the public based on authorised limits are either an order of magnitude greater than the doses based on typical discharges in 1997 or three orders of magnitude greater, depending on whether the dominant radionuclides are Iodines or C-14. For effluent or atmospheric release pathways, I-125 and I-131 are responsible for most of the dose, and the typical discharges are roughly an order of magnitude lower than the authorised discharges. For incinerator ash to landfill or sludge to landfill scenarios, C-14 is dominant and the doses reflect the typical and authorised discharges of this radionuclide; authorised discharges are three orders of magnitude greater than typical discharges in 1997.

Doses at the sewage treatment works based on authorised limits are about three times greater than those based on typical discharges. The dominant radionuclide at the STW is I-131 and the authorised disposal of this radionuclide is about three times greater than the amount typically disposed.

At Beckton the estimated doses at the works and from incineration based on authorised limits are ten times higher than the predicted doses based on typical disposals for 1997, whilst doses from effluent are 50 times higher. The doses were dominated by Rb-84 and I-131 at the works and C-14, P-32 and I-131 in effluent. The disposal limits interpreted from the authorisation assumed in the assessment were 26, 3.5, 200 and 10 times higher than the typical discharge of Rb-84, I-131, C-14 and P-32 respectively.

In both catchments, the assessment based on the typical disposals can only be considered as representative of 1997. Whilst there is no reason to consider 1997 as an atypical year, holders of authorisations are entitled to dispose at their full monthly limits without notifying the Environment Agency, so in other years disposals could be considerably different.

There is some flexibility over what disposals could occur under the authorisations because radionuclides are not always specified by name and isotope on all the authorisations. The hospitals prefer this system for reasons outlined in Section 14.7.1, however, as a result in some cases it was necessary to make some conservative assumptions about the radionuclides and quantities that were disposed. The source term used in the assessment of both works based on the authorisation was conservative. The assumptions made and the source term for Beckton and Knostrop are shown in Chapter 6 and Chapter 10 respectively.

More accurate assumptions could be made if authorisations contained radionuclide specific limits for all significant radionuclides. However, the advantages must be weighed against the disadvantages discussed in Section 14.7.1.

14.5 Dose Reduction Options

At both Knostrop and Beckton, all the predicted doses are below the proposed annual dose limit for members of the public of 1000 μSv y^{-1} and the maximum public dose constraint of 300 μSv y^{-1}. All the predicted doses at Knostrop sewage works from maximum authorised and typical disposals are above the threshold for optimisation. Doses to the public from effluent from maximum authorised levels of disposal are also above the threshold for optimisation. At Beckton predicted doses based on authorisations were above 20 μSv y^{-1} at the STW and to the public from effluent discharges. Predicted doses based on typical discharges were above 20 μSv y^{-1} only for sludge press workers at the STW.

Given that discharges at levels near the authorised limits could occur, further consideration of these doses is appropriate. Dose reduction options need only be considered if one of the premises is found to be responsible for the majority of the dose.

At Knostrop the doses to workers at the STW are dominated by I-131. If an authorised monthly disposal of 4.5 10^{10} Bq results in a dose of 67 μSv y^{-1}, or 221 μSv y^{-1} at the sludge press, then any individual disposal of greater than 1.3 10^{10} Bq $month^{-1}$ will give rise to doses of more than 20 μSv y^{-1} to average workers and disposals of greater than 4.1 10^{9} Bq $month^{-1}$ would result in doses to sludge press workers of greater than 20 μSv y^{-1}. In the Knostrop catchment one hospital has an authorisation to discharge I-123/I-125/I-131 at 2.8 10^{10} Bq $month^{-1}$ and another has an authorisation to discharge at 1.5 10^{10} Bq $month^{-1}$. The third hospital in the Knostrop catchment has an authorisation to discharge 'other $\beta\gamma$ radionuclides at up to 2 10^{10} Bq $month^{-1}$, since this hospital has no separate Iodine authorisation, I-131 could be disposed under the 'others' authorisation. Authorised disposals from each hospital individually could therefore result in doses above 20 μSv y^{-1} the threshold for optimisation. The typical disposal of I-131 and 'others' at each of the hospitals is below 1.3 10^{10} Bq $month^{-1}$, so the typical doses from each establishment to average sewer workers must be below 20 μSv y^{-1}. One of the hospitals in Leeds typically disposes more than 4.1 10^{9} Bq $month^{-1}$ of I-131 so a worker spending all of his time at the sludge press could receive a dose greater than the threshold for optimisation as a result of discharges from this hospital.

Doses to the public from discharges at the authorised limit from the disposal of effluent are dominated by C-14. If a monthly disposal of 8.1 10^{10} Bq $month^{-1}$ results in a dose of 120 μSv y^{-1} then any individual disposal above 1.4 10^{10} Bq $month^{-1}$ will cause a dose of above 20 μSv y^{-1}. One disposer in Leeds is authorised to dispose of 8.0 10^{10} Bq $month^{-1}$ of C-14 and H-3 combined, this authorisation could give rise to a dose of above 20 μSv y^{-1}

At Beckton, doses to workers at the STW from authorised disposals are dominated by Rb 84 and I-131. The dose from discharges at the authorised limit to an average worker from I 131 is 42 μSv y^{-1} and to a sludge press worker is 153 μSv y^{-1}, this results from a monthly disposal of 2.4 10^{11} Bq. Therefore any individual disposal above 1.1 10^{11} Bq $month^{-1}$ would give rise to an average worker dose above 20 μSv y^{-1} and disposals above 3.1 10^{10} Bq $month^{-1}$ would result in doses to sludge press workers of greater than 20 μSv y^{-1}. There are no individual disposals to Beckton of greater than 1.1 10^{11} Bq $month^{-1}$, but there are two disposers to Beckton who are authorised to discharge more than 3.1 10^{10} Bq $month^{-1}$ and one of these disposers typically discharges I-131 at around this level. There is only one disposer of Rb-84 whose authorisation would give rise to a dose of 160 μSv y^{-1}; this is above the threshold for optimisation.

Doses to the public from discharges at the authorised limit from effluent are dominated by C-14. If a disposal of 1.2 10^{11} Bq $month^{-1}$ results in a dose of 28 μSv y^{-1} then any individual disposal above 8.6 10^{10} Bq $month^{-1}$ will give a dose of greater than 20 μSv y^{-1}. One disposer in the Beckton catchment has an authorisation for C-14 and H-3 combined of 1.0 10^{11} Bq $month^{-1}$ which, if used for C-14, would give rise to a dose of greater than the threshold for optimisation.

Since there are some cases where a single premises could be responsible for doses above the threshold for optimisation if they discharged up to their authorised limit, some consideration of dose reduction options in both catchments is appropriate. For the particular case of sludge press workers, typical disposals from some premises may also result in doses above the threshold for optimisation.

The main dose reduction options typically involve reducing the discharges of radiologically significant radionuclides, which in this study were found to be I-131, Rb-84 and C-14. The main discharge reduction option for I-131 and Rb-84 is decay store. The practicalities of this option and possible alternatives for other radionuclides at the premises of disposers are presented below.

14.5.1 Decay Store - Practicalities

Consideration of decay storing hospital wastes containing I-131 would be most appropriate in situations where I-131 is used in thyroid ablation treatments. Decay storing typically involves construction of tanks into which waste sewage is diverted and held for the desired time. The number and type of tanks required would vary depending on the frequency of treatment and length of desired decay storing. Patients are usually treated early in the week and sent home after a few days. Sewage could be held in one tank for the remainder of that week, and either discharged at the beginning of the next week giving 7 days decay, (0.8 half lives) or retained for another week if a decay time of two half lives is required. A total decay time of 13 - 14 days is equivalent to 1.75 half-lives. If treatments occur every week, a second tank would be required for the second weeks' treatment. Therefore an organisation requiring 2 half-lives decay may need only two decay tanks. Decay store of discharges of Rb-84 would require longer storage (>33 days). However, the volume and nature of the wastes accompanying Rb-84 are much less than I-131 so decay storage may be more practical for this radionuclide.

14.5.2 Other dose reduction measures

Where decay storage is not an option, there are two main alternatives for reducing discharges. It may be possible to intervene in the waste stream itself and to extract radionuclides and transfer them to another media such as a solid. Options could include sorption onto solids or ion exchange techniques. It may be possible to intervene before the discharge to the drainage system occurs and retain the waste radionuclides.

There are several problems with employing the techniques. Firstly, a different waste is created that requires disposal. Secondly, although there are some examples of large-scale plant for removing radionuclides at sites such as Sellafield there are no examples of application in small user situations. These alternatives are therefore unlikely to be available, suitable or affordable for small users in the near future.

14.5.3 Use of dose reduction options in the catchment of Knostrop and Beckton

Doses to workers at Knostrop sewage works are dominated by iodine I-131 releases from the Leeds Hospitals. If I-131 wastes were subjected to decay store over one week the activity concentration would be halved and the resulting discharges and doses reduced by a factor of 2. If 1997 discharges are typical, the predicted doses would be approximately halved to around 11 μSv y^{-1} (risk of fatal cancer reduced from approximately 1 10^{-6} y^{-1} to approximately 5 10^{-7} y^{-1}). Discharges based on the authorised limits would be reduced to around 30 μSv y^{-1} (risk of fatal cancer reduced from approximately 3 10^{-6} y^{-1} to approximately 1.5 10^{-6} y^{-1}). Doses to sludge press workers would be reduced to around 40 μSv y^{-1} from typical discharges and 110 μSv y^{-1} from maximum authorised discharges.

Only one hospital treats in-patients in Leeds, therefore decay storage would only be of benefit at this establishment. This hospital is responsible for around 130 μSv y^{-1} of the maximum dose and about 60 μSv y^{-1} of the typical dose to sludge press workers. Addition of a decay store system to existing buildings is very costly, (£200,000) which could divert funds from primary care. It is not clear whether a decay store at this particular hospital would be indicated without a detailed optimisation study. However if discharges remain at current levels, the results indicate that decay storage is not likely to be the optimum solution. In addition, it is likely that the installation of decay storage would result in increases in doses to hospital staff.

The government's review of radioactive waste management policy[38] discusses the tolerability of risk and thresholds for optimisation. It is recognised that there is an upper limit, beyond which a risk would be unacceptable regardless of the benefit which society derived from the activity and a lower level, below which the risk was negligible in comparison with the other risks we run in our daily lives, and therefore broadly acceptable. The area in between is the 'tolerability' region, in which the risk is tolerable only if it is as low as reasonably practicable (ALARP), where to reduce the risk further would involve disproportionately high costs.

Doses to workers at the Beckton sewage works based on the authorised discharges are dominated by Rb-84 releases from one disposer. However, doses from the typical discharges of Rb-84 during 1997 were around 8 μSv y^{-1}. Decay storage of this radionuclide would need 33 days to halve the discharge. If the volumes of waste associated with the Rb-84 are small, decay storing over this time may be possible. An alternative would be to minimise the use of this radionuclide. If the disposer were to cease discharging this radionuclide, doses at the sewage works would decline to around 10 μSv y^{-1} (from typical discharges). Therefore, a simple dose reduction option may be to establish what use is being made of the Rb-84. As part of the study, the disposer was contacted and they indicated that the practice which gave rise to Rb-84 wastes has now ceased. It would appear appropriate to confirm with the disposers that disposals have ceased, and if so, to revise the authorisation.

Without discharges of Rb-84, the doses at Beckton sewage works will be dominated by iodine-131. The main approach for reducing iodine-131 disposals is decay storage. Using typical discharges, the predicted doses to average workers are below 20 μSv y^{-1} (risk of fatal cancer reduced below about 1 10^{-6} y^{-1}), however, higher doses are received by workers spending all of their time at the sludge press. Although these doses are above 20 μSv y^{-1}, as discussed earlier in this section this is a combination of discharges from at least nine hospitals, only two of these hospitals have authorisations which would result in maximum doses of more than 20 μSv y^{-1} and there is only one hospital in the catchment which typically discharges at a level which would cause a dose of around 20 μSv y^{-1} to sludge press workers. Hence it is concluded that decay storage is unlikely to be indicated at any of the disposing premises.

Doses to members of the public from discharges at the authorised limits at both Knostrop and Beckton are dominated by discharges of C-14. At Beckton, the predicted discharges based on typical discharges for 1997 gave rise to doses well below 20 μSv y^{-1}. Provided discharges remain less than 70% of the total authorisations in the Beckton catchment, then doses will remain below 20 μSv y^{-1}. All doses from typical discharges at Knostrop are also below the levels of optimisation. Only doses from effluent of 19 μSv y^{-1} are close to the threshold for optimisation. Although dose reduction measures are not necessary, it may be appropriate to monitor the environments receiving treated effluent to determine the actual levels of C-14 and to determine the extent that fish are caught and consumed in vicinity of the effluent discharge points around Knostrop.

14.6 Implications of Increased Discharges

If disposals of I-131 were to be increased to nearer the authorisation limits in the Knostrop catchment discharge reduction measures may need to be considered in more detail. Discharges of some radionuclides such as I-131 could be increased by up to a factor of three under the existing authorisations without recourse to the regulator, because of the way the authorisations are written. Therefore it may be appropriate that when the disposal authorisations are renewed, the significant radionuclides that are to be discharged should be identified by isotope and individual limits placed on them. This may help reduce significant increases in discharges of radionuclides such as I-131 occurring.

The disposals of C-14 made in 1997 to Knostrop would have to increase by a factor of over 7000 before the maximum dose constraint for members of the public is likely to be approached. Increases of up to a factor of 3000 could occur under the existing authorisations. Discharges of I-131 could be increased by up to a factor of 10 before the maximum public dose constraint was approached, or a factor of 5 if sludge press workers are considered. If the main disposer of I-131 planned to increase discharges by more than a factor of 3, decay storage for I-131 may need to be considered in more detail.

If discharges of Rb-84 in the Beckton catchment were increased by a factor of 50 over the 1997 level, the maximum dose constraint for members of the public may be reached. Under the existing authorisation an increase by a factor of 30 would be possible. Discharges of I-131 would have to be increased by a factor of 20 before the dose constraint was approached or a factor of six before doses to sludge press workers approach the dose constraint, discharge increases of up to a factor of 3 are possible under the existing authorisations. If a significant increase in use of radionuclides (factor of 5 or more) such as I-131 is expected in future, new hospital buildings may need to consider the inclusion of decay stores in the design of the sewage system. The additional cost of including decay stores should be considered in an optimisation study, which may conclude that it is not optimal.

If increases in discharges are to be authorised, the additive effect of all the significant disposals to one sewage works needs to be taken into account. Although every application for an authorisation made by every disposer must be supported by a dose assessment, these currently do not take into account the contribution of other disposers to the total discharge. When all discharges are combined at the works and subject to the various processing activities the radiological impact may become more significant.

14.7 Consultation with Disposers

The larger disposers in both catchments were contacted to gauge feelings on issues such as decay storage, the revision of authorisations to include more specific radionuclide discharge levels, practices that could potentially reduce doses within hospitals, the feasibility of decay storage, alternative medical practices to thyroid ablations and trends in radionuclide use for the future.

14.7.1 Use of radionuclide specific authorisations

At the Leeds hospitals, the use of radionuclide specific authorisations caused some concern. There are generally a large number of departments within a hospital, and it was felt that the grouped authorisations allow flexibility in handling the fluctuations of patient demands. A further problem would be the costs of changes to computerised record keeping systems, although individual departments keep radionuclide specific records, the computer system totals disposals into the authorisation groups.

The introduction of nuclide specific authorisations for the University would not be a concern. The University of Leeds uses mainly I-125 (given as I-125/I-131 in the current authorisation) and under 'others' P-32, P-33 and S-35 are disposed. It is unlikely that another isotope will be required in the foreseeable future.

The hospitals in Central London were not asked to comment on the use of radionuclide specific authorisations.

14.7.2 Discharge reduction options

Consultation on decay storage at the Leeds Hospitals found that the Environment Agency do not currently sanction the storage of any liquid hospital wastes. Other feelings were that in general no benefits or reductions in risks would be obtained by their introduction, especially considering increases in worker doses and risks associated with construction work. It was recognised that the inclusion of decay stores in the design of new facilities would be viewed differently.

In the Beckton catchment the suggestion of adding decay storage facilities to existing treatment buildings, in order to reduce discharges of radionuclides was not considered possible. Addition of decay storage facilities was considered totally impractical in terms of costs, changes to the existing plumbing and available space. Concerns were also expressed on monitoring, maintenance, contamination of pipework, cleaning, leakage, human error etc and feelings in general were that its implementation would do more harm than good. Most of the hospitals were not entirely clear on what a workable decay storage system would actually consist of. It was expected that an increase in staff doses were likely as a result of storage tank operation, but that the level of dose increase could not easily be quantified. If decay storage was implemented, a decay period of 2-4 weeks was considered by those consulted as a sensible option for maximum benefit. The number of tanks expected to be required varied considerably and included a single tank (low ablation treatment frequency), 2 -3 tanks (2 to fill and decay, one as overflow), or at least 4 separate tanks or systems (therapy rooms being located in various parts of the hospital).

Implementation of decay storage could, however, be considered at the design phase of new facilities.

Of those consulted little was known of discharge reduction measures such as ion exchange or absorption processes apart from the fact that it is likely to be expensive.

14.7.3 Future trends in the use of radionuclides

London hospitals were also asked to comment on the likely future trends in radionuclide use. It was considered that currently clinical pressures are for the increased use of radionuclides, but not always I-131. Apart from treatment loads being moved between hospitals, the general use of I-131 and Tc-99m was expected to remain fairly constant, although I-125 radio immunoassays are tending to be being replaced by other diagnostic methods and some nuclear medicine techniques were being replaced by MRI. The use of Bactec microbiology, for HIV and TB testing with C-14, is being replaced with a non-radioactive technique. Use of some other radionuclides are on the increase including Y-90 tumour therapy, Sr-89 therapy, In-111 therapy and I-125 seeds for prostate cancer (there is a concern however over the detachment of these seeds from its implanted holder and subsequent discharge with bodily fluids).

Overall the authorisations for the universities in Leeds are not likely to change in the foreseeable future. Although refurbishment at Leeds Metropolitan University is now complete there is no intention to apply for a new authorisation. Over the last few years there has only been a slight move from the use of P-32 to P-33 and a reduction in iodination at the University of Leeds.

14.7.4 Exposure of staff

Within the hospital environment the most significant nuclides used were confirmed as I-131 and Tc-99m. Of those consulted it was rare however for doses to be detected on hospital staff personal TLDs above the TLD detection limit of around 50 μSv. It was thought likely that the doses from Tc-99m would be greater than those from I-131 as a result of the frequency and quantity of Tc-99m use and differences in treatment procedures. Tc-99m patients are more numerous and are allowed more staff contact, local rules restricting patient contact and describing local shielding exist for I-131 patients.

The doses received by hospital staff are generally maintained at low levels through the agreed dose management policies and procedures, systems of work, local rules, review and audits. Specific actions used to keep doses to staff low include sharing of workloads within radio-pharmacies, development of automated Tc-99m dispensing systems, I-131 shielded therapy rooms, moveable shielding, use of injection shields and time restrictions near patients.

14.8 Implications for Health Care Provision using Radionuclides for Diagnosis and Therapy

In the context of this study there are no radiological protection reasons for revising the way radionuclides in health care are used or disposed. In the two cases studied, the doses arising from authorised disposals of radionuclides used in diagnosis or therapy are not significantly above the threshold for optimisation. If use and disposal of radionuclides such as I-131 were however, increased by a factor of 3-5 or more, then a review of the disposal options might be appropriate.

15 APPLICABILITY OF THE STUDY RESULTS TO OTHER SEWAGE WORKS

The method and results in this study can be used to provide an estimate of doses to workers and public for other sewage works. The study showed that external dose is the most important pathway for workers at the sewage treatment works, and ingestion of fish following discharge of effluent to rivers is the most important pathway for members of the public. Both of these pathways can be generalised so that a dose estimate can be obtained by calculating the activity concentration in the media (effluent, sludge or ash), and scaling by the dose rate per unit concentration. Doses to the public from some other environmental disposal options of the sewage sludges can also be estimated as described below.

15.1 Doses to Workers

Doses to workers are estimated from the activity concentration in various waste media (incoming effluent, treated effluent, separated sludges, dewatered sludges and ash). This study indicated that the most important media in terms of radiological impact are separated sludges, dewatered sludges and incinerator ash. The activity concentration can be calculated from:

Act conc = Adis / Qmedia

Where

Act conc = Activity concentration in media (effluent, sludge, dewatered sludge cake, or ash)(Bq m^{-3})

Adis = Activity discharge rate of nuclides into sewers (Bq per y, month, day etc)

Qmedia = Quantity of media (effluent, passing through or produced per unit time (m^3 per y, month, day etc).

The activity discharge rate of radionuclides is usually the total discharge received by a works from all the dischargers in the works catchment per unit time. The quantity of media passing through or produced is sewage works specific.

The activity concentration in the important media at various stages in the works (effluent, separated sludges, dewatered sludge cake or ash) can be calculated from:

Eff conc = Act conc * Reff

Slud conc = Act conc * (1-Reff)

Ash conc = Act conc * (1-Reff) * (1-Vfrac)

Where:

Eff conc	=	Activity concentration in effluents (Bq m^{-3})
Slud conc	=	Activity concentration in separated sludge or dewatered sludge cake (Bq m^{-3})
Ash conc	=	Activity concentration in ash (Bq m^{-3})
Reff	=	Removal efficiency (Table D12)
Vfrac	=	Incinerator volatilisation fraction (Table D14)

Removal efficiency (Reff) is an element specific term, which represents the ratio of the activity leaving the works in final treated effluent, divided by the total activity entering the works in the raw effluent. Activity lost from the effluent during treatment is assumed to be transferred to solids. The removal efficiencies used in the work for each radionuclide are shown in Table D12. A removal efficiency value of <0.5 means that the majority of the activity is removed to solid from the effluent, whilst a value >0.5 means the opposite. In general radionuclides that tend to sorb to suspended solids have low removal efficiency values, (eg Am-241, Tl-201), whilst elements that tend to remain in solution have high values (eg H-3, S-35).

The volatilisation fraction (Vfrac) is the ratio of the total activity of each radionuclide passing up the stack, to the total activity introduced into the incinerator in sludge cake. Radionuclides that are not volatile tend to remain in ash and have low volatilisation fractions (eg Cr-51, Ga-67), whilst radionuclides that are volatile pass up the stack and have high volatilisation fraction (eg Iodines, H-3, C-14). External dose rates at the works 1 m above a semi-infinite tank of sewage can be calculated from the activity concentration and unit dose rates:

Ext dose = Extudr * {Eff conc or Slud conc or Ash conc} * Occ

Where:

Ext dose	=	Effective external dose (Sv y^{-1})
Extudr	=	External dose rate per unit concentration (Sv h^{-1} per Bq m^{-3}) (Table D5)
Occ	=	Occupancy (h y^{-1}) (usually 2000 h per y)

Internal exposure from inhalation or inadvertent ingestion rate may also be calculated using the equations shown in Appendix B at section 5. However, the study has shown that external dose is the most important pathway for all the radionuclide considered.

15.2 Doses to Public

For members of the public the doses from disposals of effluent to river, disposals of sewage sludge to farmland and to landfill can be applied to other sewage works.

Treated effluent to river

Doses to the public from disposals of treated effluent to river can be estimated from:

River Dose = Rdpu * Adis *Reff/Rvol

Where:

River Dose = Estimated annual dose to members of the public from discharges of treated effluent to river (Sv y^{-1})

Rdpu = Estimated annual dose to members of the public per unit disposal of activity to a river flowing at 1 m^3 s^{-1} (Sv y^{-1} per GBq y^{-1}) (Table C7)

Rvol = Annual average volumetric flow of receiving river (m^3 s^{-1})

Adis = Annual discharge of effluent to sewage from all disposers in catchment (GBq y^{-1})

Reff = Removal efficiency (Table D12) (See also section 15.1).

Treated sludge to land

Doses to the public from sewage sludge disposed to land can be estimated from:

Fooddose = Fdpu * Slud conc

Where:

Fdpu = Doses from food produced on land treated with sewage sludge of unit concentration (Sv y^{-1} per Bq kg^{-1}) (Table C18)

Slud conc = Activity concentration in sludge (Bq kg^{-1})

The term Fdpu is for a sludge that is 5% solid and 95% water (density of 1000 kg m^{-3}). The activity concentration in sludge (Slud conc) is sewage works specific and may be calculated as shown in section 15.1. The concentration in a sludge that is 5% solid and 95% water should be derived.

Sludge and ash to landfill

Doses to the public from disposal of sewage sludge to landfill may be estimated from:

Land D_{sludge} = Ldpu * Adis * (1-Reff)

Doses to the public from disposal of sewage sludge ash (remaining after incineration of sludge) to landfill may be estimated from:

Land D_{ash} = Ldpu * Adis * (1 – Reff) * (1 – Vfrac)

Where:

Land D_{sludge}	=	Estimated annual dose to members of the public from disposals of sludge to landfill (Sv y^{-1})
Land D_{ash}	=	Estimated annual dose to members of the public from disposals of incinerated sludge ash to landfill (Sv y^{-1})
Ldpu	=	Estimated annual dose to members of the public per unit disposal of activity to a landfill (Sv y^{-1} per GBq y^{-1}) (Table C17)
Adis	=	Annual discharge of effluent to sewage from all disposers in catchment (GBq y^{-1})
Reff, Vfrac	=	See section 15.1.

15.3 Parts of the Study that cannot be Used for Other Sewage Works

Doses from incineration and from disposals of effluents to inter-tidal waters are site specific and cannot be applied readily to other sewage works.

Incineration doses are dependent on stack height and the locations of nearest resident and food producing farm. Both of these factors are folded into the dose per unit disposals given in the study (Tables C11 and C12). For other incinerators of different height and where residents and farms are located nearer or further from the stack, the predicted doses may be different. Therefore it would only be appropriate to scale the values given in this report for similar stack heights and exposed group locations.

Doses from disposals to tidal waters are dependent on basic oceanographic features (water body volume, water resident time) which are included in the dose per unit disposals to the Tidal Thames (Table C4). Therefore it is not appropriate to scale the dose per unit disposals given.

15.4 Future Work

Future work could be directed at providing a more complete database for incineration and disposals to tidal waters. This could comprise doses per unit discharge from incinerators of various realistic heights, exposed group locations with realistic generic habits. Doses per unit discharge to appropriate or representative inter-tidal/coastal water bodies representative of those around the England and Wales may also be needed. Alternatively, databases could be generated for additional specific sewage systems. Consideration should also be given as to whether the database should be expanded to include any other radionuclides.

16 SUMMARY AND CONCLUSIONS

16.1 Summary

The practice of disposal of small amounts of radioactive wastes to sewers subject to authorisation is used in the UK and other countries for hospital and research wastes.

A review of current sewage treatment practices in the UK, and the disposal options for treated effluents and sludges was undertaken. Treatment practices in Europe and North America have been considered and future trends in the treatment of sewage identified. The likely effects of sewage treatment and disposal on radionuclides have been described.

The uses and behaviour of seventeen radionuclides that are authorised for disposal to sewers were reviewed. The main uses of the radionuclides were identified. A description of each of the radionuclides including radioactive half-lives, emission energies and usage is given. Information on the behaviour of radionuclides during sewage treatment is given in terms of K_d and removal efficiencies. Limited information was found on the behaviour of radionuclides at sewage works.

A sewer assessment model (SMART) was developed for estimating doses from disposal of seventeen radionuclides to sewers. The model provides estimates of concentrations of radionuclides during collection, treatment and disposal of effluents and sludges and the resulting doses to workers and members of the public. The structure of the model and the processes modelled were compared with other similar models.

The eleven main sewage catchment and treatment systems in London and the main works in Leeds (Knostrop) were described in terms of catchment area, flows, treatment processes and sludge production rates. The largest sewage works (at Beckton) in London was identified for a more detailed study.

Sixty five authorised disposers of radioactive material to the Beckton catchment and six disposers to the Knostrop catchment were identified. The major disposers and important radionuclides disposed to each works were selected by performing a screening radiological assessment. Information on the disposal authorisations of the major disposers and on typical disposals made in 1997 was used to establish a source term for each STW for use in the model. Some disposal authorisations did not name radionuclides and in these cases conservative assumptions about the radionuclides disposed were made. At both works the largest discharges permitted under the authorisations were of C-14, I-131, I-125 and Tc-99m.

The new model (SMART) was used to estimate doses from disposals of radionuclides to Beckton and Knostrop from the major disposers based on an estimate of the permitted authorised disposals and the typical disposals (based on 1997 data). Doses were calculated to workers unblocking a sewer pipe, workers inside man accessible trunk pipes, at the sewage works, and at the incinerator plant and in the sludge pressroom. Doses were calculated to members of the public from disposals of treated effluent, waste gases produced during incineration of sludge and disposal of ash.

Based on the disposal authorisations, estimated doses to workers were 210 μSv y^{-1} at Beckton, from I-131 and Rb-84 in incinerator cake and ash. It was noted that Rb-84 discharges ceased before the incinerator came on line. Without Rb-84, doses were around 50 μSv y^{-1} mainly from I-131 at the incinerator or 161 μSv y^{-1} from sludge cake. At Knostrop doses were around 80 μSv y^{-1} at the works and 238 μSv y^{-1} in the sludge pressroom. Estimated doses to the public were 30 and 180 μSv y^{-1} from disposals of effluent from Beckton and Knostrop respectively. At both locations doses from incineration were < 2 μSv y^{-1}.

Based on typical disposals the estimated doses to workers at sewage works were 19 μSv y^{-1} at Beckton, from Rb-84 and I-131 in incinerator cake and ash. Without Rb-84, doses were around 13 μSv y^{-1} from I-131 and to sludge press workers 43 μSv y^{-1} resulting mainly from I-131. At Knostrop works doses were 25 μSv y^{-1} from I-131, with sludge press workers receiving a dose of 79 μSv y^{-1}. Estimated doses to the public were 0.6 and 19 μSv y^{-1} from disposals of effluent from Beckton and Knostrop respectively from C-14 and P-32. At both locations doses from incineration were < 1 μSv y^{-1}.

At the sewage works, the majority of the dose arises from external exposure. Inadvertent ingestion and inhalation pathways are relatively unimportant for the radionuclide mix considered in this study. The highest external dose rates occur where activity concentrations are at their highest, ie separated sludges, dewatered sludges and ash.

Monitoring for radionuclides in sewage materials at Knostrop and Beckton in summer and winter was carried out. Incoming effluent, primary and secondary sludges, treated effluent, incinerator cake and ash were sampled and analysed for up to thirteen of the discharged radionuclides, fallout and natural radionuclides were also measured. In general, most of the radionuclides were below detection limits in most of the sampled media. However, most of the radionuclides were detected in sewage incinerator cake and in residual ash. Results of the sampling were compared with the model and used to assess the behaviour of radionuclides during incineration.

The measurements confirmed the predicted concentrations in sludge, ash and cake and indicated that C-14 in effluent is overestimated by the model.

TLDs were placed around the sewage works and incinerator plant to investigate possible doses around Beckton and Knostrop. The additional dose from artificial radionuclides was less than about 100 μSv y^{-1} in both cases, but could not be quantified further. The doses predicted by the model using typical discharges were less than 100 μSv y^{-1}.

The use of incinerators for sewage sludge and the behaviour of radionuclides in incinerators was assessed, using the literature and the results from sampling of incinerator cake and ash at each sewage works. Metallic radionuclides tended to remain in the ash, whilst volatile elements such as iodine, carbon and hydrogen tended to be lost up the stack. It was not possible to justify a tracer study.

The predicted doses to workers and public from disposals to the Knostrop and Beckton sewage works (based on both disposal authorisations and typical disposals) are well below the dose limits and constraints for members of the public. Predicted doses to workers at the works based on authorisations from I-131 and Rb-84 and the public from disposed effluent containing C-14 were above the threshold for optimisation of $20\mu Sv\ y^{-1}$ at both sites.

The case for decay storage at the main disposers of I-131 in both catchments is weak if discharges remain near their current levels. In the Beckton catchment it may be appropriate to review the use of the disposal authorisation for Rb-84.

Provided C-14 discharges remain near typical 1997 levels, doses around the Beckton outfall should remain well below the threshold for optimisation. At Knostrop it may be sensible to assess how much aquatic foods are consumed from the River Aire in the vicinity of the outfalls. It may then be appropriate to monitor the identified foods to establish if the predicted doses are over estimates.

It is suggested that when authorisations for radionuclide disposals are renewed in the Beckton and Knostrop catchments, the practice of grouping radionuclides together and giving one overall limit be replaced by providing individual limits for the more important radionuclides such as C-14, I-131 and I-125. Where "other β/γ" is used as generic grouping for less significant radionuclides, an indication of the main radionuclide to be discharged under this title would be useful.

When authorisations from individual disposers are assessed, it may be necessary to consider the contribution from other disposers in the catchment of a sewage works. The model SMART allows this to be done. In addition, if patients are encouraged to return home or go to smaller hospitals earlier, then discharges may occur to other sewage treatment works and this needs to be taken into account.

The practice of disposing radionuclides to sewers at current levels remains acceptable on radiological protection grounds in the Beckton and Knostrop catchments. If significant increases in discharges of I-131 are expected, an optimisation study may be required.

Sewage sludge incineration is acceptable on radiation protection grounds. In the cases studied, sewage sludge incineration gave rise to very low doses to members of the public and doses to workers were below the principal dose constraint for members of the public. It is unlikely that incineration at other locations would give rise to significantly higher doses than those estimated at the two sites studied.

The results of the study can be used to provide an estimate of doses from discharges to other sewage works. A set of formulae have been proposed which can be used with data in the report to estimate doses from the some of the important pathways for sewage treatment works. Doses can be calculated to workers at sewage treatment works and incinerator. Doses can be calculated to the public from disposal of radionuclides in treated effluents to rivers, in sewage sludges applied to land and disposed. However, exposure from effluent discharge to tidal waters and incineration of sewage sludge cannot be calculated for other works using the information in this study. An extended database is needed for this.

16.2 Overall Conclusions

Two sewage works were studied in detail for this report, the largest sewage works (at Beckton) in London and the main works in Leeds (Knostrop). Sixty-five authorised disposers of radioactivity to the Beckton catchment and six to the Knostrop catchment were identified. A model (SMART) was developed to look at the overall impact of all authorised discharges to the catchment. Based on the disposal authorisations, estimated doses to workers at Beckton were 210 μSv y^{-1} to average workers and 207 μSv y^{-1} to those workers spending all of their time in the sludge pressroom. At Knostrop doses were estimated at 80 μSv y^{-1} at the works and 238 μSv y^{-1} in the sludge pressroom. Estimated doses to the public were 30 and 180 μSv y^{-1} from disposals of effluent from Beckton and Knostrop respectively. At both locations doses from incineration were < 2 μSv y^{-1}.

Doses were also estimated on the basis of typical disposals made in 1997 in both catchments. On this basis doses to workers at Beckton STW were around 19 μSv y^{-1} with 45 μSv y^{-1} to sludge press workers. At Knostrop, worker doses were 25 μSv y^{-1} with 79 μSv y^{-1} to sludge press workers. Estimated doses to the public were 0.6 and 19 μSv y^{-1} from disposals of effluent from Beckton and Knostrop respectively. At both locations doses from incineration were < 1 μSv y^{-1}.

All of these doses are well below the dose limit to members of the public of 1000 μSv y^{-1} and below the maximum dose constraint of 300 μSv y^{-1} for a single practice. The majority of doses estimated from typical disposals are below the threshold for optimisation of 20 μSv y^{-1}. However, doses at the sewage works, and to the public as a result of discharges from the sewage works are a result of disposals from a number of establishments. The threshold for optimisation should only be considered when looking at discharges from a single disposer. SMART was used to investigate this and there are some cases where doses at the STW may be above the threshold for optimisation resulting from the disposals of only one establishment. However, the case for decay storage at the premises of the main disposers of I-131 in both catchments is weak, if discharges remain near their current levels.

Sewage sludge incineration is considered to be acceptable on radiation protection grounds. In the cases studied, sewage sludge incineration gave rise to very low doses to members of the public and doses to workers were below the maximum upper value of the dose constraint for members of the public. It is unlikely that incineration at other locations would give rise to significantly higher doses than those estimated at the two sites studied.

The results in this study can be used to estimate some of the important doses from discharges to other sewage works. It would be useful to extend the database to study other urban sewer systems. The main conclusion is that the practice of disposing radionuclides into the sewer system is a safe disposal option and is an acceptable option in the urban situations studied.

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Table 1 **Water consumption and sewage sludge production in England and Wales**

	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993
Water consumption											
England and Wales, Ml d^{-1}	16260	16498	16577	16961	16850	16626	17021	17148	16967	16638	16433
England and Wales, m^3 y^{-1}	$5.9\ 10^9$	$6.0\ 10^9$	$6.1\ 10^9$	$6.2\ 10^9$	$6.2\ 10^9$	$6.1\ 10^9$	$6.2\ 10^9$	$6.3\ 10^9$	$6.2\ 10^9$	$6.1\ 10^9$	$6.0\ 10^9$
Sludge arisings (dry solids)											
England and Wales, t y^{-1}	-	-	$8.84\ 10^5$	$8.96\ 10^5$	$9.00\ 10^5$	$9.54\ 10^5$	$9.58\ 10^5$	$9.45\ 10^5$	$9.45\ 10^5$	$9.07\ 10^5$	$9.02\ 10^5$

Table 2 **Effects of the duration of sewage treatment on radionuclides**

Process	Duration of process, h		Fraction of radionuclide undecayed [B,C]			
	Likely range	Estimate	Tc-99m (6.02 h)	Ga-67 (78.3 h)	In-111 (67.9 h)	Tl-201 (73.6 h)
1. Sewer transit	< 1 - 10	5	0.56	0.96	0.95	0.95
2. Preliminary treatment (detritors)	< 1	0.1	0.56	0.96	0.95	0.95
3. Primary treatment (sedimentation)	2 - 6	4	0.35	0.92	0.91	0.92
4. Secondary treatment						
Percolating filters	2 - 6	4	0.22	0.89	0.87	0.88
Activated sludge	4 - 10	7	0.16	0.87	0.85	0.86
5. Tertiary treatment						
Filtration	2	2	-	-	-	-
Micro straining	< 1	0.1	-	-	-	-
Chemicals	< 1	0.1	-	-	-	-

Notes:

A Tritium and carbon-14 have not been included as their half-lives are sufficiently long to lead to insignificant decay over these timescales

B These values do not take into account removal by sorption onto filtered out solids.

C Half-lives of radionuclides have been presented in brackets

Table 3 Hydrological conditions in storm events for a typical urban area

	Antecedent dry period	Depth of water, mm	Storm duration	Peak rainfall intensity, mm h^{-1}
Nottingham[4]				
Maximum	27 d 12 h	25.1	11 h 2.5 m	54.86
Minimum	6 h	0.26	13 m	0.76
Average	4 d 17.5 h	6.95	4 h 5 m	17.0
Munich[5]				
Average	-	24.8	3 h 26 m	-

Notes:
A Maximum and minimum values are the highest and lowest recorded value in a category and do not necessarily relate to a single storm event. The average is the arithmetic mean of 15 storm events.
B Data have been obtained from reference 4

Table 4 Sludge stabilisation treatments in a number of European countries

Country	Sludge production (t y^{-1})[A]	Stabilisation treatment (% of sludge produced)[B]				
		None	Anaerobic digestion	Aerobic digestion	Composting	Lime
UK	$1.1\ 10^6$	40	58 (96.6)	1 (1.7)	---------- 1 (1.7) ----------	
Ireland	$3.7\ 10^4$	62	---------- 38 (100) ----------		0	0
Belgium	$5.9\ 10^4$	19	---------- 81 (100) ----------		0	0
Netherlands	$3.2\ 10^5$	35	43 (66.2)	21 (32.3)	---------- 1 (1.5) ----------	
Denmark	$1.7\ 10^4$	4	50 (52.1)	40 (41.7)	1 (1.0)	5 (5.2)
Germany	$2.9\ 10^6$	25	65 (86.7)	7 (9.3)	3 (4.0)	0
France	$8.5\ 10^5$	41	49 (83.1)	7 (11.9)	3 (5.1)	0
Italy	$8.2\ 10^5$	25	---------- 75 (100) ----------		0	0
Norway	-	76	10 (41.7)	5 (20.8)	4 (16.7)	5 (20.8)
Sweden	-	10	50 (55.6)	18 (20.0)	7 (7.8)	15 (16.7)
Finland	-	58	20 (47.6)	13 (31.0)	1 (2.4)	8 (19.0)
Switzerland	-	19	77 (95.1)	2 (2.5)	2 (2.5)	0
Austria	-	30	---------- 50 (71.4) ----------		20 (28.6)	0
Average[C]	-	34.2	41.8 (63.5)	18.2 (27.7)	3.2 (4.9)	2.6 (3.9)

Notes:
A Data on sludge production obtained from reference 14, other data obtained from reference 2
B Values in brackets show the percentage of stabilised sludge treated by the process
C The average is not population weighted

Table 5 Options considered before 1999 to replace the sea disposal route

Region	Licensed disposal to sea, t y^{-1}	Location	Proposed main alternative disposal route
Anglian	$3.12\ 10^5$	Off Ipswich, Colchester, Tilbury	Agriculture. Considering incineration or landfill/reclamation for one works
Northumbrian	$6.59\ 10^5$	-	Incineration
North West	$1.97\ 10^6$	Liverpool bay	Incineration or landfill
Southern	$3.00\ 10^5$	-	Composting
South West	$1.38\ 10^5$	Off Plymouth, Exeter	Incineration but also looking at recycling
Thames	$4.50\ 10^6$	Thames estuary	Incineration
Welsh	$5.50\ 10^4$	Bristol channel	Agriculture, in the interim
Yorkshire	$1.40\ 10^5$	-	Incineration
Strathclyde	$2.50\ 10^6$	-	Incineration being considered
Lothian	$5.00\ 10^5$	-	Incineration
Northern Ireland	$3.20\ 10^5$	-	Agriculture and Incineration (incinerator operational in Belfast since early 1999)

Table 6 Typical sewage sludge application rates

Type of sludge	Rate of application	Type of crop
Liquid untreated	80 m^3 ha^{-1}	Hay
Liquid digested	60 m^3 ha^{-1}	Winter cereals, oilseed rape
Untreated dewatered	50 t ha^{-1} 150 t ha^{-1}	Cereals, grass re-seed Restored areas

Table 7 Uses of radionuclides considered in this study

Nuclide	Half-life	Typical uses
H-3	12.4 y	Light devices (GTLDs). Labelling of molecules and compounds in biology and chemistry. Elimination of charge, vacuum gauges, thickness gauges. Ground water tracer. Used in electron capture instruments used with chromatography.
C-14	5,730 y	Labelling of molecules and compounds in biology and chemistry. Tracer studies in agriculture *etc*
P-32	14.3 d	Used in medical therapy and for diagnostic purposes. Used for labelling nucleotides. Used in industry for labelling plasticising substances and dissolving reagents. Used in agriculture for studying the effectiveness of inorganic fertilisers and a wide range of other labelling and tracer applications.
S-35	87.5 d	Wide range of labelling applications, such as the study of amino acids, metabolism, chemicals such as surfactants and insecticides and pesticides.
Cr-51	27.7 d	Labelling of red blood corpuscles (e.g. glomerular filtration studies). In-situ irradiation of tumours (in the form of a wire). Used in industry to evaluate corrosion rates.
Ga-67	3.08 d	Treatment of tumours
Rb-83/ Rb-84	86.2 d 32.8 d	By products in the synthesis of Kr-81m, which is used for lung ventilation imaging and lung perfusion imaging.
Sr-89	50.5 d	Treatment of bone metastases.
Y-90	2.67 d	Treatment of arthritic conditions and malignant diseases.
Tc-99m	6.02 h	Used very widely in clinical biology as a tracer.
In-111	2.83 d	Medical diagnostic scans
I-123	13.2 h	Used widely in medical diagnosis, for thyroid and renal imaging and thyroid uptake.
I-125	60.1 d	Treatment of thyroid cancers, scintigraphy and the study of many tissue functions. Tracer for pests. Study of the penetration of water into soils and rocks.
I-131	8.04 d	Study of many tissue functions.
Tl-201	3.04 d	Myocardial scans, thyroid tumour imaging and parathyroid imaging.
Am-241	432 y	Sealed sources used in smoke detectors, unsealed sources may be used in research.

Table 8 Most common applications of nuclear medicine in the financial year 1989/90

Procedure	Radionuclide	Pharmaceutical	Estimated Annual no. of procedures	Mean administrated activity (MBq)
Bone scan	^{99m}Tc	Phosphate comps	141 200	545
Lung perfusion scan	^{99m}Tc	MAA	50 900	84
Kidney scan	^{99m}Tc	DTPA	19 400	196
Thyroid scan	^{99m}Tc	Pertechnate	18 800	85
Lung ventilation scan	^{81m}Kr	Gas	16 400	*
Kidney scan	^{99m}Tc	DMSA	14 900	80
Lung ventilation scan	^{99m}Tc	Aerosol	12 900	52
Dynamic cardiac scan	^{99m}Tc	Labelled red cells	12 000	722
Glomerular filtration rate measurement	^{51}Cr	EDTA	11 700	2.7
Lung ventilation scan	^{133}Xe	Gas	10 900	*
Myocardial perfusion scan (planar)	^{201}Tl	Thallous ion	9 200	74
Thyrotoxicosis therapy	^{131}I	Iodide	9 000	446
Kidney scan	^{99m}Tc	MAG3	8 500	87
Brain scan	^{99m}Tc	Pertechnate	7 600	530
Liver scan	^{99m}Tc	Colloid	6 500	87
Brain scan	^{99m}Tc	Gluconate	5 300	523
Vitamin B12 absorption	^{57}Co	Cyanocobalamin	4 700	0.03
Brain scan (tomography)	^{99m}Tc	HMPAO	3 500	559
Thyroid uptake	^{131}I	Iodide	3 300	1
Myocardial perfusion scan (tomography)	^{201}Tl	Thallous ion	3 100	78

* Data unreliable

Table 9 Administration of radionuclides in a major hospital in Northern Ireland

Radionuclide	Maximum usual activity, MBq	Description	Number carried out, 1995	Number carried out, 1996	Est. total activity administered, 1996 MBq
^{99m}Tc [A]	555	Bone scan	953	948	$5.26\ 10^5$
	74	Kidney scan	1023	1178	$8.72\ 10^4$
	460	Parathyroid	49	75	$3.45\ 10^4$
	74	Lung perfusion	357	406	$3.00\ 10^4$
	80	Thyroid scan	111	130	$1.04\ 10^4$
I-123	185	Adrenal glands	11	11	$2.04\ 10^3$
I-131	1200	Thyroid thyrotoxicosis	117	121	$1.45\ 10^5$
	3700	Ablation therapy	8	14	$5.18\ 10^4$
	74	Thyroid metastases	5	12	888
	0.74	Thyroid uptake	33	41	30.3
Ga-67	70	Tumour/lung/abscess imaging	43	34	$2.38\ 10^3$
Tl-201	92	Myocardium	69	57	$5.24\ 10^3$
In-111	185	Neuroendocrine tumours	41	42	$7.77\ 10^3$
	18.5	Labelled red blood cells	7	4	74
Kr-81m	55 MBq-mins	Lung ventilation	357	406	

Note:
A Five applications responsible for the highest discharges have been listed only

Table 10 Sewage treatment removal efficiencies for heavy metals

Element	Removal efficiency of stage (%)			
	Primary sedimentation	Activated sludge	Percolating filter	Average across works
Mercury		72		83
Copper	47	77	59	77 (49 - 86[A])
Cadmium	41	47		67 (16 - 80[A])
Nickel	49	41	47	56 (0 - 53[A])
Lead	82	53	70	64 (49 - 93[A])
Zinc	47	60	54	73 (41 - 77[A])
Iron		88	62	73 - 79 [A]
Sodium		7[A]	13[A]	
Cobalt		86.5[A]	79[A]	
Bromine		14[A]	6[A]	
Manganese		28		

Note:
All data from reference 24 with the exception of (A): Reference 2

Table 11 Distribution of tritium in sewage materials in Cambridge, 1987

Location of tritium	Estimated total activity in 1987, GBq	% of disposal in given material	% of activity in treatment only[A]
Total disposed to drain	502	100	
Total in supernatant	113.3	22.6	79.2
Total in suspended solids in supernatant	0.9	0.2	0.7
Total in all supernatant	**114.2**	**22.8**	**80.0**
Total in sludge cake	8.18	1.6	5.6
Total in methane	20.4	4.1	14.4
Total removed in treatment	**28.58**	**5.7**	**20.0**

Notes:
A The total activity estimated in treatment is thought to be much less than the total disposed to drains because the sampling programme was not continuous and may therefore miss peaks in activity due to periodic disposal practices.
B Taken from reference 24

Table 12 Sewage treatment removal for elements considered in the study

Element	Removal efficiency of process (%)			Notes
	Activated sludge	Percolating filter	All treatment	
Hydrogen	-	-	7[A]	Field studies: Cambridge
Carbon	-	-	4.4 -11.8[A]	Field studies: Cambridge; leaving large amounts of C-14 unaccounted for.
	-	-	25 - 75[A]	Implied removal efficiency
	-	-	30	Measured value, this study
Phosphorus	84	77	20 - 30	PO_4^{3+}
Sulphur	-	-	7.7[A]	Field studies: Cambridge
Chromium	-	-	38 - 88	Field studies (no chemical treatment)
Chromium	82[B]	58[B]	78[B]	
Chromium	-	-	38[A]	Field studies: Cambridge
Chromium	-	-	45 - 49	Field studies: Lund STW in Sweden
Gallium	-	-	~80	No data available Extrapolated from literature values for other metallic elements (conservative estimate)
Rubidium	-	-	45 - 80	No data available Lower removal efficiency extrapolated from Cs (similar chemical properties) upper limit from comparison with heavy metal behaviour.
Strontium	-	-	40 - 80	Lower limit from ref.2 upper limit from comparison with heavy metal behaviour
Yttrium	-	-	40 - 80	No data available Extrapolated from Sr
Technetium	-	-	0 - 10	Measured values, this study
Indium	-	-	~80	No data available Extrapolated from literature values for other metallic elements (conservative estimate)
Iodine	12.5	5	-	I^- (with carrier KI)
Iodine	-	-	0 - 44	Field studies (no chemical treatment)
Thallium	-	-	~80	No data available Extrapolated from literature values for other metallic elements (conservative estimate)
Americium	-	-	~80	No data available. Extrapolated from Pu values (similar chemical properties)

Notes:
All data from reference 2 with the exception of:
A Reference 24
B Reference 25

Table 13 **Best estimate removal efficiencies for primary and secondary treatment combined**

Element	Basis for estimate	Removal efficiency estimate (%)	
		Primary and activated sludge secondary treatment	Primary and percolating filter secondary treatment
Hydrogen	Full scale sewage plant	10	10
Carbon	Full scale sewage plant	30	30
Phosphorus	Full scale sewage plant	85	75
Sulphur	Full scale sewage plant	10	10
Chromium	Full scale sewage plant	80	60
Gallium	Extrapolated from behaviour of elements with similar chemical properties	80	80
Rubidium	Extrapolated from behaviour of elements with similar chemical properties	80	80
Strontium	Extrapolated from behaviour of elements with similar chemical properties	80	80
Yttrium	Extrapolated from behaviour of elements with similar chemical properties	80	80
Technetium	Measured data, full scale sewage plant	10	10
Indium	Extrapolated from behaviour of elements with similar chemical properties	80	80
Iodine	Primary & secondary treatment	10	10
Thallium	Extrapolated from behaviour of elements with similar chemical properties	80	80
Americium	Extrapolated from behaviour of elements with similar chemical properties	80	80

Note:
A Values have been rounded to the first ten percent

Table 14 Summary of input data for the GDC model

Characteristic	Parameter
Raw effluent	
Suspended solids load, %	0.05
Volumetric Flow rate, $m^3 s^{-1}$ ($m^3 d^{-1}$)	$6.95\ 10^{-4}$ (69.5)
Liquid effluent treatment times, h	
Primary settlement	4
Biological filtration	7
Final settlement	4
Total time	15
Solid sludge treatment times, h	
Primary settlement	4
Pasteurisation	2
Digestion	300
Storage	350
Total time	656
Treated sludge	
Solids content, %	5

Table 15 K_ds used in the Electrowatt study

Element	K_d value ($m^3 kg^{-1}$)
Hydrogen	$3.0\ 10^{-5}$
Carbon	$1.0\ 10^{-1}$
Cobalt	$1.0\ 10^{2}$
Iodine	$3.0\ 10^{-1}$

Table 16 Activity Concentrations in sludge Bq m^{-3}

Nuclide	Electrowatt	Smart
H-3	118.6	1600
C-14	380.1	800
I-125	783.6	900

Table 17 **Annual effective doses per MBq d^{-1} for disposal of sludge to farmland, from Electrowatt study and as predicted by SMART**

Radionuclide	Total annual effective dose (Sv/y)	
	Electrowatt	SMART
H-3	$1.5\ 10^{-9}$	$2.9\ 10^{-10}$
C-14	$4.6\ 10^{-8}$	$6.0\ 10^{-8}$
Co-57	$8.4\ 10^{-8}$	--
Co-58	$1.9\ 10^{-7}$	--
Co-60	$3.9\ 10^{-6}$	--
Zn-65	$2.2\ 10^{-5}$	--
I-125	$1.2\ 10^{-7}$	$1.3\ 10^{-8}$

Table 18 **Typical sewage treatment data in MOIRA and site specific data in SMART**

Stage	Average detention time (h)			Typical SS reduction (%)	
	MOIRA	SMART		MOIRA	SMART
		Beckton	Knostrop		
Primary treatment	1.5 - 2.5	2 -4	2 -4	60 - 70	70
Primary treatment (with chemical treatment)	1.5 - 2.5			70 - 90	
Primary + secondary treatment	4 - 6	12 - 316	6.5 - 10.5	80 - 95	96
Primary +secondary +tertiary treatment				>95	

Table 19 **Sludge treatments characteristics in MOIRA + SMART**

Stage	Typical solids concentrations (%)	Retention times (days)
Thickening	3 - 6	1
Anaerobic stabilisation	4 - 10	30 (20 - 90)
Aerobic stabilisation	4 - 10	15 (12 - 20)
Dewatering	12 - 50	>10

Table 20 Sewage treatment works in Greater London

Sewage Treatment Works	Population equivalent (p.e.)	Fraction of total served	Consented flow m3/d	Flow per person m3/d per p.e.
Beckton	$3.0\ 10^6$	31%	$2.7\ 10^6$ DWF	0.9
Mogden	$1.7\ 10^6$	17%	$4.2\ 10^5$ DWF	0.3
Crossness	$1.5\ 10^6$	16%	$9.8\ 10^5$ DWF	0.6
Long reach	$8.4\ 10^5$	9%	$1.7\ 10^5$ DWF	0.2
Deephams	$8.0\ 10^5$	8%	$2.0\ 10^5$ DWF	0.3
Maple Lodge	$5.0\ 10^5$	5%	$1.3\ 10^5$ DWF	0.3
Riverside	$4.3\ 10^5$	5%	$2.5\ 10^5$ DWF	0.6
Beddington	$3.2\ 10^5$	3%	$2.3\ 10^5$ DWF	0.7
Hogsmill	$2.0\ 10^5$	2%	$8.6\ 10^4$ Max	0.4
Worcester Park	$1.2\ 10^5$	1%	$7.1\ 10^4$ Max	0.6
Esher	$1.1\ 10^5$	1%	$3.5\ 10^4$ DWF	0.3
Blackbirds	$1.0\ 10^5$	1%	$1.2\ 10^5$ Max	1.2
Total	$9.6\ 10^6$		$5.4\ 10^6$	Mean = 0.5

Notes:
DWF Consented flow is in terms of Dry Weather Flow (DWF). Maximum flow rate is typically three times this value.
Max Indicates the consented flow is a maximum flow rate.

Table 21 Sludge treatment

Sewage Treatment Works	Sludge produced wet t y^{-1}	Solids fraction at disposal	Avg. treatment time, d	Avg. storage time, d	Treatment method	Disposal route	No. of workers
Beckton	2649900	0.06	2	0	Thickening	Incinerator	27
Mogden	1218852	0.23	20	0	Digestion, centrifuging	Land	26
Crossness	2119920	-	-	-	Thickening	Incinerator	27
Long reach	2104225	0.035	-	-	Digestion	Land	-
Deephams	655200	0.2	18	56	Digestion	Land	29
Maple Lodge	210000	0.205	22	14	Digestion	Land	15
Riverside	358430	-	-	-	To Beckton	Incinerator	-
Beddington	237250	0.24	24	0	Digestion, dewatering	Land	15
Hogsmill	219000	0.2	20	0	Digestion, dewatering	Land	11
Worcester Park	Closing 1999	-	-	-	-	-	-
Esher	54000	0.05	14	0	None	Land	-
Blackbirds	-	-			All to Maple Lodge	-	-

Note:
A Where information on current and future practices is available, the future practices have been presented if they will be operational in 1999

Table 22 Authorised discharges to Beckton, by nuclide

Radiation weighted rank	Nuclide	Monthly authorised limits (Bq)			Number specified	Ingestion Dose coefficient, Sv Bq^{-1}	Estimated activity conc, Bq m^{-3}
		Total	Mean	Maximum			
1	I-125	$2.0\ 10^{11}$	$6.0\ 10^{9}$	$9.5\ 10^{10}$	33	$1.5\ 10^{-8}$	$2.4\ 10^{3}$
2	I-131	$8.6\ 10^{10}$	$7.8\ 10^{9}$	$3.0\ 10^{10}$	11	$2.2\ 10^{-8}$	$1.0\ 10^{3}$
3	C-14	$1.3\ 10^{11}$	$4.7\ 10^{9}$	$1.0\ 10^{11}$	28	$5.8\ 10^{-10}$	$1.6\ 10^{3}$
4	P-32	$1.6\ 10^{10}$	$1.1\ 10^{9}$	$1.0\ 10^{10}$	15	$2.4\ 10^{-9}$	$2.1\ 10^{2}$
5	Rb-84	$4.0\ 10^{9}$	$4.0\ 10^{9}$	$4.0\ 10^{9}$	1	$2.8\ 10^{-9}$	$4.8\ 10^{1}$
6	Sr-89	$3.5\ 10^{9}$	$8.7\ 10^{8}$	$1.0\ 10^{9}$	4	$2.6\ 10^{-9}$	$4.2\ 10^{1}$
7	Tc-99m	$9.7\ 10^{11}$	$3.6\ 10^{10}$	$1.2\ 10^{11}$	27	$2.2\ 10^{-11}$	$1.2\ 10^{4}$
8	H-3	$2.5\ 10^{12}$	$2.3\ 10^{11}$	$2.5\ 10^{12}$	11	$1.8\ 10^{-11}$	$8.4\ 10^{4}$
9	S-35	$2.9\ 10^{10}$	$2.4\ 10^{9}$	$1.5\ 10^{10}$	12	$1.3\ 10^{-10}$	$3.5\ 10^{2}$
10	I-123	$1.7\ 10^{10}$	$1.8\ 10^{9}$	$1.0\ 10^{10}$	9	$2.1\ 10^{-10}$	$2.0\ 10^{2}$
11	Y-90	$7.4\ 10^{8}$	$7.4\ 10^{8}$	$7.4\ 10^{8}$	1	$2.7\ 10^{-9}$	$8.9\ 10^{0}$
12	Ga-67	$9.0\ 10^{9}$	$1.0\ 10^{9}$	$2.5\ 10^{9}$	9	$1.9\ 10^{-10}$	$1.1\ 10^{2}$
13	Tl-201	$1.8\ 10^{10}$	$2.0\ 10^{9}$	$1.2\ 10^{10}$	9	$9.5\ 10^{-11}$	$2.1\ 10^{2}$
14	In-111	$3.5\ 10^{9}$	$5.0\ 10^{8}$	$1.0\ 10^{9}$	7	$2.9\ 10^{-10}$	$4.2\ 10^{1}$
15	P-33	$1.5\ 10^{9}$	$7.5\ 10^{8}$	$1.0\ 10^{9}$	2	$2.4\ 10^{-10}$	$1.8\ 10^{1}$
16	Se-75	$8.1\ 10^{7}$	$2.7\ 10^{7}$	$4.0\ 10^{7}$	3	$2.6\ 10^{-9}$	$9.8\ 10^{-1}$
17	Cr-51	$2.8\ 10^{9}$	$3.1\ 10^{8}$	$1.0\ 10^{9}$	9	$3.8\ 10^{-11}$	$3.3\ 10^{1}$
18	Co-58	$5.0\ 10^{6}$	$5.0\ 10^{6}$	$5.0\ 10^{6}$	1	$7.4\ 10^{-10}$	$6.0\ 10^{-2}$
19	Co-57	$8.0\ 10^{6}$	$4.0\ 10^{6}$	$5.0\ 10^{6}$	2	$2.1\ 10^{-10}$	$9.6\ 10^{-2}$
20	Xe-133	$1.5\ 10^{8}$	$1.5\ 10^{8}$	$1.5\ 10^{8}$	1	-	$1.8\ 10^{0}$
-	Other β/γ	$1.6\ 10^{11}$	$3.4\ 10^{9}$	$5.0\ 10^{10}$	48	-	$2.0\ 10^{3}$
-	Other alpha	$4.0\ 10^{5}$	$1.0\ 10^{5}$	$4.0\ 10^{5}$	4	-	$4.8\ 10^{-3}$

Note:
A For ranking purposes, authorisations are specified as "H-3 and C-14" have been assumed to be C-14. Likewise, authorisations specified as "I-125 and I-131" have been assumed to be I-125.

Table 23 Summary of discharges to Beckton, by premises

Radiation weighted rank	Organisation	Monthly authorised limits (Bq of each nuclide)	Typical monthly disposals in 1997 (Bq of each nuclide)	Contribution to total indicative dose[A]	Authorisation permit number
1	St Bartholomew's Hospital, The Royal Hospitals NHS Trust	8 10^{10} Tc-99m, 9.5 10^{10} I-125/I-131, 9 10^{9} other β/γ	1.8 10^{8} H-3, 8.9 10^{6} C-14, 4.8 10^{7} P-32, 4.8 10^{7} S-35, 2.3 10^{8} Cr-51, 1.9 10^{4} o-57, 4.4 10^{8} Ga-67, 1.0 10^{6} Se-75, 3.8 10^{7} Sr-89, 1.9 10^{7} Y-90, 1.9 10^{10} Tc-99m, 6.3 10^{8} I-123, 2.5 10^{8} I-125, 3.2 10^{10} I-131, 1.8 10^{9} Tl-201, 6.2 10^{4} other β/γ	18.9 %	AU5556
2	Royal Postgraduate Medical School	5 10^{9} C-14/H-3, 1 10^{10} I-131, 5 10^{10} other β/γ	2.7 10^{7} H-3, 2.4 10^{5} C-14, 4.7 10^{5} P-32 & P-33, 1.4 10^{7} S-35, 1.3 10^{5} Cr-51, 2.3 10^{6} I-125, 5.0 10^{8} I-131	14.9 %	AY8743
3	The Middlesex Hospital, Bloomsbury and Islington HA	5 10^{10} Tc-99m, 3.7 10^{10} I-125/I-131, 2 10^{10} other β/γ	7.4 10^{10} Tc-99m, 8.0 10^{9} I-125/I-131, 6.7 10^{9} other β/γ	11.5 %	AA0230
4	Charing Cross Hospital, Hammersmith Hospitals NHS Trust	5 10^{10} Tc-99m, 1 10^{10} I-131, 2.5 10^{10} other β/γ	7.8 10^{7} H-3, 7.7 10^{7} P-32, 4.6 10^{10}, Tc-99m, 5.1 10^{7}, I-125, 7.4 10^{9} I-131	8.7 %	AY7569
5	Oldchurch Hospital, Havering Hospitals NHS Trust[B]	5 10^{8} P-32, 2.5 10^{9} Ga-67, 1 10^{9}Sr-89, 7.5 10^{10} Tc-99m, 1 10^{9} In-111, 2 10^{9} I-123, 1 10^{8} I-125, 3 10^{10} I-131, 1 10^{9} Tl-201, 2 10^{8} other β/γ	1.5 10^{7} P-32, 2.9 10^{7} Ga-67, 1.0 10^{8} Sr-89, 3.7 10^{10} Tc-99m, 8.3 10^{7} I-123, 4.3 10^{4} I-125, 8.6 10^{8} I-131, 1.2 10^{7} Tl-201, 1.2 10^{6} other β/γ	7.6 %	AT3671
6	University College Hospital, Bloomsbury and Islington HA	7.4 10^{8} P-32, 7.4 10^{8} Y-90, 3 10^{10} Tc-99m, 5 10^{8} I-111, 3.8 10^{9} Iodine isotopes, 2 10^{8} other β/γ	5.8 10^{7} H-3, 9.8 10^{7} Cr-51, 2.5 10^{5},C0-57, 1.3 10^{7} Ga-67, 9.3 10^{9} Tc-99m, 5.0 10^{6} In-111, 3.3 10^{6} I-123, 5.0 10^{5}, I-125	6.8 %	AA0256
7	St Mary's Hospital Medical School, Imperial College School of Med.	2.5 10^{10} Tc-99m, 1.7 10^{10} I-131, 3.5 10^{9} other β/γ	1.5 10^{10} Tc-99m, 2.0 10^{9} I-131, 6.7 10^{8} other β/γ	5.1 %	AZ4735
8	Royal London Hospital, The Royal Hospitals NHS Trust	2.4 10^{9} C-14/H-3, 8 10^{10} Tc-99m, 1 10^{10} I-123, 2 10^{10} I-125/I-131, 5 10^{9} other β/γ	6.1 10^{7} H-3, 1.5 10^{6} C-14, 4.4 10^{7} P-32, 7.0 10^{7} S-35, 9.8 10^{7} Cr-51, 1.3 10^{5} ,Co-57, 2.4 10^{7} Ga-67, 1.8 10^{4} Se-75, 2.5 10^{7} Sr-89, 1.5 10^{7} Y-90, 2.3 10^{10} , Tc-99m, 9.2 10^{6} In-111, 1.5 10^{7} I-123, 1.0 10^{7} I-125, 1.4 10^{9}, I-131, 5.8 10^{7}, Tl-201, 3.0 10^{8} other β/γ	4.8 %	AO5280
	Royal Free Hospital, Royal Free Hampstead NHS Trust	6 10^{10} Tc-99m, 1.5 10^{10} I-131, 4 10^{9} other β/γ	4.9 10^{10} Tc-99m, 1.1 10^{10} I-131, 2.5 10^{9} other β/γ	4.7 %	AV8011
10	Institute of Neurology	1.5 10^{9} H-3/C-14, 5 10^{8} I-125/I-131, 7.5 10^{9} other β/γ	3.2 10^{5} H-3, 1.7 10^{7} P-32 & P-33, 3.8 10^{6} S-35, 1.9 10^{6} I-125, 2.2 10^{4} other β/γ	2.0 %	AR3488

Radiation weighted rank	Organisation	Monthly authorised limits (Bq of each nuclide)	Typical monthly disposals in 1997 (Bq of each nuclide)	Contribution to total indicative dose[A]	Authorisation permit number
11	Imperial Cancer Research Fund	4.1 10^9 H-3/C-14, 1 10^{10} P-32, 1.5 10^{10} S-35, 2.5 10^9 I-125/I-131, 4.5 10^9 other β/γ	2.0 10^8 H-3, 2.0 10^6 C-14, 7.8 10^7 P-32 & P-33, 2.5 10^8 S-35, 2.5 10^8 Cr-51, 1.8 10^9, Y-90, 4.2 10^7 I-125	1.9 %	AZ4093
12	Whittington Hospital NHS Trust	1.5 10^{10} Tc-99m, 1 10^9 I-131, 6 10^9 other β/γ	3.8 10^9 Tc-99m, 8.9 10^7 I-131, 4.8 10^7 other β/γ	1.7 %	AW7673
13	Royal Brompton, Sydney St.	2.4 10^9 Ga-67, 1.2 10^{11} Tc-99m, 1.2 10^{10} Tl-201, 5.9 10^9 other β/γ	3.6 10^8 Ga-67, 4.2 10^{10} Tc-99m, 4.6 10^9 Tl-201, 2.8 10^8 other β/γ	1.4 %	AI4344
14	Imperial College of Science and Technology	2 10^{10} H-3, 4 10^5 alpha, 5 10^9 other β/γ	5.2 10^8 H-3, 1.3 10^7 C-14, 3.6 10^8 P-32, 2.8 10^8 S-35, 7.9 10^6 Cr-51, 5.8 10^7, I-125, 4.5 10^6 I-131, 2.9 10^8 other β/γ	1.2 %	AX6940
15	Institute of Cancer Research, Fulham Rd	1 10^{11} H-3/C-14, 1 10^9 P-32, 3.4 10^9 S-35, 5 10^8 other β/γ	3.7 10^7 H-3, 3.0 10^8 P-32, 2.7 10^8 S-35, 6.0 10^7 other β/γ	0.9 %	AQ3920
16	EISAI London Research Laboratories Ltd	4 10^8 H-3/C-14, 5 10^7 I-125, 3 10^9 other β/γ	1.8 10^7 C-14, 1.8 10^4 I-125, 1.4 10^8 other β/γ	0.8 %	AP8284
17	Institute of Child Health	2 10^9 H-3/C-14, 7 10^8 I-125/I-131, 2.3 10^9 other β/γ	2.7 10^7 C-14, 2.2 10^7 I-125, 5.1 10^8 other β/γ	0.7 %	AH9871
18	University College London	1.5 10^9 C-14, 2.1 10^9 S-35, 9 10^8 I-125, 2.1 10^9 other β/γ	4.9 10^8 C-14, 1.8 10^8 P-32, 1.5 10^7 I-125, 3.0 10^7 other β/γ	0.7 %	BA0765
19	Whipps Cross Hospital, Forest Healthcare Trust	3.5 10^{10} Tc-99m, 6 10^7 I-125, 2.2 10^9 other β/γ	2.1 10^{10} Tc-99m, 3.6 10^5 I-125, 5.5 10^8 other β/γ	0.6 %	AK2831
20	Medical Research Council, Imperial College School of Medicine	1 10^9 H-3, 1 10^8 C-14, 2 10^9 P-32, 1 10^9 P-33, 2.5 10^9 S-35, 7 10^8 Cr-51, 4 10^9 Rb-84, 1 10^9 I-125, 2 10^8 I-131, 5 10^8 other β/γ	7.9 10^7 H-3, 1.3 10^5 C-14, 2.1 10^8 P-32, 2.0 10^8 S-35, 5.5 10^7 Cr-51, 1.8 10^8 , Rb-84, 1.5 10^8 I-125, 1.9 10^8 other β/γ	0.5 %	AZ5286

Notes

A Indicative dose from the organisation ÷ total indicative dose from all discharges to Beckton STW.

B Not in Beckton catchment area but sludges are treated at Beckton.

Table 24 **Individual doses to workers in large pipes, Beckton catchment. Doses assessed using authorised and typical discharges**

	Authorised Discharge	Total Dose	Typical Discharge	Total Dose
Nuclide	Bq/d	Sv/y	Bq/d	Sv/y
H-3	$7.0\ 10^{8}$	$2.88\ 10^{-14}$	$4.14\ 10^{7}$	$1.17\ 10^{-15}$
C-14	$4.0\ 10^{9}$	$5.69\ 10^{-12}$	$1.85\ 10^{7}$	$2.63\ 10^{-14}$
P-32	$4.7\ 10^{8}$	$1.92\ 10^{-8}$	$4.41\ 10^{7}$	$1.81\ 10^{-9}$
S-35	$7.7\ 10^{8}$	$1.11\ 10^{-12}$	$3.79\ 10^{7}$	$5.48\ 10^{-14}$
Cr-51	$2.3\ 10^{7}$	$2.84\ 10^{-9}$	$2.47\ 10^{7}$	$3.00\ 10^{-9}$
Ga-67	$1.6\ 10^{8}$	$1.64\ 10^{-8}$	$2.89\ 10^{7}$	$2.92\ 10^{-9}$
Rb-84	$1.6\ 10^{8}$	$5.63\ 10^{-7}$	$5.83\ 10^{6}$	$2.02\ 10^{-8}$
Sr-89	$3.3\ 10^{7}$	$1.07\ 10^{-11}$	$5.42\ 10^{6}$	$1.74\ 10^{-12}$
Y-90	$2.5\ 10^{7}$	$3.58\ 10^{-10}$	$6.80\ 10^{7}$	$9.83\ 10^{-10}$
Tc-99m	$2.1\ 10^{10}$	$4.03\ 10^{-8}$	$1.13\ 10^{10}$	$2.21\ 10^{-8}$
In-111	$5.0\ 10^{7}$	$1.28\ 10^{-7}$	$4.74\ 10^{5}$	$1.21\ 10^{-9}$
I-123	$4.0\ 10^{8}$	$3.20\ 10^{-9}$	$2.46\ 10^{7}$	$1.97\ 10^{-10}$
I-125	$5.3\ 10^{9}$	$5.64\ 10^{-6}$	$2.85\ 10^{8}$	$3.02\ 10^{-7}$
I-131	$8.1\ 10^{9}$	$5.71\ 10^{-6}$	$2.25\ 10^{9}$	$1.58\ 10^{-6}$
Tl-201	$4.3\ 10^{8}$	$2.01\ 10^{-8}$	$2.14\ 10^{8}$	$9.96\ 10^{-9}$
Am-241	$1.3\ 10^{4}$	$2.07\ 10^{-11}$	0	0
	Totals	$1.21\ 10^{-5}$		$1.94\ 10^{-6}$

Table 25 **Individual doses to workers fixing small pipes, Beckton catchment**

	Discharge	External Dose	Inhalation Dose	Ingestion Dose	Total
Nuclide	Bq	Sv/event	Sv/event	Sv/event	Sv/event
Tc-99m	$8.00\ 10^{8}$	$1.63\ 10^{-5}$	$1.42\ 10^{-12}$	$1.75\ 10^{-11}$	$1.63\ 10^{-5}$
I-131 (includes 'other' nuclides)	$1.09\ 10^{9}$	$9.13\ 10^{-5}$	$9.34\ 10^{-10}$	$2.94\ 10^{-8}$	$9.14\ 10^{-5}$
	Totals	$1.08\ 10^{-4}$	$9.35\ 10^{-10}$	$2.94\ 10^{-8}$	$1.08\ 10^{-4}$

Notes:
See Chapter 7 for the assumptions made about the radionuclide discharge.

Table 26 **Summary of doses and disposals for Beckton in Central London, showing contributions from dominant radionuclides. Doses from authorised and typical disposals are shown**

		Individual doses (Sv/y)						
		Total	C-14	P-32	Rb-84	Tc-99m	I-125	I-131
	half-life		5730 y	14.3 d	32.8 d	6.02 h	60.1 d	8.04 d
Hospital workers blocked pipes.		**$1.1\ 10^{-4}$**	- (note4)	-	-	$1.6\ 10^{-5}$	-	$9.1\ 10^{-5}$
Sewer workers - large pipes	Auth.	**$1.2\ 10^{-5}$**	-	-	$5.6\ 10^{-7}$	-	$5.6\ 10^{-6}$	$5.7\ 10^{-6}$
	Typ.	**$1.9\ 10^{-6}$**	-	-	$2.0\ 10^{-8}$	$2.2\ 10^{-8}$	$3.0\ 10^{-7}$	$1.6\ 10^{-6}$
Sewer workers, at STW inc. incin.	Auth.	**$2.1\ 10^{-4}$**	-	-	$1.6\ 10^{-4}$	-	-	$4.2\ 10^{-5}$
	Typ.	**$1.9\ 10^{-5}$**	-	-	$5.7\ 10^{-6}$	-	-	$1.2\ 10^{-5}$
Sewer workers - at incinerator	Auth.	**$2.0\ 10^{-4}$**	-	-	$1.6\ 10^{-4}$	-	-	$4.1\ 10^{-5}$
	Typ.	**$1.8\ 10^{-5}$**	-	-	$5.6\ 10^{-6}$	-	-	$1.1\ 10^{-5}$
Sludge press workers	Auth.	**$2.1\ 10^{-4}$**	-	-	$4.9\ 10^{-5}$	-	-	$1.5\ 10^{-4}$
	Typ.	**$4.5\ 10^{-5}$**	-	-	$2.0\ 10^{-6}$	-	-	$4.2\ 10^{-5}$
Public - effluent (note5)	Auth.	**$3.1\ 10^{-5}$**	$2.8\ 10^{-5}$	$2.0\ 10^{-6}$	-	-	-	-
	Typ.	**$6.1\ 10^{-7}$**	$1.3\ 10^{-7}$	$1.9\ 10^{-7}$	-	-	-	$2.5\ 10^{-7}$
Public - incinerator release to atmosphere	Auth.	**$5.2\ 10^{-7}$**	-	-	-	-	$3.3\ 10^{-7}$	$1.7\ 10^{-7}$
	Typ.	**$6.6\ 10^{-8}$**	-	-	-	-	$1.8\ 10^{-8}$	$4.8\ 10^{-8}$
Farmer - incinerator release to atmosphere	Auth.	**$1.7\ 10^{-6}$**	-	-	-	-	$1.1\ 10^{-6}$	$5.6\ 10^{-7}$
	Typ.	**$2.2\ 10^{-7}$**	-	-	-	-	$6.1\ 10^{-8}$	$1.5\ 10^{-7}$
Public - incinerator ash to landfill	Auth.	**$2.2\ 10^{-6}$**	$2.2\ 10^{-6}$	-	-	-	-	-
	Typ.	**$1.0\ 10^{-8}$**	$1.0\ 10^{-8}$	-	-	-	-	-
Discharges Bq/month	Auth.		$1.2\ 10^{11}$ (note3)	$1.4\ 10^{10}$	$4.8\ 10^{9}$	$6.3\ 10^{11}$	$1.6\ 10^{11}$ (note1)	$2.4\ 10^{11}$ (note2)
	Typ.		$5.6\ 10^{8}$	$1.3\ 10^{9}$	$1.7\ 10^{8}$	$3.4\ 10^{11}$	$8.5\ 10^{9}$ (note1)	$6.8\ 10^{10}$ (note2)

Notes
1 Authorisations including 'iodines' or I-131/I-125 are assumed to be I-125 discharges.
2 Authorisations including 'other beta/gamma' are assumed to be I-131 discharges
3 Authorisations including H-3/C-14 are assumed to be C-14 discharges
4 Hyphen indicates zero or very small doses
5 The individual doses after 50 years of disposal have been given. In all cases except ash to landfill the doses are the same for 1 and 50 years. For ash to landfill the doses are 50 times lower after only 1 year of disposal.

Table 27 **Collective effective doses to the UK population truncated at 500 y for 1 and 50 years of disposal from Beckton sewage treatment works. Doses assessed using authorised and typical discharges**

Disposal options	Nuclide	Collective dose (manSv)[1,2]			
		Authorised disposals		Typical disposals	
		1 year disposal	50 years of disposal	1 year disposal	50 years of disposal
Disposal of effluent to Thames estuary	C-14	$1.0\ 10^{-1}$	$5.3\ 10^{0}$	$4.7\ 10^{-4}$	$2.5\ 10^{-2}$
	I-131			$8.9\ 10^{-4}$	$4.4\ 10^{-2}$
	Total	**$1.1\ 10^{-1}$**	**$5.6\ 10^{0}$**	**$1.5\ 10^{-3}$**	**$7.7\ 10^{-2}$**
Disposal of sludge by incineration					
Ash to landfill	C-14	$9.2\ 10^{-3}$	$4.6\ 10^{-1}$	$4.2\ 10^{-5}$	$2.1\ 10^{-3}$
	Total	**$9.2\ 10^{-3}$**	**$4.6\ 10^{-1}$**	**$4.2\ 10^{-5}$**	**$2.1\ 10^{-3}$**
Atmospheric release	C-14	$1.2\ 10^{-1}$	$6.2\ 10^{0}$		
	I-125	$4.1\ 10^{-1}$	$2.0\ 10^{1}$	$2.2\ 10^{-2}$	$1.1\ 10^{0}$
	I-131	$2.2\ 10^{-1}$	$1.1\ 10^{1}$	$6.2\ 10^{-2}$	$3.1\ 10^{0}$
	Total	**$7.6\ 10^{-1}$**	**$3.8\ 10^{1}$**	**$8.4\ 10^{-2}$**	**$4.2\ 10^{0}$**

Notes:
1 Radionuclides considered are the same as table 24.
2 Doses from dominant radionuclides and total doses are given

Table 28 Beckton August 1998 sampling

Results in mBq g^{-1}

	Raw Sewage	(Filtered)	Primary Sludge		Activated Sludge	Filtered Outfall	Incinerator	
	10:00 to 15:00 11/8/98	15:00 to 20:00 11/8/98	14:00 11/8/98	20:00 11/8/98	09:45 12/8/98	9:10 12/8/98	Cake 10:30 12/8/98	Ash 10:30 12/8/98
			@ 1.45% dry	@ 2.4% dry	@ 0.65% dry		@ 28.3% dry	
Natural radionuclides								
^{7}Be	< 2	< 1	< 3	2.1 ± 0.8	< 1	< 1	29 ± 2	966 ± 58
^{40}K	< 3	2 ± 1	< 5	< 4	< 2	1.9 ± 0.6	195 ± 11	431 ± 22
^{75}Se	< 0.4	< 0.2	< 0.4	< 0.4	< 0.2	< 0.2	< 0.8	4 ± 1
^{214}Bi	< 0.6	0.4 ± 0.2	< 0.6	0.5 ± 0.3	< 0.2	< 0.3	0.8 ± 0.2	47 ± 2
^{226}Ra	< 6.8	< 4	< 8.81	< 8	< 4.4	< 4	27 ± 11	317 ± 44
^{228}Ac	< 0.8	< 0.6	< 1	< 1	< 0.4	< 0.5	4 ± 2	76 ± 27
^{235}U	< 2	< 2	< 3	< 2	< 1	< 2	1.6 ± 0.7	20 ± 3
Fallout radionuclides								
^{137}Cs	< 0.3	< 0.2	0.13 ± 0.06	< 0.2	< 0.1	< 0.1	0.3 ± 0.1	5.9 ± 0.4
Study radionuclides								
^{3}H	22 ± 4	29 ± 5	7.5 ± 0.8	11.8 ± 1.2	7.5 ± 0.7	9.5 ± 3.8	136 ± 12	199 ± 21
^{14}C	0.22 ± 0.09	0.2 ± 0.1	1.6 ± 0.9	5 ± 1	1.2 ± 0.4	0.14 ± 0.08	50 ± 11	< 3
^{32}P	11.1 ± 4	191 ± 16	59 ± 28	59.3 ± 12	8 ± 2	< 15.0	155 ± 57	Sample lost
^{35}S	< 0.1	0.2 ± 0.1	1.6 ± 0.1	2.9 ± 0.1	1.3 ± 0.1	0.17 ± 0.06		
^{51}Cr	< 2	< 1	< 3	< 2.5	< 1.2	< 1	10 ± 2	250 ± 17
^{67}Ga	< 2	< 1	< 0.2	< 2	< 0.9	< 0.8	5.6 ± 0.9	70 ± 6
^{83}Rb	< 0.4	< 0.3	< 0.5	< 0.4	< 0.2	< 0.2	< 0.5	< 1
^{84}Rb	< 0.3	< 0.3	< 0.4	< 0.3	< 0.2	< 0.2	< 0.4	< 0.8
^{99m}Tc	0.35 ± 0.14	4.9 ± 0.3	2.5 ± 0.3	2.0 ± 0.2	1.6 ± 0.2	0.4 ± 0.1	Half life too short.	
^{111}In	< 0.3	< 0.2	< 0.4	< 0.4	< 0.1	< 0.2	1.0 ± 0.3	6.5 ± 0.7
^{123}I	< 0.6	< 0.7	< 0.9	< 1	< 0.6	< 0.3	< 25.0	< 34
^{125}I	< 0.1	< 0.06	< 0.03	< 0.5	0.13 ± 0.01	< 0.05	0.33 ± 0.04	Sample lost
^{131}I	1.1 ± 0.4	0.35 ± 0.02	18 ± 1	32 ± 2	43 ± 3	0.52 ± 0.06	417 ± 21	608 ± 30

Table 29 Beckton January 1999 sampling

Results in mBq g^{-1}

	Primary Tank		Secondary Tank		Final Effluent			Incinerator		Incinerator	
	Solids	Liquids	Solids	Liquids	11:30 - 15:30 27/1/99	15:30 - 19:30 27/1/99	19:30 27/1/99 - 00:30 28/1/99	Sludge	Ash	Scrubber venturi (M2004)	Scrubber (M2002)
	10:45 27/1/99 @ 51.1% dry		16:00 27/1/99 @ 1.89% dry					12:00 28/1/99 @ 21.9% dry		12:00 28/1/99	
					Natural radionuclides						
^{7}Be	11 ± 2	< 2	2 ± 1	< 2	< 1	< 1	< 1	27 ± 6	727 ± 90	< 1	< 1
^{40}K	< 5	< 4	< 3	< 2	< 2	< 2	< 3	21 ± 8	564 ± 61	< 1.9	< 2
^{75}Se	< 0.6	< 0.3	< 0.5	< 0.2	< 0.2	< 0.2	< 0.2	< 2	8 ± 1	< 0.2	< 0.2
^{214}Bi	3.3 ± 0.7	2.8 ± 0.6	< 0.5	0.3 ± 0.2	0.4 ± 0.3	< 0.3	< 0.3	1.8 ± 0.9	51 ± 6	< 0.3	< 0.3
^{226}Ra	< 13	< 0.6	< 11	< 4	< 3	< 4	< 3	< 48	317 ± 44	< 5.7	< 3
^{228}Ac	1.2 ± 1.0	< 0.9	< 1.0	< 0.5	< 0.5	< 0.5	< 0.5	3 ± 2	106 ± 38	< 0.4	< 0.5
^{235}U	< 3	< 2	< 3	< 1	< 1	< 0.9	< 0.9	< 13	< 51	< 1.6	< 0.9
					Fallout radionuclides						
^{137}Cs	< 0.3	< 0.3	< 0.3	< 0.1	< 0.1	< 0.1	< 0	< 0.7	6 ± 1	< 0.1	< 0.1
					Study radionuclides						
^{3}H	24 ± 4	< 5	< 1	< 5		5 ± 4		32 ± 5	< 5	66 ± 6	44 ± 5
^{14}C	26 ± 8	< 0	0 ± 0	< 0		<0.1		92 ± 18	32 ± 8	< 0.2	< 0
^{32}P	9 ± 2	1.3 ± 0.4	0 ± 0	< 1		<0.1		15 ± 6	lost	< 0.2	< 0
^{35}S	16 ± 1	2.7 ± 0.5	0.9 ± 0.1	1.6 ± 0.4		1.2 ± 0.4		31 ± 2	42 ± 3	245 ± 15	3.9 ± 0.4
^{51}Cr	< 4	< 2	< 3	< 1	< 1	< 1	< 1	< 13	48 ± 10	< 2	< 1
^{67}Ga	< 2	< 1	< 0.2	< 0.8	< 0.7	< 0.9	< 0.8	< 9	73 ± 7	< 1	< 2
^{83}Rb	< 0.5	< 0.4	< 0.05	< 0.2	< 0.2	< 0.2	< 0.2	< 1	< 1	< 0.2	< 0.2
^{84}Rb	< 0.3	< 0.3	< 0.03	< 0.2	< 0.2	< 0.2	< 0.2	< 1	< 1	< 0.2	< 0.2
^{99m}Tc	< 0.9	< 0.5	< 4	0.4 ± 0.2	< 0.5	< 4	2 ± 1	< 20	Half life too short.	< 6.4	Half life too short.
^{111}In	< 0.4	< 0.2	< 0.4	< 0.1	< 0.1	< 0.2	< 0.1	< 2	< 3	< 0.2	< 0.3
^{123}I	< 0.7	< 0.3	< 1	< 0.2	< 0.2	< 0.6	< 0.5	< 6	< 359	< 1.2	< 0.3
^{125}I	2 ± 3	< 0.0 3	0.12 ± 0.02	< 0	<.02			lost	10 ± 3	2.0 ± 0.3	2.6 ± 0.6
^{131}I	49 ± 6	2 ± 0.35	55 ± 7	0.3 ± 0.2	0.6 ± 0.2	0.4 ± 0.1	0.5 ± 0.2	326 ± 37	723 ± 40	69 ± 8	0.7 ± 0.2

Table 30 Volatilised fractions in the Beckton incinerator

Radionuclide	Estimated Volatilised Fraction		Notes
	August	January	
Naturals			
^{7}Be	-1[1]	-0.3	
^{40}K	0.9	-0.3[1]	
^{226}Ra	0.3	-	Very high uncertainties
^{228}Ac	-0.3[1]	-0.6[1]	Very high uncertainties
^{235}U	0.3	-	Very high uncertainties
Fallout			
^{137}Cs	- 0.1	-	High uncertainties
Discharged			
^{14}C	1		
^{51}Cr	-0.6	-	High uncertainties
^{67}Ga	0.2	-	
^{111}In	0.6	-	Very high uncertainties
^{131}I	0.9	0.9	

[1]For these isotopes more activity was measured in the ash than would be inferred from measurements in the feedstock.

- denotes that measured concentrations were below detection limits, and so no estimate of the proportion volatilised could be made.

Table 31 Ranking of all disposers to Knostrop and the radionuclides disposed

Radiation weighted rank	Establishment	Radionuclide	Monthly authorised limits (Bq)	Typical monthly disposals in 1997 (Bq)	Authorisation permit numbers
1	Cookridge Hospital	Tc-99m, I-123/I-125/I-131 other*	$4\ 10^{10}$ $2.8\ 10^{10}$ $2\ 10^{9}$	$1.25\ 10^{10}$ Tc-99m, $1.9\ 10^{8}$ I-123, $1.25\ 10^{10}$ I-131, $6.9\ 10^{6}$ P-32, $6.7\ 10^{7}$ Cr-51, $2.0\ 10^{8}$ Sr-89	AJ3824
2	General Infirmary, United Leeds Teaching Hospitals, NHS Trust	Tc-99m, I-123/I-125/I-131 other*	$2\ 10^{11}$ $1.5\ 10^{10}$ $1\ 10^{10}$	$3.25\ 10^{10}$ Tc-99m, $2\ 10^{9}$ Iodines, $1.4\ 10^{9}$ other βγ	AO2647
3	St James University Hospital	Tc-99m, other*	$2\ 10^{11}$ $2\ 10^{10}$	$1\ 10^{11}$ Tc-99m, $9.2\ 10^{8}$ other βγ	AS4265
4	University of Leeds, Woodhouse Lane	H-3 and C-14 I-125/I-131, other* alphas	$8\ 10^{10}$ $2\ 10^{9}$ $1.2\ 10^{10}$ $1\ 10^{7}$	$3.7\ 10^{9}$ H-3, $2.7\ 10^{7}$ C-14, $8.3\ 10^{7}$ I-125, $2.4\ 10^{8}$ P-32, $5.8\ 10^{7}$ S-35	BB1147
5	Leeds Metropolitan University, Calvery St.	H-3 and C-14 I-125/I-131, other*	$3\ 10^{7}$ $1.9\ 10^{8}$ $1.9\ 10^{8}$	Closed for refurbishment Actual discharges in 1997 = 0	AQ7739
6	NRPB, Hospital Lane, Cookridge	H-3 other* alphas	$1\ 10^{9}$ $1\ 10^{6}$ $5\ 10^{5}$	$2.9\ 10^{5}$ H-3, $3.3\ 10^{3}$ other βγ	AV4679
Totals		H-3 H-3 + C-14 Tc-99m Iodines Alphas Others*	$1\ 10^{9}$ $8\ 10^{10}$ $4.4\ 10^{11}$ $4.5\ 10^{10}$ $1.1\ 10^{7}$ $4.4\ 10^{10}$	See Table D28 in Appendix D	

* Others specifically exclude alphas
Cookridge hospital others are Cr-51, P-32 and Sr-89.
University of Leeds others are mainly P-32 and S-35 but P-33, Cr-51, Co-57, Rb-89, Ca-45, Fe-55 and Ni-63 are also disposed.
NRPB others are C-14, Cd-109, Fe-55 and Ni-63. Alpha emitting radionuclides are generally Po-210, Po208 and Am-241.

Table 32 **Individual doses to workers in large sewage pipes, Leeds**
Doses assessed using authorised and typical discharges

Nuclide	Authorised Discharge Bq/d	Total Dose Sv/y	Typical Discharge Bq/d	Total Dose Sv/y
H-3	$3.3\ 10^{7}$ (note1)	$1.88\ 10^{-14}$	$1.23\ 10^{8}$	$6.93\ 10^{-14}$
C-14	$2.7\ 10^{9}$	$5.26\ 10^{-11}$	$8.90\ 10^{5}$	$1.73\ 10^{-14}$
P-32	-(note2)	-	$8.33\ 10^{6}$	$3.43\ 10^{-10}$
S-35	-	-	$1.93\ 10^{6}$	$3.81\ 10^{-14}$
Cr-51	-	-	$2.23\ 10^{6}$	$2.72\ 10^{-10}$
Sr-89	-	-	$6.67\ 10^{6}$	$2.64\ 10^{-12}$
Tc-99m	$1.5\ 10^{10}$	$2.92\ 10^{-8}$	$5.00\ 10^{9}$	$9.73\ 10^{-9}$
I-123	-(note3)	-	$6.67\ 10^{6}$	$5.34\ 10^{-11}$
I-125	$1.5\ 10^{9}$	$1.59\ 10^{-6}$	$6.94\ 10^{7}$	$7.35\ 10^{-8}$
I-131	$1.5\ 10^{9}$	$1.05\ 10^{-6}$	$5.10\ 10^{8}$	$3.58\ 10^{-7}$
Am-241	$3.7\ 10^{5}$	$6.07\ 10^{-10}$	-	-
	Totals	$2.67\ 10^{-6}$		$4.42\ 10^{-7}$

Notes:
1 H-3 can be discharged under a combined C-14/H-3 authorisation, for the 'authorised' model this was conservatively assumed to be all C-14.
2 P-32, S-35, Cr-51and Sr-89 are disposed under an 'other βγ ' authorisation, within the 'authorised' model all of the 'other βγ ' authorisation was assumed to be I-131.
3 I-123 is authorised under a general 'iodines' authorisation, within the 'authorised' model 'iodines' were considered to be all I-125.

Table 33 **Individual worker doses arising from a blockage in a small sewer pipe, Leeds**

Nuclide	Discharge Bq	External Dose Sv/event	Inhalation Dose Sv/event	Ingestion Dose Sv/event	Total Sv/event
Tc-99m	$4.00\ 10^{8}$	$8.13\ 10^{-6}$	$7.12\ 10^{-13}$	$8.74\ 10^{-12}$	$8.13\ 10^{-6}$
I-131 (includes 'other' nuclides)	$1.02\ 10^{9}$	$8.55\ 10^{-5}$	$8.74\ 10^{-10}$	$2.75\ 10^{-8}$	$8.55\ 10^{-5}$
	Totals	$9.36\ 10^{-5}$	$8.75\ 10^{-10}$	$2.75\ 10^{-8}$	$9.36\ 10^{-5}$

Table 34 **Summary of doses and disposals in Leeds, showing contributions of dominant radionuclides**
Doses from authorised and typical disposals are shown

		Individual doses (Sv/y)					
		Total	C-14	P-32	Tc-99m	I-125	I-131
	half-life		5730 y	14.3 d	6.02 h	60.1 d	8.04 d
Hospital workers blocked pipes.		**$9.4\ 10^{-5}$**	- (note4)	-	$1.8\ 10^{-6}$	-	$8.6\ 10^{-5}$
Sewer workers - large pipes	Auth.	**$2.7\ 10^{-6}$**	-	-	-	$1.6\ 10^{-6}$	$1.1\ 10^{-6}$
	Typ.	**$4.4\ 10^{-7}$**	-	-	-	$7.4\ 10^{-8}$	$3.6\ 10^{-7}$
Sewer workers, at STW inc. incin.	Auth.	**$7.8\ 10^{-5}$**	-	-	$3.8\ 10^{-6}$	-	$6.7\ 10^{-5}$
	Typ.	**$2.5\ 10^{-5}$**	-	-	-	-	$2.3\ 10^{-5}$
Sewer workers - at incinerator	Auth.	**$7.3\ 10^{-5}$**	-	-	$2.7\ 10^{-6}$	$1.5\ 10^{-6}$	$6.4\ 10^{-5}$
	Typ.	**$2.3\ 10^{-5}$**	-	-	$9.1\ 10^{-7}$	-	$2.2\ 10^{-5}$
Sludge press workers	Auth.	**$2.4\ 10^{-4}$**	-	-	-	-	$2.2\ 10^{-4}$
	Typ.	**$7.9\ 10^{-5}$**	-	-	-	-	$7.5\ 10^{-5}$
Sewer workers, at STW sludge stabilisation (note6)	Auth.	**$1.6\ 10^{-5}$**	-	-	-	-	$1.4\ 10^{-5}$
	Typ.	**$5.3\ 10^{-6}$**	-	-	$5.5\ 10^{-7}$	-	$4.7\ 10^{-6}$
Public - effluent (note5)	Auth.	**$1.8\ 10^{-4}$**	$1.2\ 10^{-4}$	-	-	$2.5\ 10^{-5}$	$3.6\ 10^{-5}$
	Typ.	**$1.9\ 10^{-5}$**	$3.9\ 10^{-8}$	$5.5\ 10^{-6}$	-	$1.2\ 10^{-6}$	$1.2\ 10^{-5}$
Public - incinerator release to atmosphere	Auth.	**$2.1\ 10^{-7}$**	$1.3\ 10^{-8}$	-	-	$1.3\ 10^{-7}$	$6.5\ 10^{-8}$
	Typ.	**$2.8\ 10^{-8}$**	-	-	-	$5.8\ 10^{-9}$	$2.2\ 10^{-8}$
Farmer - incinerator release to atmosphere	Auth.	**$2.2\ 10^{-6}$**	-	-	-	$1.5\ 10^{-6}$	$7.0\ 10^{-7}$
	Typ.	**$3.1\ 10^{-7}$**	-	-	-	$6.7\ 10^{-8}$	$2.4\ 10^{-7}$
Public - incinerator ash to landfill	Auth.	**$1.5\ 10^{-6}$**	$1.5\ 10^{-6}$	-	-	-	-
	Typ.	**$6.3\ 10^{-10}$**	$4.9\ 10^{-10}$	-	-	-	-
Public - sludge to landfill (note7)	Auth.	**$1.5\ 10^{-4}$**	$1.5\ 10^{-4}$	-	-	-	-
	Typ.	**$6.3\ 10^{-8}$**	$4.9\ 10^{-8}$	-	-	-	-
Discharges Bq/month	Auth.		$8.1\ 10^{10}$ (note3)	- (note6)	$4.4\ 10^{11}$	$4.5\ 10^{10}$ (note1)	$4.5\ 10^{10}$ (note2)
	Typ.		$2.7\ 10^{7}$	$2.5\ 10^{8}$	$1.5\ 10^{11}$	$2.1\ 10^{9}$ (note1)	$1.5\ 10^{10}$ (note2)

Notes
1 Authorisations including 'iodines' or I-131/I-125 are assumed to be I-125 discharges.
2 Authorisations including 'other beta/gamma' are assumed to be I-131 discharges
3 Authorisations including H-3/C-14 are assumed to be C-14 discharges
4 Hyphen indicates zero or very small doses
5 The individual doses after 50 years of disposal have been given. In all cases except ash to landfill the disposals are the same for 1 and 50 years. For ash to landfill the doses are 50 times lower after only 1 year of disposal.
6 P-32, S-35, Cr-51and Sr-89 are disposed under an 'other βγ ' authorisation, within the model all of the 'other βγ ' authorisation was assumed to be I-131.
7 Considers 1 year of disposal to landfill, in fact this disposal option only occurs occasionally during incinerator shutdown.

Table 35 **Collective, effective doses to the UK population truncated at 500 y for 1 and 50 years of disposal from Knostrop sewage treatment works. Doses assessed using authorised and typical discharges**

Disposal options	Nuclide	Collective dose (manSv)[1,2]			
		Authorised disposals		typical disposals	
		1 year disposal	50 years of disposal	1 year disposal	50 years of disposal
Disposal of effluent to River Aire	I-125	$1.3\ 10^{0}$	$6.9\ 10^{1}$	$6.2\ 10^{-2}$	$3.2\ 10^{0}$
	I-131	$1.9\ 10^{0}$	$9.4\ 10^{1}$	$6.4\ 10^{-1}$	$3.2\ 10^{1}$
	Total	**$3.3\ 10^{0}$**	**$1.7\ 10^{2}$**	**$7.0\ 10^{-1}$**	**$3.5\ 10^{1}$**
Disposal of sludge by incineration					
Ash to landfill	C-14	$6.2\ 10^{-3}$	$3.1\ 10^{-1}$	$2.0\ 10^{-6}$	$1.0\ 10^{-4}$
	Total	**$6.2\ 10^{-3}$**	**$3.1\ 10^{-1}$**	**$2.5\ 10^{-6}$**	**$1.3\ 10^{-4}$**
Atmospheric release	C-14	$9.0\ 10^{-2}$	$4.5\ 10^{0}$	$3.0\ 10^{-5}$	$1.5\ 10^{-3}$
	I-125	$2.0\ 10^{-1}$	$1.0\ 10^{1}$	$9.3\ 10^{-3}$	$4.6\ 10^{-1}$
	I-131	$9.7\ 10^{-2}$	$4.9\ 10^{0}$	$3.3\ 10^{-2}$	$1.7\ 10^{0}$
	Total	**$3.9\ 10^{-1}$**	**$1.9\ 10^{1}$**	**$4.2\ 10^{-2}$**	**$2.1\ 10^{0}$**
Disposal of sludge to landfill[3]	C-14	$6.2\ 10^{-1}$	$3.1\ 10^{1}$	$2.0\ 10^{-4}$	$1.0\ 10^{-2}$
	Total	**$6.2\ 10^{-1}$**	**$3.1\ 10^{1}$**	**$2.5\ 10^{-4}$**	**$1.3\ 10^{-2}$**

Notes:
1 Radionuclides considered are the same as table 25.
2 Doses from dominant radionuclides and total doses are given
3 Considers 1 year of disposals to landfill, in fact this disposal option only occurs occasionally during incinerator shutdown.

Table 36 Knostrop September 1998 sampling

Results in mBq g^{-1}

	Primary Tank	Secondary Tank		Final Effluent			Incinerator	
	Liquids	Liquids	Solids	16:00 15/9/98 to 08:30 16/9/98	12:15 16/9/98	12:15 to 16:00 16/9/98	Sludge	Ash
	11:45 15/9/98	16:15 15/9/98					11:30 16/9/98	15:00 16/9/98
		As measured[1]					@ 4.8% Dry	
Natural radionuclides								
^{7}Be	< 0.8	< 0.6	< 15	< 0.9	< 1.0	< 0.5	< 3	353 ± 47
^{40}K	< 3.30	< 2	< 78	< 4	< 1	< 2	< 11	222 ± 60
^{214}Bi	< 0.38	< 0.2	< 8	< 3	4 ± 1	0.8 ± 0.3	< 1	< 25
^{226}Ra	< 3.67	< 2	< 65	< 5	< 5	3 ± 2	< 0.1	87 ± 22
^{228}Ac	< 1.70	< 0.3	< 9	< 0.6	< 0.6	< 0.3	< 1	35 ± 6
^{235}U	< 0.91	< 0.7	< 19	< 1	< 1	< 0.6	< 2	7 ± 5
Fallout radionuclides								
^{137}Cs	< 0.12	< 0.08	< 2	< 0.2	< 0.09	< 0.07	< 0.3	10 ± 2
Study radionuclides								
^{3}H	14.80 ± 4.1	29.20 ± 4.80			Bulked to give 7 ± 4		5.19 ± 1.83	154 ± 18
^{14}C	< 0.28	0.19 ± 0.16	< 9		Bulked to give <0.3		1.0 ± 0.1	< 8
^{32}P	< 0.67	< 0.6	57 ± 22		Bulked to give <1		< 0.1	lost
^{35}S	< 0.34	< 0.2	< 30		Bulked to give <0.3		< 0.06	< 2
^{51}Cr	< 1.02	< 0.5	< 0	< 0.7	< 1	< 0.4	< 1	31 ± 9
^{67}Ga	< 1.70	0.6 ± 0.3	< 5	1.5 ± 0.8	< 3	0.6 ± 0.3	2.1 ± 1.6	16 ± 3
^{83}Rb	< 0.15	< 0.07	< 4	< 0.2	< 0.2	< 0.06	< 0.5	< 1.0
^{84}Rb	< 0.12	< 0.1	< 2	< 0.2	< 0.08	< 0.09	< 2	< 0.8
^{99m}Tc	0.85 ± 0.3	1.3 ± 0.4	189 ± 24	3.6 ± 1.0	< 0.3	< 0.1	0.8 ± 0.5	3 ± 2
^{111}In	< 0.10	< 0.06	< 2	< 0.2	< 0.1	< 0.03	< 0.1	< 0.6
^{123}I	< 0.08	< 0.08	< 3	< 0.3	< 0.1	< 0.06	< 0.2	< 0.5
^{125}I	< 0.05	< 0.05	< 14		Bulked to give <0.01		< 0.1	< 0.01
^{131}I	25.11 ± 3.0	28 ± 3	880 ± 103	31 ± 4	21 ± 3	18 ± 2	7 ± 1	7 ± 2

Table 37 Knostrop November 1998 sampling

Results in mBq g^{-1}

	Primary Tank	Secondary Tank		Final Effluent		Incinerator	
	Liquids	Liquids	Solids	12:00 to 16:00 19/11/98	16:00 19/11/98 to 10:00 20/11/98	Sludge	Ash
	11:30 18/11/98	16:00 18/11/98				10:30 19/11/98 @ 9.7% Dry	16:00 19/11/98
		As measured[1]					
Natural radionuclides							
^{7}Be	< 0.8	< 0.7	110 ± 79	< 0.5	< 1	23 ± 4	560 ± 74
^{40}K	< 5	< 5	703 ± 162	< 30	2 ± 1	12 ± 7	457 ± 66
^{214}Bi	< 1.00	< 2	85 ± 31	< 2	< 0.3	1.8 ± 0.8	31 ± 7
^{226}Ra	< 4.00	< 5	309 ± 170	< 6	< 4	< 8	89 ± 32
^{228}Ac	< 2.00	< 2	< 100	< 2	< 0.4	2 ± 1	53 ± 10
^{235}U	< 1.13	< 0.8	< 57	< 0.6	< 1	< 1	< 6
Fallout radionuclides							
^{137}Cs	< 0.09	< 0.08	< 5	< 0.1	< 0.1	0.7 ± 0.3	20 ± 4
Study radionuclides							
^{3}H	6 ± 4	< 5	34 ± 16	Bulked to 5 ± 4		7 ± 4	23 ± 18
^{14}C	< 0.1	< 0.1	< 10	Bulked to <0.1		16 ± 8	< 7
^{32}P	0.3 ± 0.1	< 0.1	n/a	Bulked to 0.3 ± 0.2		< 4	Lost
^{35}S	< 0.2	0.24 ± 0.20	n/a	Bulked to < 0.2		4.0 ± 0.2	12 ± 1
^{51}Cr	< 0.8	< 0.3	< 100	0.6 ± 0.2	< 1	2 ± 2	47 ± 19
^{67}Ga	< 0.1	< 1	< 50	< 1	< 0.8	< 1	15 ± 4
^{83}Rb	< 0.2	< 0.1	< 6	< 0.1	< 0.2	< 0.1	< 2
^{84}Rb	< 0.1	< 0.1	< 20	< 0.1	< 0.2	< 0.2	< 1
^{99m}Tc	3.4 ± 0.5	5.7 ± 0.7	1071 ± 126	< 0.6	3 ± 1	< 0.2	5 ± 1
^{111}In	< 0.06	< 0.1	< 3	< 0.1	< 0.2	< 0.1	< 0.8
^{123}I	< 0.12	< 0.1	< 3	< 0.1	< 0.4	< 0.1	< 0.7
^{125}I	< 0.09	< 0.2		Bulked to <0.06		< 0.2	< 0.06
^{131}I	5.16 ± 0.7	8 ± 1	2363 ± 276	3.4 ± 0.5	3.1 ± 0.4	13 ± 2	3.0 ± 2.0

[1] 12.5 l of sludge as collected gave 12.6 g of paste as measured

Table 38 Volatilised fractions in the Knostrop incinerator

Radionuclide	Estimated Volatilised Fraction		Notes
	September	November	
Naturals			
^{7}Be	-	0.2	High uncertainties
^{40}K	-	-0.3[1]	
^{228}Ac	-	0.2	Very high uncertainties
Fallout			
^{137}Cs	-	0.03	High uncertainties
Discharged			
^{3}H			
^{14}C	0.9	1	
^{32}P	Sample lost	0.9	
^{51}Cr	-	0.3	Very high uncertainties
^{67}Ga	0.9	-	Very high uncertainties
^{131}I	0.98	0.99	

Notes
[1] For these isotopes more activity was measured in the ash than in the equivalent amount of feedstock.
- denotes that measured concentrations were below detection limits, and so no estimate of the proportion volatilised could be made.

Table 39 Volatilised fractions in incinerators

		Radionuclides								
		H-3	C-14	P-32	S-35	Cr-51	Ga-67	Tc-99m	In-111	Iodines
Beckton sampling	Aug	n/a(1)	1	-	-	-0.6(3)	0.2	-	0.61	0.9
	Jan	n/a	-	-	-	-	-	-	-	0.9
Knostrop sampling	Sept	n/a	0.9	-	-	-	-	-	-	0.98
	Nov	n/a	1	0.9	-	0.3 (3)	0.9(3)	-	-	0.99
Ref 40		0.99	0.99	0	0.1	0	-	0.95	-	0.95
Ref 41		-	0.97	0.05	0.4	0.1	-	-	-	0.99
Ref 42		-	-	-	-	-	-	-	-	0.98
SMART2 (best estimates)		0.99	0.99	0.01	0.1	0.01	0.23	0.95	0.61	0.95

Notes:
(1) n/a results not available
(2) n/m s not measured in sludge
(3) very high uncertainties

Table 40 Summary of Beckton and Knostrop sewage works

Feature	Beckton sewage works	Knostrop high level sewage works
Incoming effluent $m^3 d^{-1}$	$1\ 10^6$ (Average)	$7.9\ 10^4$ (DWF)
Average suspended solids mg 1^{-1}	269	444
Dry solids production t d^{-1}	269	35
Average sludge production $m^3 d^{-1}$	16 048 (@1.5% solid)	702 (@5% solids, derived)
Population equivalent	$3\ 10^6$	$4.2\ 10^5$
Secondary treatment	Aerobic digestion	Biological filtration
Incinerator cake production ($m^3 d^{-1}$)	840 (Derived @32% solids)	159 (Derived @22% solids)
Ash production rate (t d^{-1})	54 (Derived @20% solids)	11 (Derived @30% solids)
Disposal options		
Sludge	Incineration	Incineration
Effluent	Tidal Thames	River Aire
Ash	Landfill	Landfill
Stack height (m)	50	40
Nearest resident (km)	1	1
Nearest farm (km)	13	5
Radionuclides authorised for discharge[1]	I-125, I-131, C-14, P-32, Tc-99m, Sr-89, Tc-99m, H-3, Rb-84, S-35, Tl-201, I-123, Ga-67, In-111, P-33, Cr-51	Tc-99m, I-123/I-125/I-131, H-3/C-14
Dose to incinerator worker (1997 discharges) μSv y^{-1}	19 12 (excluding Rb-84)	25
Dose to public from liquid effluent (1997 discharges) μSv y^{-1}	0.6	19
Dose to public from incinerator (1997 discharges) μSv y^{-1}	0.2 (farmer) <0.1 (resident)	0.4 (farmer) <0.1 (resident)

Note:
1 Authorisations in excess of 10^9 Bq per month

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Figure 1: Use of water and production of sewage sludge, 1983-1993

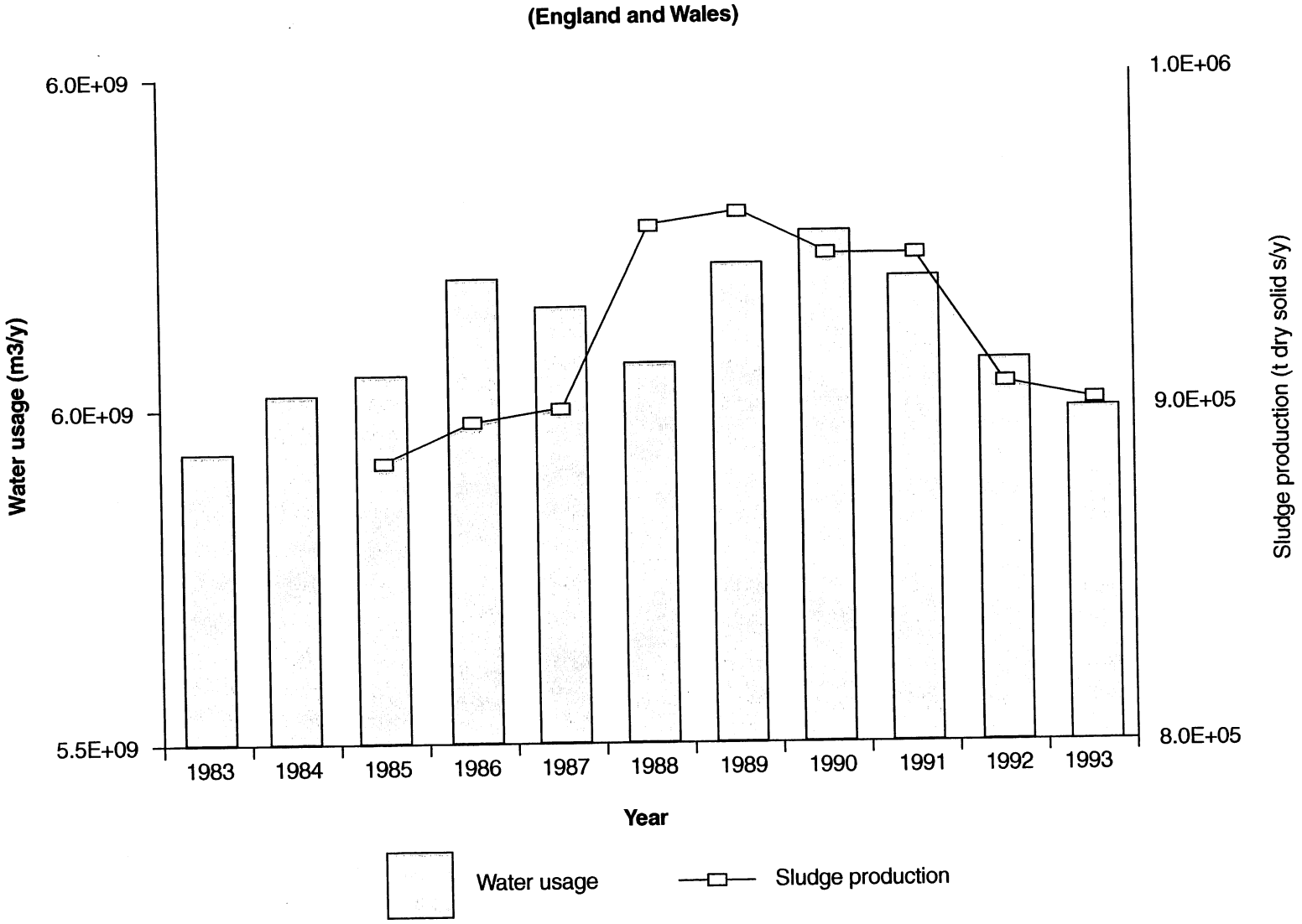


Figure 2: Use of water and production of sewage sludge, 1983 - 1993

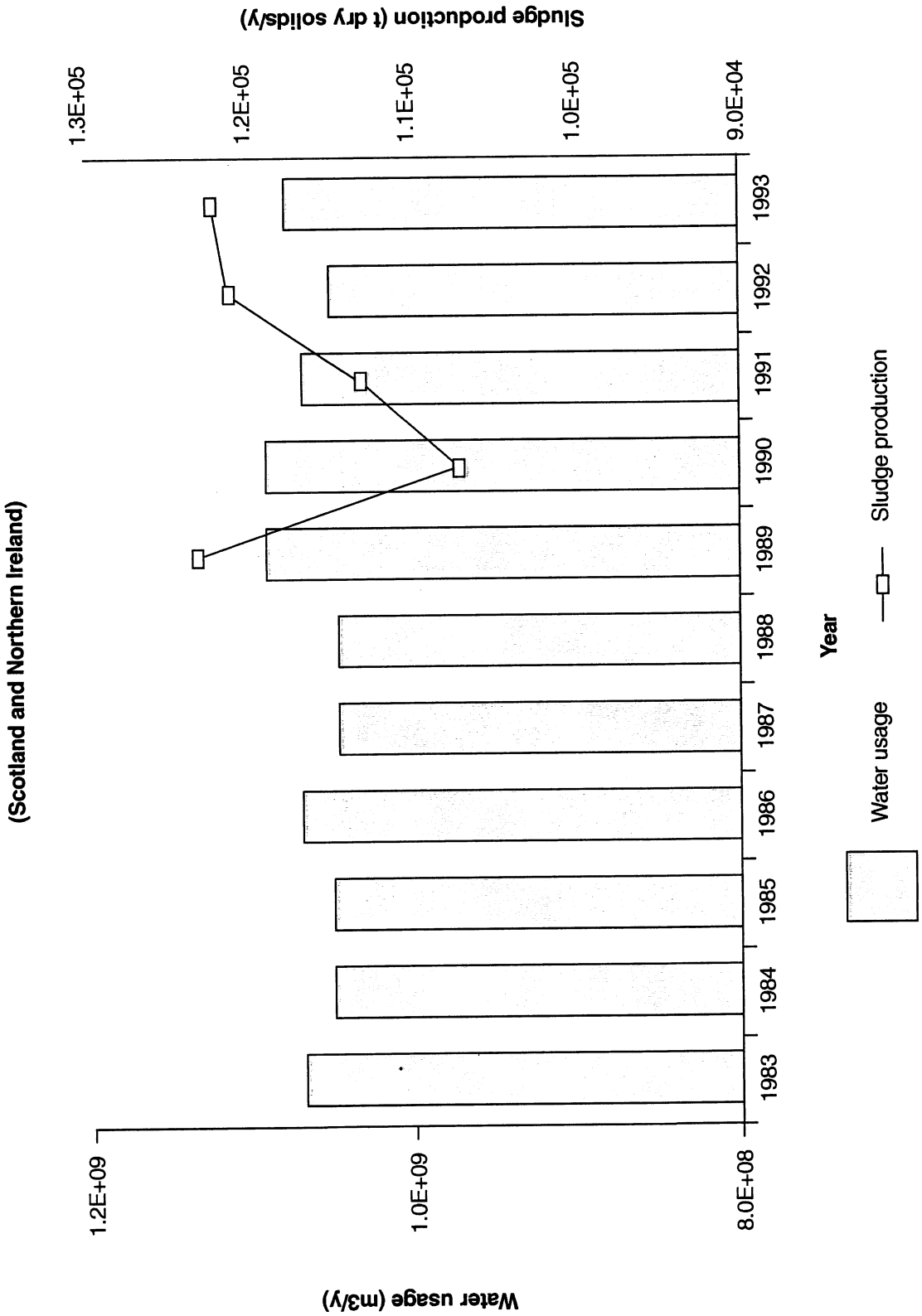


Figure 3: Process Diagram for Assessing Radionuclide Transport in Waste Water Treatment

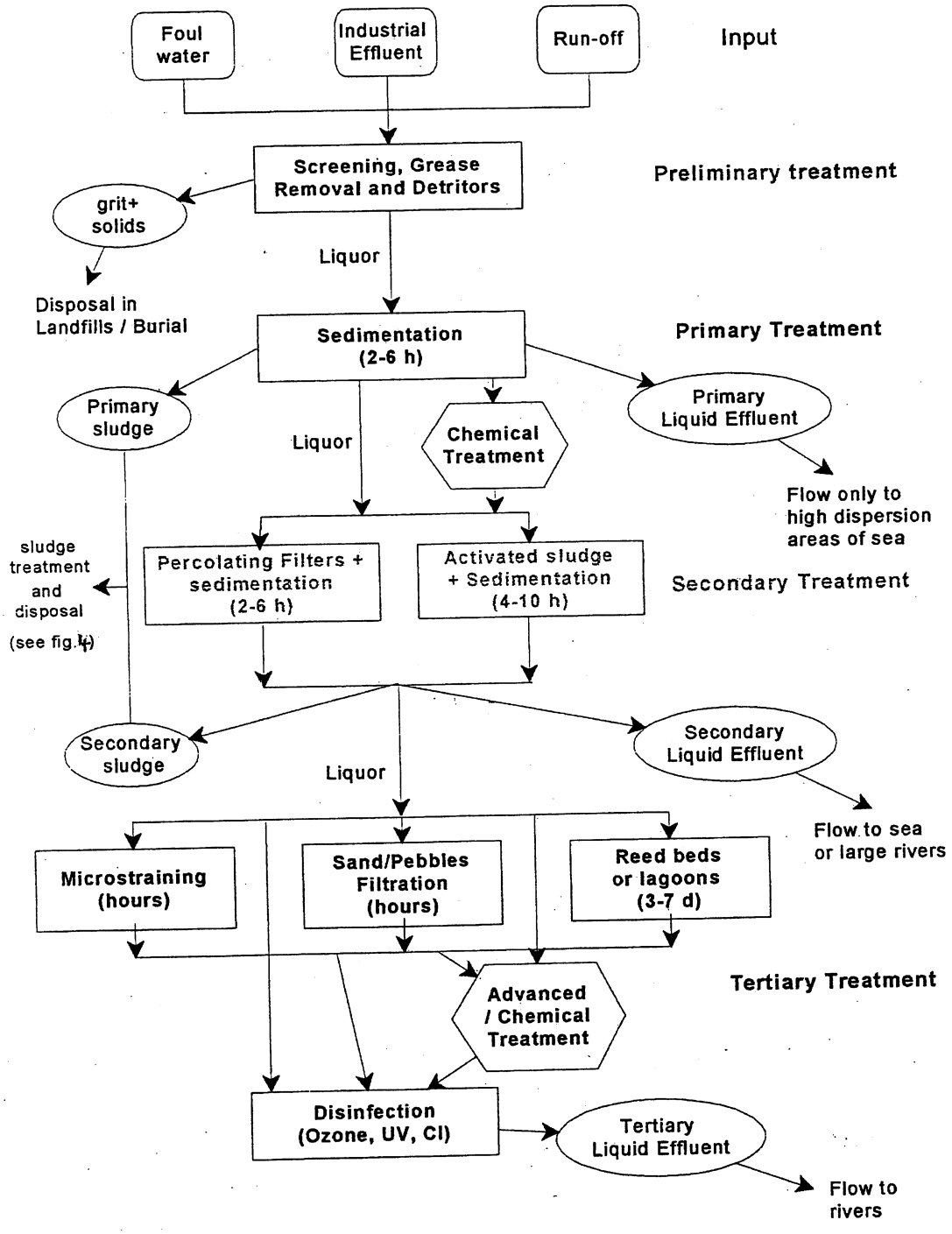
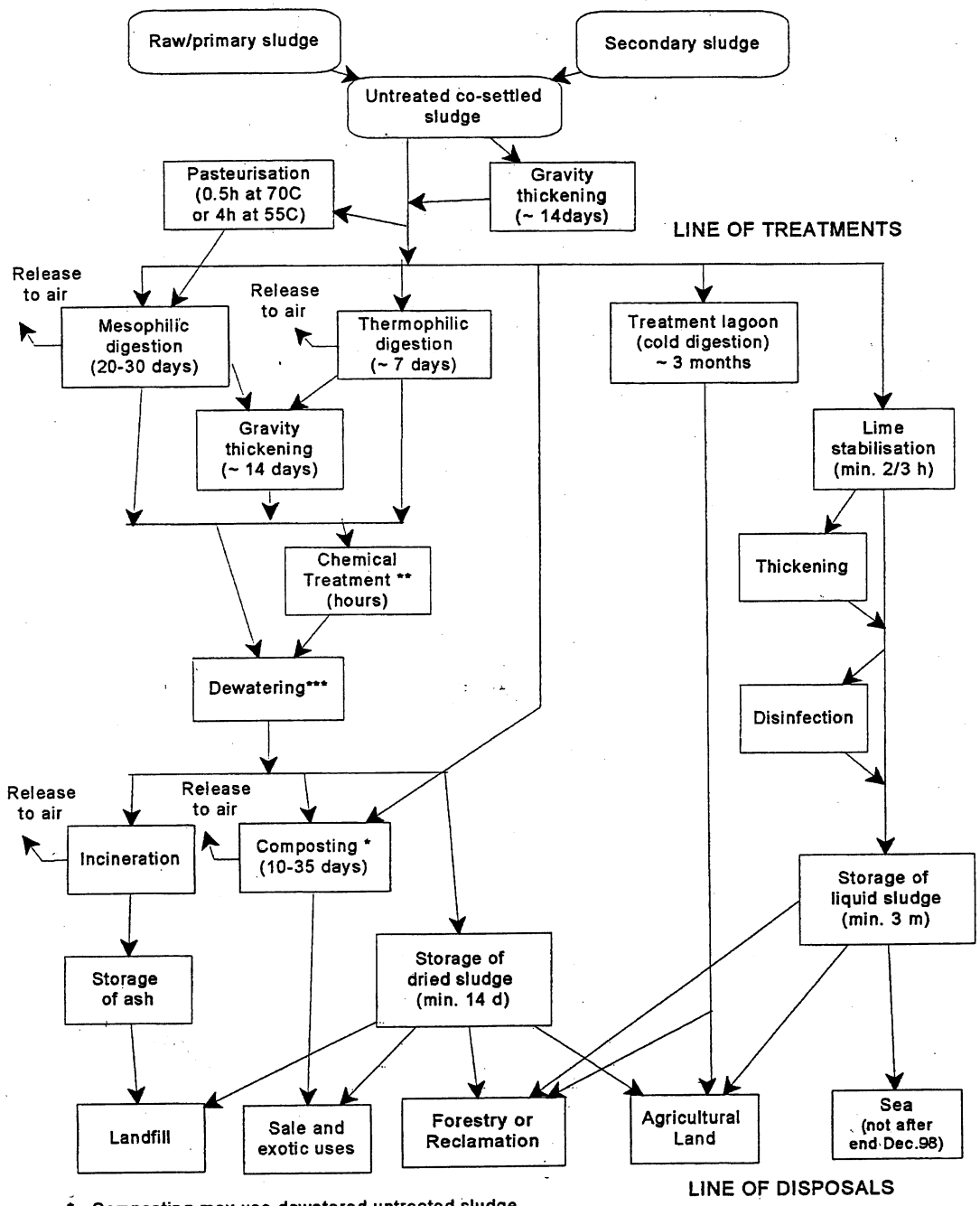


Figure 4: Process Diagram for Assessing Radionuclide Transport in Sludge Treatment



* Composting may use dewatered untreated sludge.
** Chemical treatments include lime, aluminium chlorohydrate and poly-electrolyte.
*** Dewatering techniques include filter press, vacuum filter, belt press and drying beds.

Figure 5: Disposal routes for sludge

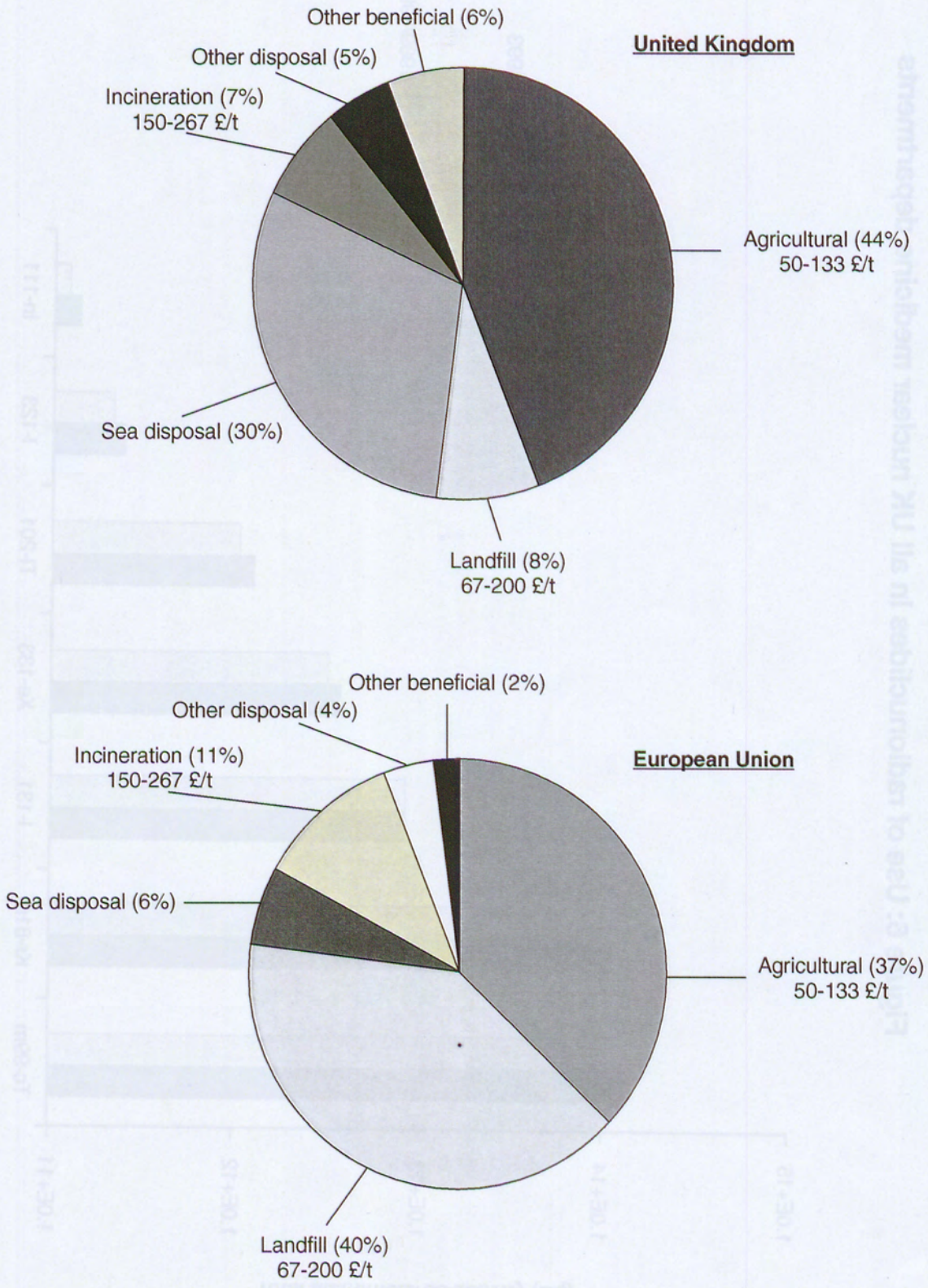


Figure 6: Use of radionuclides in all UK nuclear medicine departments

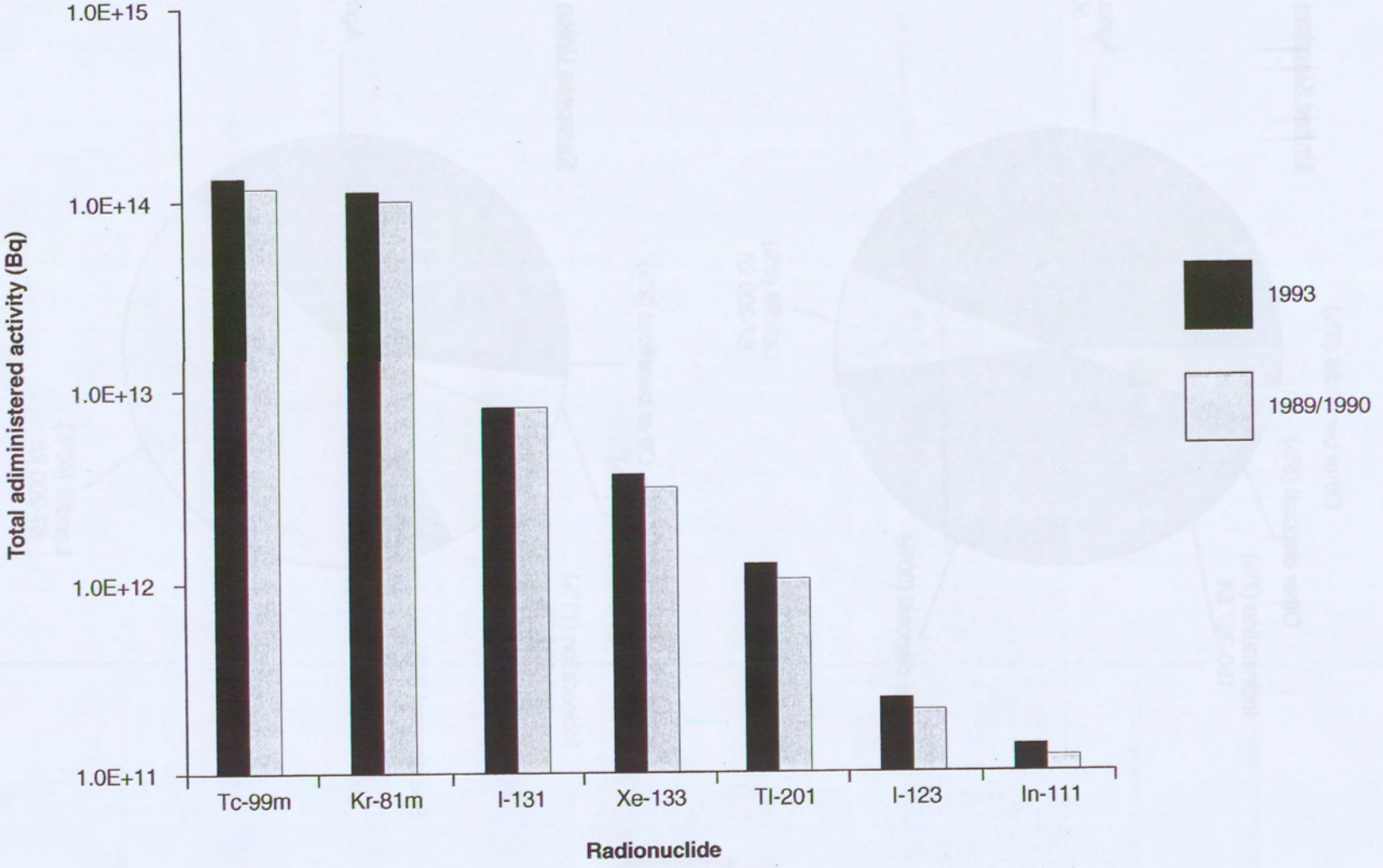


Figure 7:
Effluent and Sludge Treatments Diagram for Sewer Model

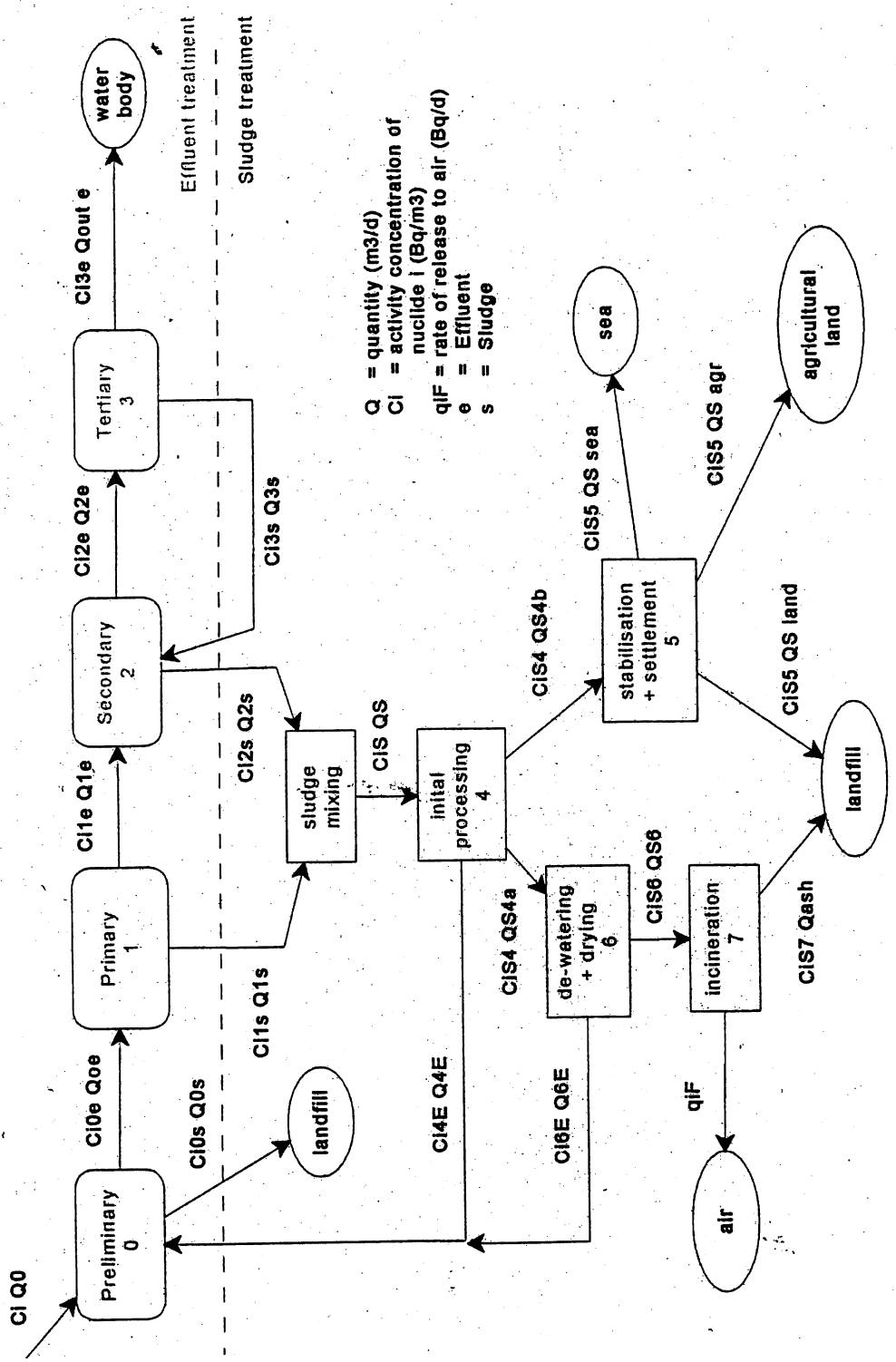


Figure 8 Sewage catchment map for Greater London

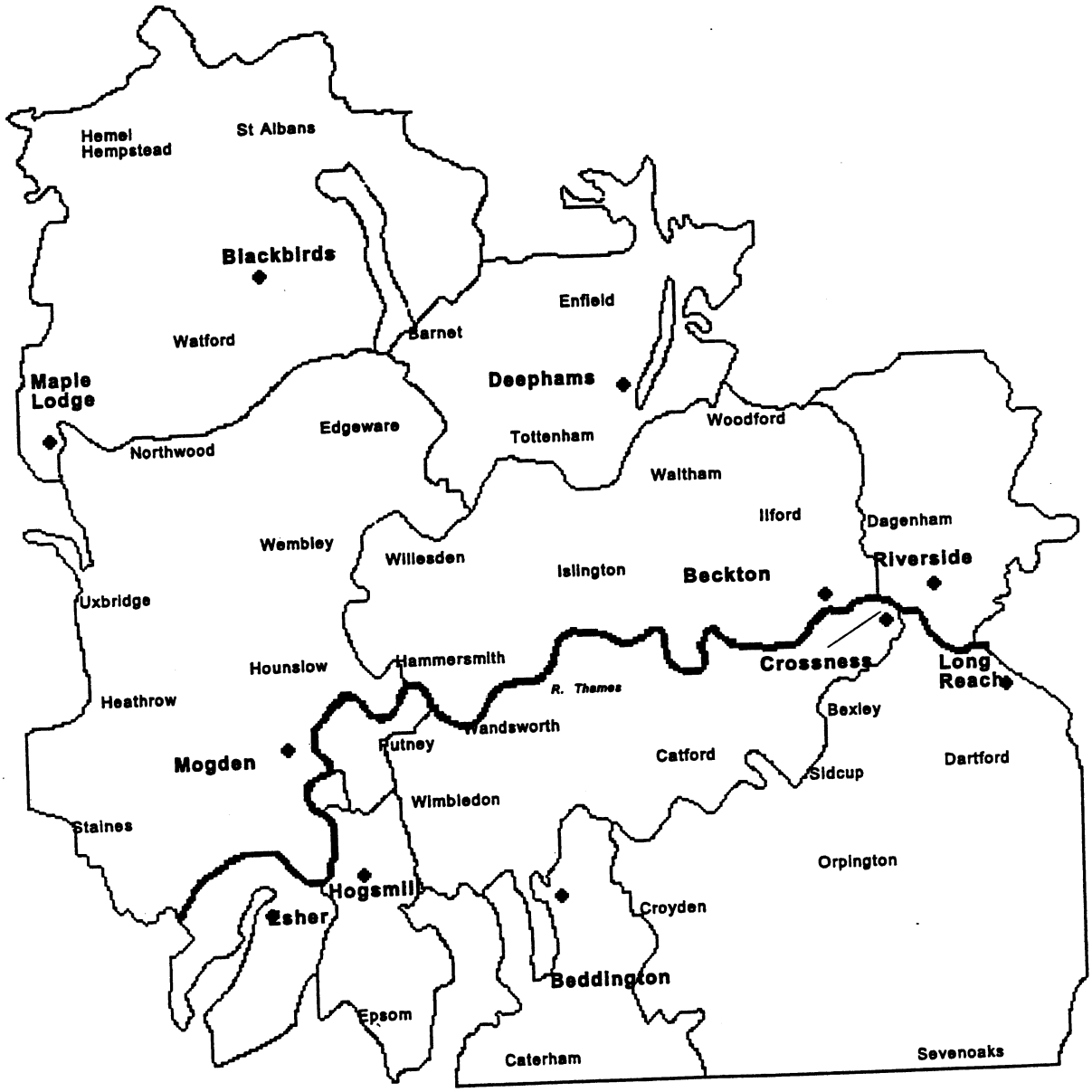


Figure 9: Populations served and flow rates of Greater London sewage works

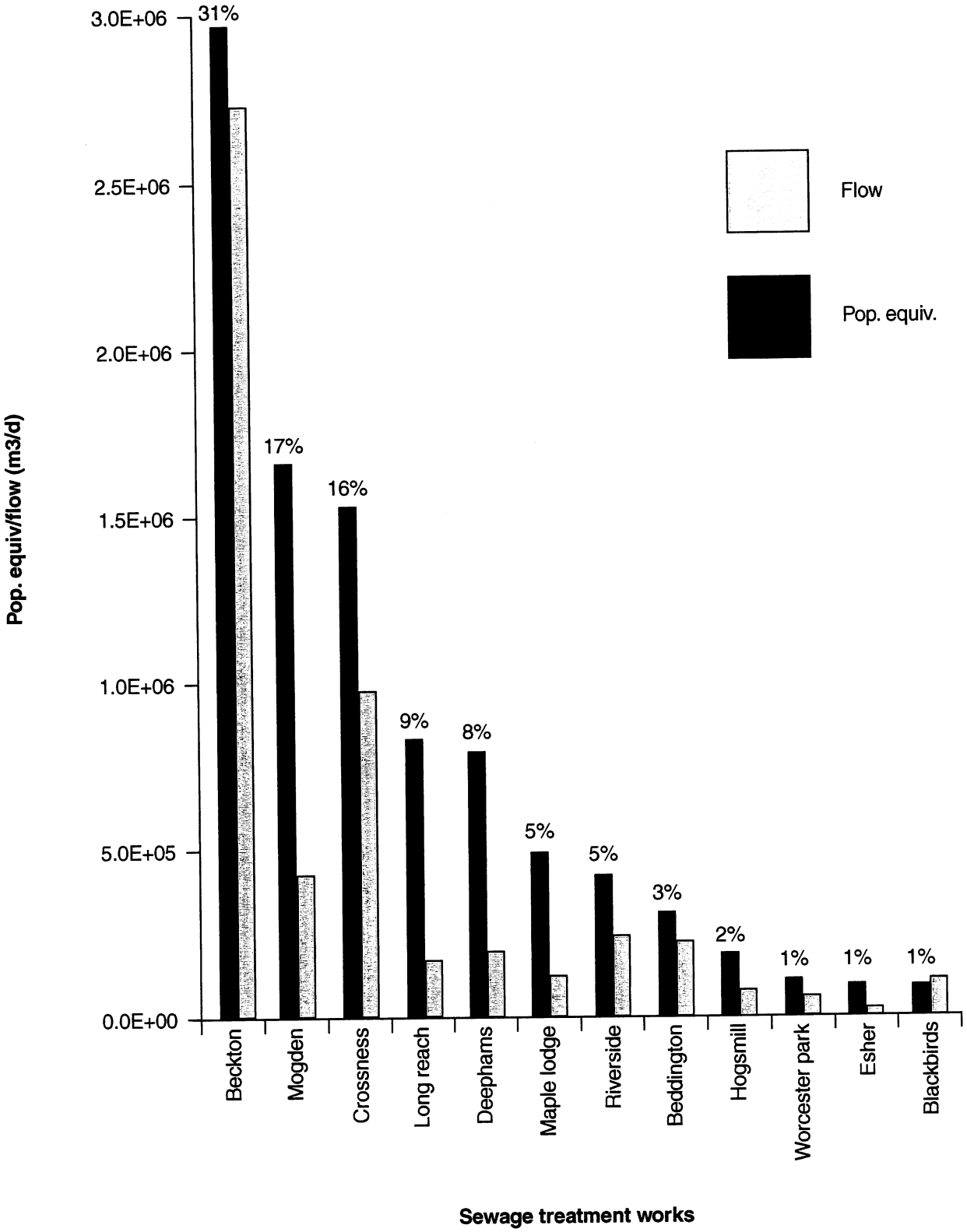


Figure 10: Suspended solids in effluent

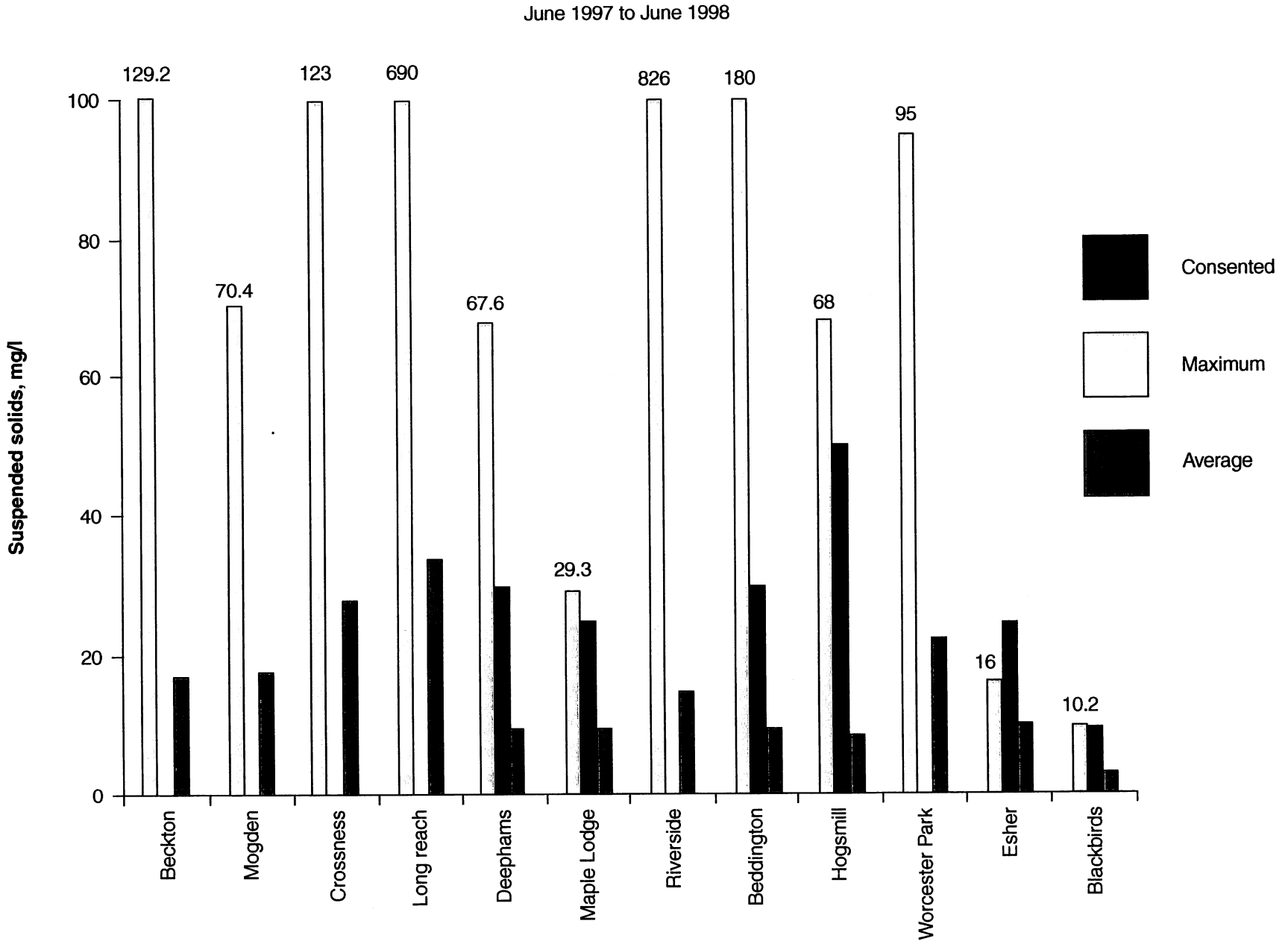


Figure 11: London works- population served and sludge produced

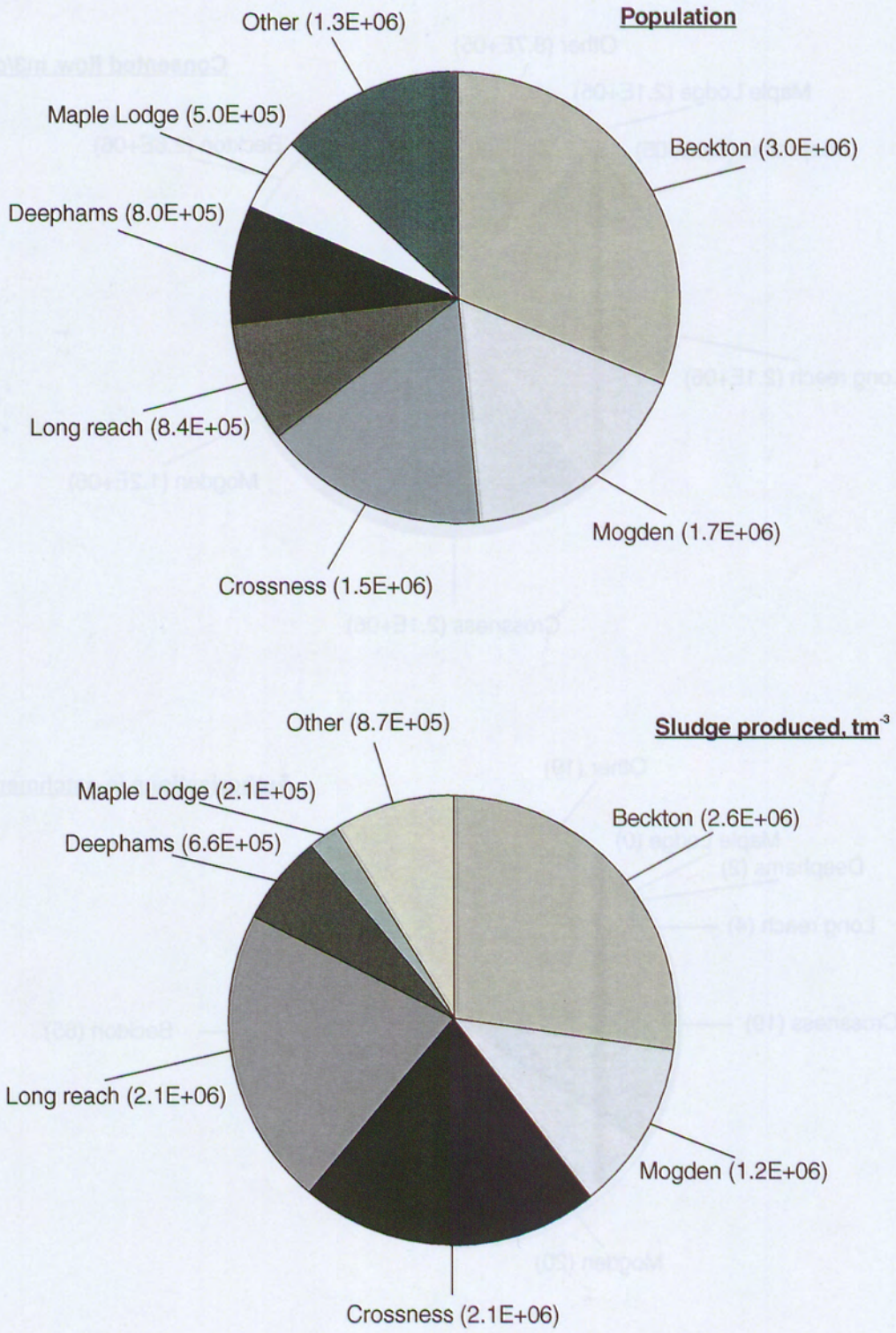
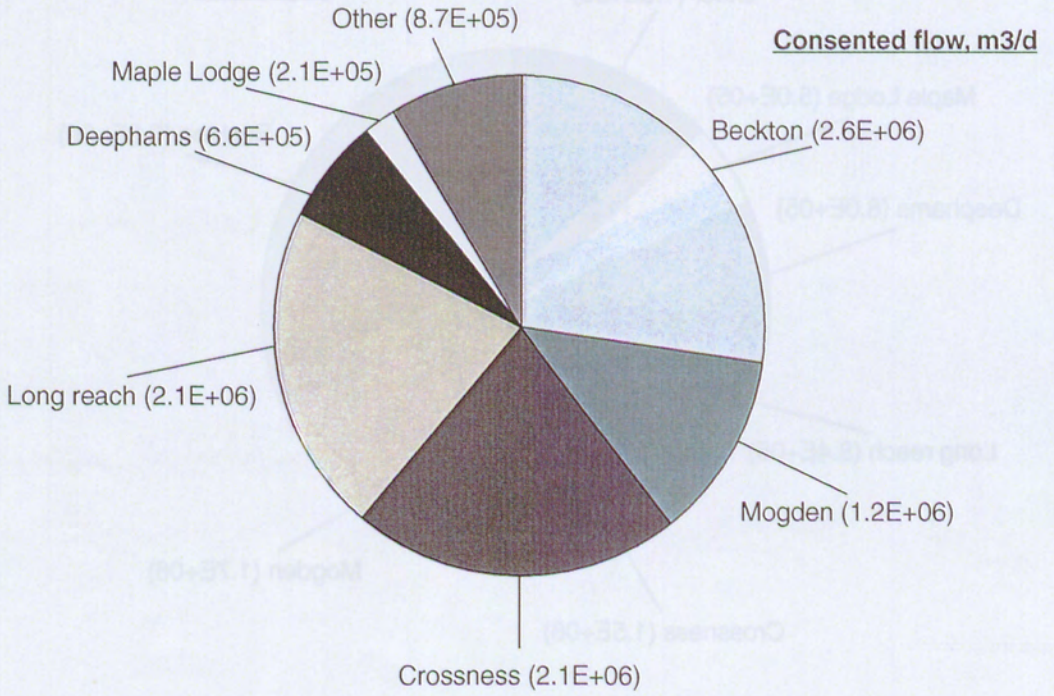
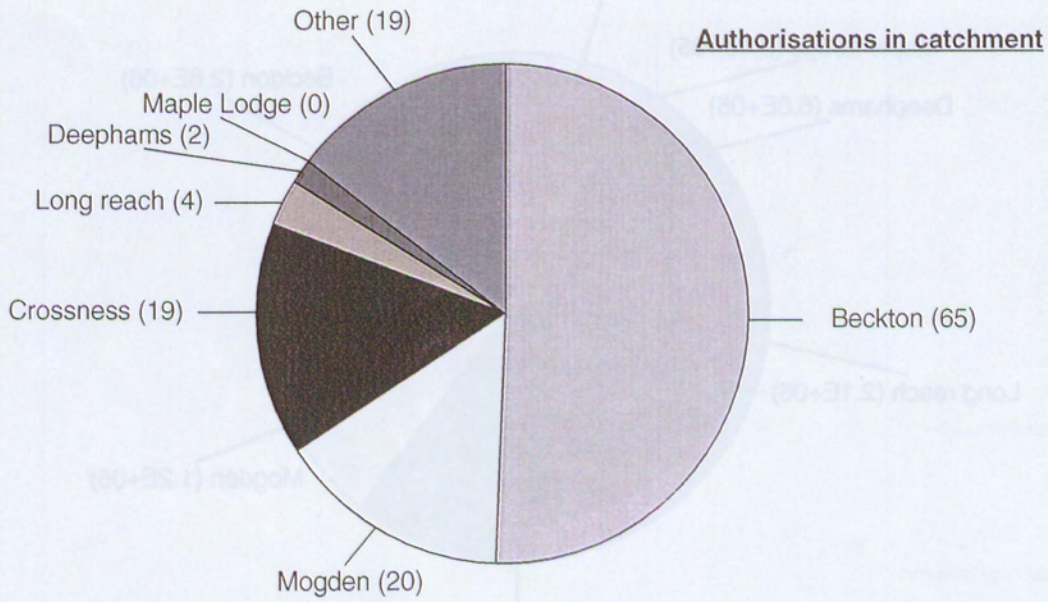


Figure 12: London works- flow rate and number of authorised discharges





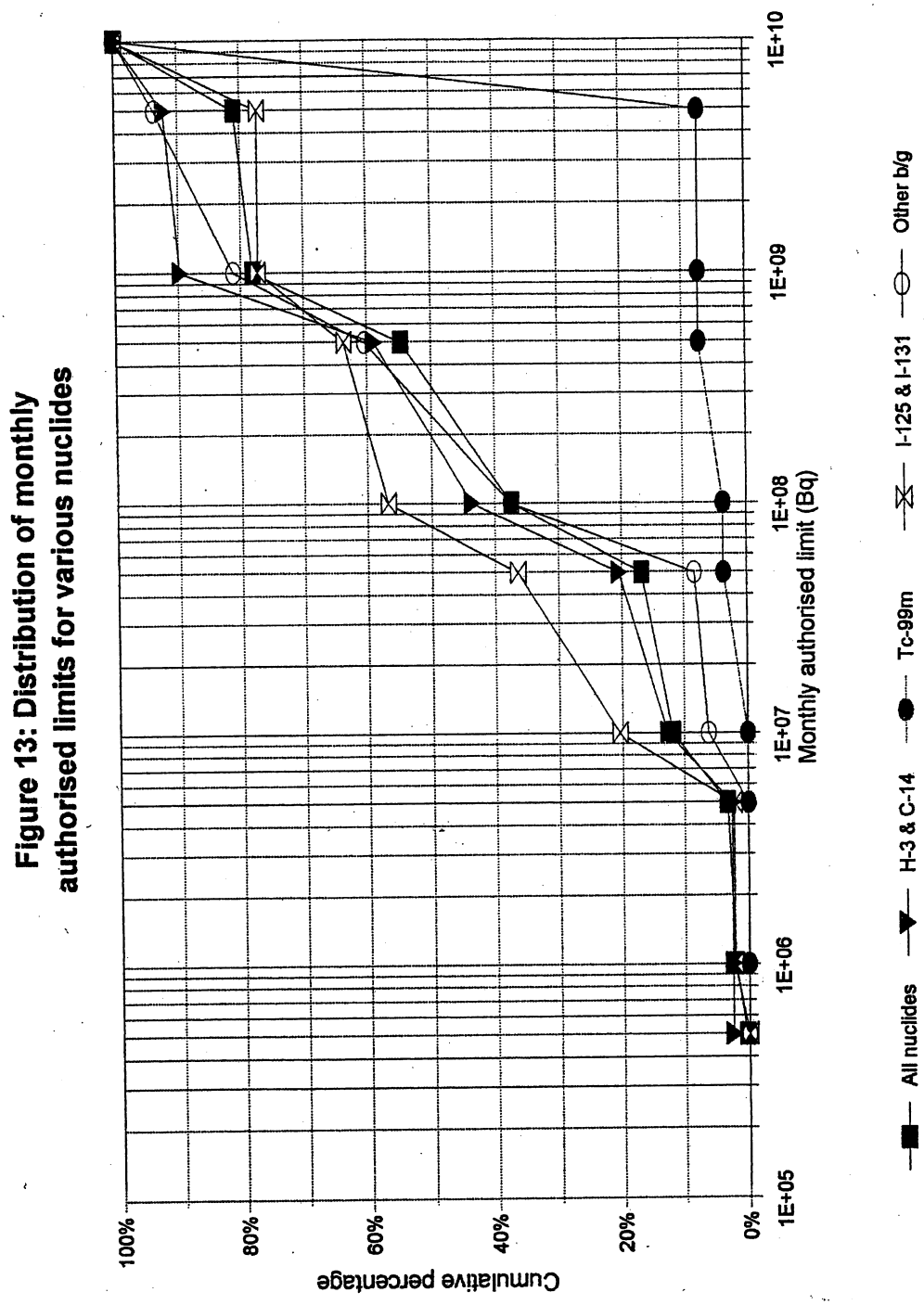


Figure 13: Distribution of monthly authorised limits for various nuclides

Figure 14: Frequency nuclides are specified in authorisations

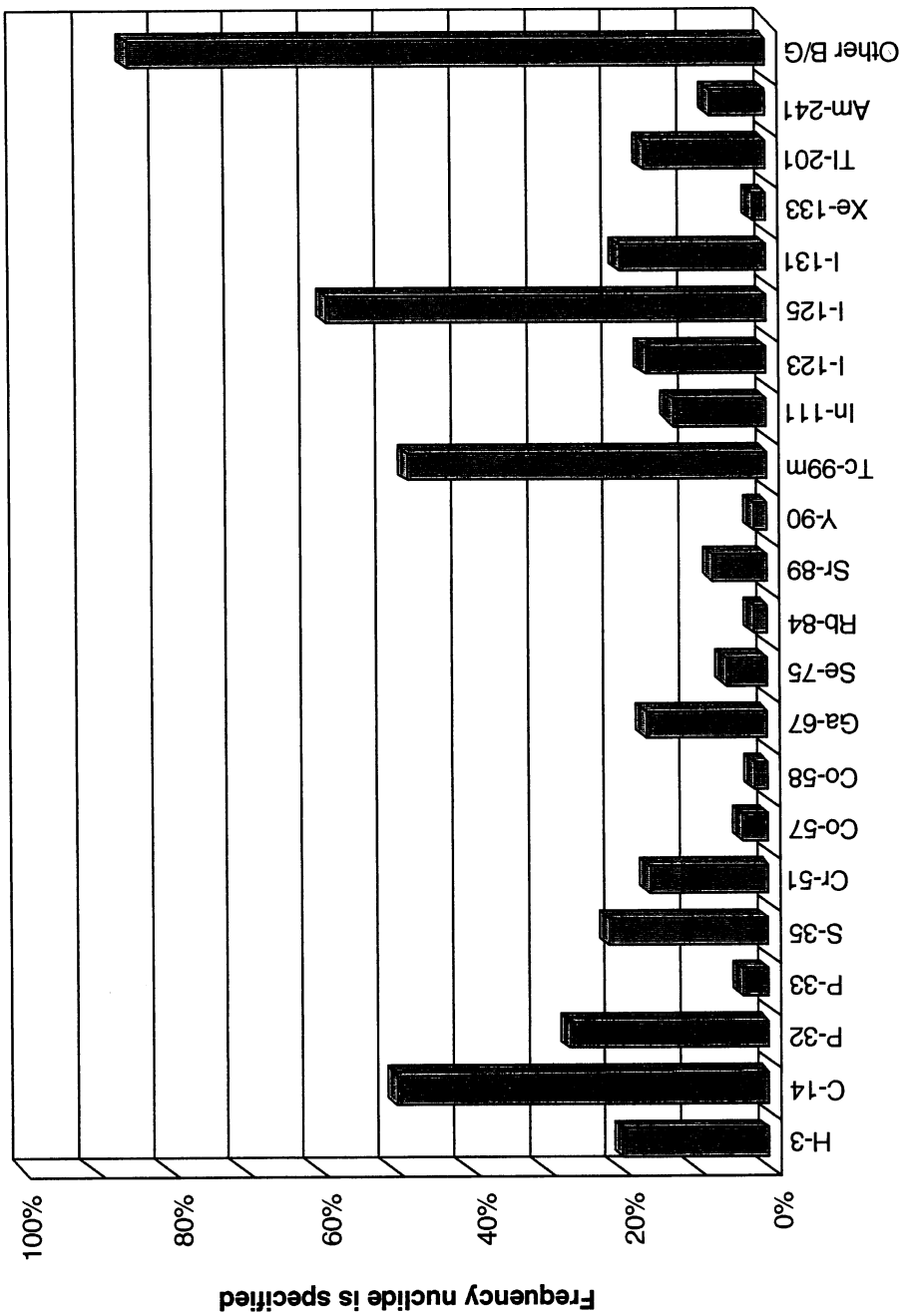


Figure 15:Distribution of conservative dose estimates for each disposer

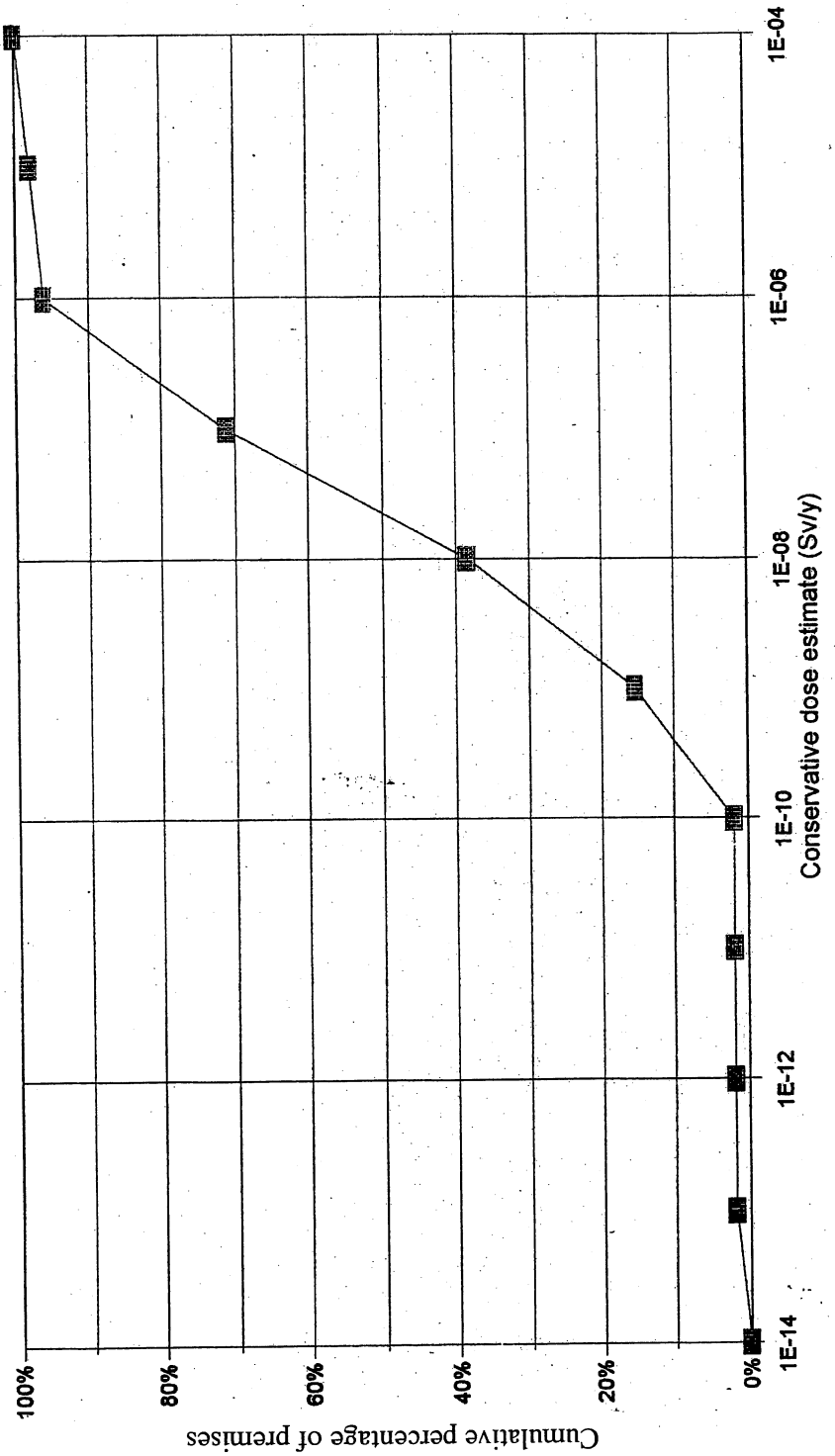
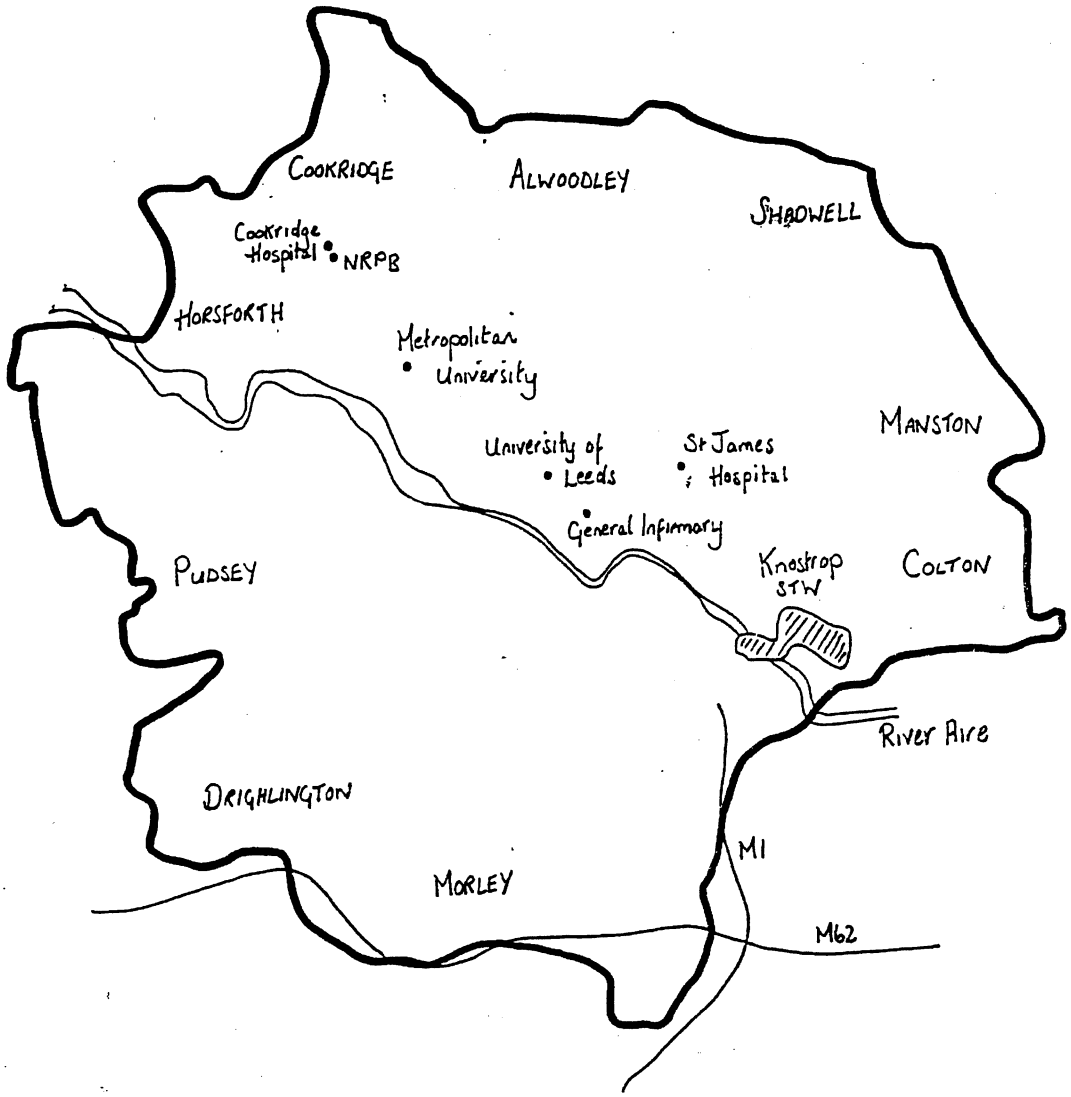


Figure 16 Sewage catchment map for Leeds



APPENDIX A

Removal efficiencies

1 APPROACH FOR ESTIMATING REMOVAL EFFICIENCIES

Data relating to removal coefficients during sewage treatment was not available for a number of nuclides considered in the study. For these nuclides estimates were made based on measured data, sediment Kd's and removal efficiencies of chemically analogous nuclides. These values were compared, a suitable range was identified and a best estimate chosen.

1.1 Sediment distribution coefficients

The sorption of elements onto sediments and particles, and the subsequent removal of these particles, is the main process of radionuclide partitioning and removal from the effluent phases during sewage sludge processing, especially during primary treatment. A measure of sorption onto solids is the equilibrium distribution coefficient, usually denoted by K_d, in $m^3\ t^{-1}$. This value represents the activity concentration on the solid phase (in Bq t^{-1}) divided by the concentration in the aqueous phase (in Bq m^{-3}), at equilibrium. Distribution coefficients have been measured for many media and may provide a method of estimating removal efficiencies for other radionuclides.

The distribution coefficient may be converted into an effective removal efficiency for a given suspended solid fraction in the sewage (F_S), and solids removal efficiency of the process (ϵ_S). These may be combined with the density of water (ρ_W) and the K_d to predict the total removal efficiency, ϵ_{TOT}:

$$\epsilon_{TOT} = \frac{k_D \times F_S \times \rho_W}{1 + (k_D \times F_S \times \rho_W)} \times \epsilon_S$$

Distribution coefficients for river sediments have been published in reference 1. Whilst not having the same organic matter content of the material in sewage these values may provide a guide to the elemental attachment to suspended particles. Values of ϵ_{TOT} have been calculated using these K_ds and assuming a solids fraction of 0.03 % (i.e. a solids loading of 300 g m^{-3}) and removal efficiency of 100%. Use of Kd may be reasonably appropriate for estimating removal efficiencies during primary treatment.

However, Kd is less likely to be appropriate to calculate removal to solid during secondary treatment, because this process is not controlled by surface sorption, but is mainly a bacterial growing process, where dissolved and finely suspended matter is used as food by bacteria and incorporated into the bacterial body mass.

1.2 Comparison of removal efficiencies with sediment K_d

The range of reported removal efficiencies have been compared with the removal efficiencies estimated using the river sediment K_ds for four radionuclides. These data have been presented in Table 1. No comparison was possible for other nuclides because of lack of data. The following comments may be made.

The removal efficiencies for tritium and carbon-14 estimated using K_d do not fit the measured ranges well. Both have measured removal efficiency of about 20%, whereas the K_d estimate varies from 0% (for tritium) to 37.5% for carbon-14. This is likely to be a result of the high rate of microbial action during sewage treatment, which results in a rapid turnover of hydrogen and carbon. This acts to increase the tritium content of solids (a portion of which is taken into bacteria) and decrease the carbon-14 uptake (because the same action is also releasing previously incorporated carbon isotopes).

For chromium, the removal efficiency estimated using sediment K_d is higher than the measured range. However, in both cases, chromium has high removal efficiency.

The estimated value for iodine appears to be below the measured range. The reason for this has not been established although it is likely to be a result of either the characteristics of the chemical compounds in which the iodine is incorporated or the high rate of microbial action during the processing.

In conclusion, the K_d approach does not appear to agree well with the measured data, although trends are in agreement. Therefore the K_d will only be used to indicate if a radionuclide is removed efficiently (a range of 60 - 80% for the removal efficiency would appear suitable to use for such nuclides) or inefficiently (with a range of 10 - 20% used for the removal efficiency).

Table A.1 Comparison of K_ds with best estimate removal efficiencies

Element	River sediment k_d [A] m^3t	Range of removal efficiency, %		
		Based on k_d	Activated sludge	Percolating filter
Hydrogen	1	0	20	20
Carbon	$2\ 10^3$	38	8.9 - 24	8.9 - 24
Chromium	$5\ 10^4$	94	38 - 88	38 - 88
Iodine	20	0.6	15 - 98	50 - 85

NOTES

A River sediment K_ds have been obtained from reference 29 (main report)

2 References

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

APPENDIX B

METHODOLOGY FOR ASSESSING THE FATE OF RADIOACTIVE DISCHARGES TO SEWERS (SMART)

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1 INTRODUCTION

This appendix describes the sewer assessment model developed for this study.

The model is divided into 4 main parts: transport of the sewage to the treatment works, treatment of sewage at the works, treatment of sludge at the works, and disposal of created sewage effluent and sludge.

In each part, the activity concentration of each nuclide in each medium is calculated, and used to estimate collective and/or individual doses for the public or workers, as relevant. As identified in the main part of the report, 17 radionuclides are considered. These are tritium, carbon-14, phosphorus-32, sulphur-35, chromium-51, gallium-67, rubidium-83 and -84, strontium-89, yttrium-90, technetium-99m, indium-111, iodines-123, -125, -131, thallium-201 and americium-241.

In this appendix, sewage and sewage effluent will refer to the liquid which undergoes preliminary, primary, secondary and tertiary treatment. Sludge will refer to the part of higher solids content, which is removed at each stage (raw sludge, primary sludge, secondary sludge after each treatment type) and then separately treated before disposal. Treated effluent, or final effluent, is the liquid eventually disposed of after treatment, in a river or an estuary.

2 TRANSPORT OF SEWAGE TO THE WORKS

The sewage transport system is designed to carry raw effluent from the place of disposal to the sewage treatment works. The transport system is normally a network of underground pipes of varying diameter, the smallest pipes being near each disposer, which may then discharge into increasingly larger trunk sewers which carry the combined effluent to the works. The transport system is designed to transport effluents with sufficient velocity to prevent solids settling out and causing blockages. However, blockages do occur, and larger pipes may require maintenance. In these situations, individuals will be exposed to any radionuclides carried in the effluent.

2.1 Transport in small pipes

Small pipes are the pipes of around 0.2 m in diameter, which take the sewage from the authorised disposer to the main sewers.

The activity concentration (Bq m^{-3}) is the total amount of activity released per nuclide i per discrete discharge divided by the typical volume of such a discharge. It is assumed that at this point all the activity is dissolved in liquid.

$$C_{i,d} = \frac{B_{i,d}}{V_d}$$ Equation 1

where $C_{i,d}$ is the average activity concentration of nuclide i disposed in a single discharge by disposer d (in Bq m^{-3}),
$B_{i,d}$ is the average number of becquerels of nuclide i disposed in a typical single discharge (Bq),
V_d is the average volume of liquid disposed of in one single discharge by disposer d (m^3).

Values for $B_{i,d}$ are presented in tables D17 (for Beckton) and D24 (for Knostrop) of Appendix D. A mean value of 2 m^3 is used for V_d.

Note: if volumes are given in litres, they have to be converted to m^3:
1000 Bq m^{-3} = 1 Bq l^{-1} and 1 m^3 = 1000 l

2.2 Exposure to radionuclides transported in small pipes on and off site

Exposure to the radioactivity disposed of in public sewers may arise when the lateral pipes between the disposer and the main sewer get blocked or need repairing. Maintenance workers would then have to spend some time repairing the damaged or blocked pipes. This is likely to happen on a rare and irregular basis, although some workers may be specifically working on one site over several disposals and therefore there may be repetition of exposures. In addition workers may be involved in repairing pipes immediately off site. Releases of radionuclides over about 5 hours are considered. It is assumed that discharged activity is retained in a blockage and the workers are exposed during repair. The released activity is treated as a point source for external dose. Consideration is also given to exposure of gangs repairing pipes off site. Exposure assumes dilution of radionuclides released in the total flow of sewage from a disposer.

The exposure pathways associated with repairing or unblocking pipes will therefore be: external irradiation from standing next to the pipe, inhalation of sprayed or re-suspended material, and inadvertent ingestion of effluent. Given the nature of the material handled, skin contact is unlikely to be an important pathway, especially since protective clothing is likely to be worn.

2.2.1 Individual doses to workers

There may be a delay between the discharge of radionuclides and the arrival of the worker on site, during which short-lived nuclides will have decayed:

$$C_{i,D} = C_{i,d} \cdot e^{-\frac{\ln 2 * T_d}{T_{Hi}}}$$ Equation 2

where $C_{i,D}$ is the activity concentration of nuclide i in the pipe after decay (Bq m^{-3}),
T_d is the delay between discharge and work being carried out (h), as presented in table D2 of Appendix D, T_{Hi} is the half-life of nuclide i (h), see table D1 of Appendix D.

External irradiation
The annual effective external dose received by a person working for one of the disposers can be expressed as follows:

$$_{ext}D_{i,d} = C_{i,D} \cdot V_d \cdot H_{i,ext,d} \cdot T_w \qquad \text{Equation 3}$$

where
$_{ext}D_{i,d}$ is the annual external dose to a worker for nuclide i and disposer d ($Sv.y^{-1}$),
$H_{i,ext,d}$ is the dose rate for external exposure to nuclide i in the specified geometry for pipes ($Sv\ h^{-1}$ per Bq, or $Sv\ h^{-1}$ per $Bq\ m^{-3}$)
T_w is the number of hours per year spent by a worker carrying out repair/maintenance for a particular disposer ($h\ y^{-1}$), as presented in table D16 of Appendix D.

T_w can be derived from the number of hours taken to un-block a pipe and the number of blockages attended per year.

Blockages in small pipes are assumed to be point sources. External dose rates at 1 m from point sources are presented in table D3 of Appendix D.

Inhalation of re-suspended material
The annual effective inhalation dose to a worker is:

$$_{inh}D_{i,d} = (C_{i,D} / \rho_e) \cdot H_{i,inh} \cdot L_d \cdot Brh \cdot T_w \qquad \text{Equation 4}$$

where $_{inh}D_{i,d}$ is the annual inhalation dose for nuclide i from disposer d ($Sv.y^{-1}$),
ρ_e is the density of effluent, approximately equal to that of water ($\rho_e = 1000\ kg\ m^{-3}$),
$H_{i,inh}$ is the inhalation dose coefficient ($Sv\ Bq^{-1}$), as presented in table D6 of Appendix D,
L_d is the suspended solids in air concentration ($kg\ m^{-3}$), as presented in table D7 of Appendix D,
Brh is the average breathing rate of a worker ($m^3\ h^{-1}$).

Inadvertent ingestion
The annual effective ingestion dose is calculated as follows:

$$_{ing}D_{i,d} = (C_{i,D} / \rho_e) \cdot H_{i,ing} \cdot Ing \cdot T_w \qquad \text{Equation 5}$$

where $_{ing}D_{i,d}$ is the annual inadvertent ingestion dose from nuclide i from disposer d ($Sv\ y^{-1}$),
$H_{i,ing}$ is the ingestion dose coefficient ($Sv\ Bq^{-1}$), as presented in table D6 of Appendix D,
Ing is the inadvertent ingestion rate ($kg\ h^{-1}$), as presented in table D8 of Appendix D.

Total annual dose
The total annual individual effective dose received by a worker is the sum of the doses from the 3 pathways for all nuclides, for disposer d:

$$D_{ind,\,d} = \sum_{i=1}^{10} ({}_{ext}D_{i,d} + {}_{inh}D_{i,d} + {}_{ing}D_{i,d}))$$ Equation 6

where $D_{ind,\,d}$ is the annual individual effective dose (Sv y^{-1}),
10 is the total number of radionuclides considered.

If workers carried out work at several different premises disposing radionuclides, $C_{i,\,d}$ should be the average activity concentration over all disposers.

2.2.2 Collective doses to workers

Annual collective doses to workers will be estimated from the annual individual doses, and the number of workers involved in maintenance:

$${}_{x}D_{i,\,coll,\,d} = {}_{x}D_{i,\,d} \cdot n_d$$ Equation 7

where ${}_{x}D_{i,\,coll,\,d}$ is the collective dose for nuclide i and pathway x (x = ext, inh, ing) (man Sv y^{-1}),
${}_{x}D_{i,\,d}$ is the individual annual dose from pathway x (Sv y^{-1}),
and n_d is the number of workers carrying out maintenance (man), as presented in tables D19 and D26 of Appendix D.

The total annual collective dose is:

$$D_{coll,\,d} = D_{ind,\,d} \cdot n_d$$ Equation 8

where $D_{coll,\,d}$ is the total annual collective dose (man Sv y^{-1}), and all others as above.

2.3 Transport in man-accessible pipes: dilution and sorption

The main sewers carry all the sewage from one area (catchment area) to the works. They may be large enough for a person to stand in them, to carry out repairs or inspections. The 'large pipes' in this section are assumed to be 2 m in diameter.

Exposure to the radioactivity in the sewage will arise whenever workers need to spend time within the pipe. The total amount of radionuclides discharged in the catchment area is mixed in the total amount of sewage produced by the catchment area. The radioactivity is effectively diluted and mixed in the whole flow of sewage. Some of the solids may be deposited on the pipe walls, creating a thin film (thickness 0.01 m) of material covering the walls of the pipe.

2.3.1 Dilution

The total amount of radionuclide i discharged by all the disposers in the works' catchment area is diluted in the total sewage flow:

$$C_{i,\,p} = \frac{q_i}{Q_0}$$ Equation 9

where $C_{i,p}$ is the average activity concentration in sewage in the pipe for nuclide i (Bq m^{-3});
q_i is the disposal rate of nuclide i summed over all disposers (Bq d^{-1}), as presented in tables D20 and D27 of appendix D,
Q_0 is the sewage volumetric flow rate (m^3 d^{-1}) in the large pipes, for Beckton and Knostrop, as presented in tables D22 and D29 of appendix D.

2.3.2 Sorption onto pipe walls

Solid matter has been assumed to build up on sewer walls by a process of particle deposition. Radionuclides may be associated with these solids.

The process is represented by a simple system of compartmental models representing the transit of solids in the sewage flow, and the rate of deposition of solids onto walls. It has been assumed that a depth of 1 cm of solid matter builds up on the walls per month. A storm event, or other process, has then been assumed to wash all the matter from the sewer walls. The rate of deposition onto walls r_W (in cm^3 of solids per m^2 of walls per day) is given by:

$$r_W = (d_W \times a_W) \div t_W \qquad \text{Equation 10}$$

where d_W = Depth of solids on walls, assumed to be 1 cm
a_W = Area of walls covered, 10^4 cm^2 per m^2
t_W = Time taken for sludge to build up (30 days)

The volumetric rate of deposition onto sewer walls is therefore 333.3 cm^3 m^{-2} d^{-1}. Now, this rate is related to the total rate of transfer of sewage solids in the sewer. Assuming the trunk sewer carries the entire consented flow rate, the rate of transit of sewage solids in this sewage may be calculated by:

$$r_S = v_S \times cpm \times f_S \qquad \text{Equation 11}$$

where cpm = Conversion factor for cm^3 per m^3 (10^6)
f_S = Fraction of sewage that is solid matter, assumed to be $3\ 10^{-4}$

The transfer rates r_W and r_S represent the transit and deposition of solids in the sewer. They also apply to continuous, uniform releases of radionuclides which are associated with the solids. Therefore the transfer coefficients may be normalised so that they are expressed in Bq per day, rather than cm^3 of solids per day. The rate of transfer of activity onto walls, r_{AW} in Bq m^{-2} d^{-1} per Bq d^{-1}, and in sewage, r_{AS} in Bq d^{-1} per Bq d^{-1}, therefore become:

$$r_{AW} = r_W \div (r_W + r_S) \qquad \text{Equation 12}$$

$$r_{AS} = r_S \div (r_W + r_S) \qquad \text{Equation 13}$$

A simple compartmental model has been set up to solve this problem for nuclides with a range of half lives and thereby examine their build up on sewer walls.

At this stage, it will be assumed that all the activity present in the sewage has sorbed on solids which uniformly cover the pipes walls (cautious assumption). The transfer coefficients to pipe

walls (G_i in Bq m^{-2} per Bq d^{-1}) have been calculated for the radionuclides considered in this study and are presented in table 9 of Appendix D.

Short half life radionuclides such as Tc-99m reach a position of equilibrium quickly, whereas longer lived radionuclides (such as H-3, with an effectively infinite half life in comparison to the time period of 30 d) continue to build up. Many radionuclides of interest in the study have half lives such that they lie between these two results. Peak concentrations have been used in this assessment.

2.4 Exposure to radionuclides transported in man-accessible pipes

2.4.1 Individual doses to workers

External irradiation

It will be considered that the average distance between the sewer worker and the pipe walls is 1m. The annual effective dose from external exposure to the sewage in large pipes can be calculated as:

$$_{ext}D_{i,p} = q_i \cdot (G_i / w) \cdot H_{i,ext,p} \cdot T_p \qquad \text{Equation 14}$$

where $_{ext}D_{i,p}$ is the annual effective dose from external exposure in large pipes (Sv y^{-1}),
q_i is the disposal rate (Bq d^{-1}),
G_i is the transfer coefficient to walls (Bq m^{-2} per Bq d^{-1}),
w is the weight of a m^2 of solids deposited on the pipe walls over a 1cm thick layer, assuming a density of 1000 kg m^{-3} (i.e. 10 kg m^{-2} or 10 000 g m^{-2}),
$H_{i,ext,p}$ is the external dose rate from inside a sewage pipe (Sv h^{-1} per Bq g^{-1}), as presented in table D4, and T_p is the time spent in large pipes per year (h y^{-1}) by a typical worker, as presented in table D16 of Appendix D.

Inhalation

Annual doses from the inhalation of re-suspended sludge for radionuclide i in the large pipes can be calculated as:

$$_{inh}D_{i,p} = (C_{i,p} / \rho_e) \cdot H_{i,inh} \cdot L_p \cdot Brh \cdot T_p \qquad \text{Equation 15}$$

where $_{inh}D_{i,p}$ is the inhalation dose for nuclide i (Sv y^{-1}),
ρ_e is the density of effluent, approximately equal to that of water (ρ_e = 1000 kg m^{-3}),
$H_{i,inh}$ is the inhalation dose coefficient (Sv Bq^{-1}),
L_p is the suspended solids in air concentration (kg m^{-3}), as presented in table D7 of Appendix D,
Brh is the breathing rate of a typical worker (m^3 h^{-1}),
T_p is the time spent by workers in man accessible pipes (h y^{-1}).

Inadvertent ingestion
The inadvertent ingestion dose for workers in large pipes can be calculated as:

$$_{ing}D_{i,\,p} = (C_{i,\,p} / \rho_e) \,.\, H_{i,\,ing} \,.\, Ing \,.\, T_p \qquad \text{Equation 16}$$

where $_{ing}D_{i,\,p}$ is the inadvertent ingestion dose (Sv y^{-1}),
$H_{i,\,ing}$ is the ingestion dose coefficient (Sv Bq^{-1}),
Ing is the inadvertent ingestion rate (kg h^{-1}).

Total annual dose
The total annual individual dose incurred by a typical sewers worker is the sum of the effective doses presented above:

$$D_{ind,\,p} = \sum_{i=1}^{10} (_{ext}D_{i,p} + _{inh}D_{i,p} + _{ing}D_{i,p}) \qquad \text{Equation 17}$$

where $D_{ind,\,p}$ is the individual annual dose from exposure in large pipes (Sv y^{-1}),
i is the index referring to each nuclide.

2.4.2 Collective doses to workers

The collective doses to workers will be calculated from the individual annual doses and the number of workers in the catchment area:

$$_xD_{i,\,coll,\,p} = _xD_{i,\,p} \,.\, n_p \qquad \text{Equation 18}$$

where $_xD_{i,coll,\,p}$ is the collective dose for nuclide i and pathway x (x = ext, inh, ing) (man Sv y^{-1}),
$_xD_{i,\,p}$ is the individual annual dose from pathway x (Sv y^{-1}),
and n_p is the number of workers carrying out maintenance (man), as presented in tables D19 and D26 of appendix D.

The total annual collective dose is:

$$D_{coll,\,p} = D_{ind,\,p} \,.\, n_p \qquad \text{Equation 19}$$

where $D_{coll,\,p}$ is the total annual collective dose (man Sv y^{-1}).

2.5 Arrival at the works: decay and sorption

Transport time has an influence on the amount of radionuclides still present at the entrance of the works since some of them may have significantly decayed. The total amount of radioactivity discharged per day is considered to be diluted within the total amount of sewage arriving at the works per day.

$$C_i = \frac{q_i}{Q_0} * e^{-\frac{\ln 2 * T_t}{T_{Hi}}} \qquad \text{Equation 20}$$

where C_i is the activity concentration in sewage at arrival at the works for nuclide i (Bq m^{-3}),

q_i is the disposal rate of nuclide i summed over all disposers (Bq d^{-1}),
Q_0 is the sewage volumetric flow rate (m^3 d^{-1}) input to the works, which is the Dry Weather Flow (DWF) in normal conditions and may also include the flow from storm events,
T_t is the transport time from disposers to the works (h), defined as

$$T_t = \frac{dis\tan ce\_to\_the\_works(m)}{Average\_velocity\_of\_sewage(m.h^{-1})} \qquad \text{Equation 21}$$

Distances to the works are given in tables D18 and D25 of appendix D, the average velocity of sewage is assumed to be 0.5 m s^{-1} (30 m h^{-1}).
T_{Hi} is the half-life of nuclide i (h).

3 TREATMENT OF SEWAGE AT THE WORKS

The sewage treatment process is usually made up of 3 stages: a preliminary treatment, which removes large floating solids and fat by screening, and settles grit through a rapid process. This is followed by a primary sedimentation, which sediments suspended solids to remove them from the main flow. Then the secondary treatment is designed to remove the biological oxygen demand (BOD) from the effluent, so that liquids can be returned to water courses without causing the oxygen levels in the receiving water body to be reduced significantly. BOD in effluent is reduced by removing suspended materials from the effluent and aerobic digestion of the remaining liquid phase. In the UK, two types of aerobic digestion are commonly used: activated sludge and percolating filters. A tertiary treatment may be applied in some cases, for example in areas with specific problems, such as nitrates pollution, or when the liquid effluent is disposed of in small rivers. Tertiary treatment will be treated as optional in this model. See figure 3 for illustration.

The behaviour of radionuclides during these processes cannot be explained fully by K_d. Elements removal efficiencies are normally obtained by measuring the concentration of a trace constituent before and after a process, the difference being the removal efficiency. Measured removal efficiencies were available for some of the radionuclides in this study. These are shown in tables 10 – 12 of the main report. Where removal efficiencies were not available they have been derived. The equations for calculating removal efficiencies are given below, followed by the equations for calculating the radionuclide concentrations at each of the stages of sewage treatment. Equations for the calculation of doses are presented in chapter 5.

3.1 Removal efficiencies of radionuclides

Radionuclides removal efficiencies are dependent on the removal of solids in the sewage and the fraction of activity attached to solids.

The solids removal efficiency is defined as the efficiency of a certain treatment at removing the suspended solids in the sewage:

$$E_{s\,j} = \frac{F_j}{F_{j-1}} \qquad \text{Equation 22}$$

where $E_{s\,j}$ is the efficiency of treatment j at removing solids (kg m^{-3} per kg m^{-3});
F_j is the suspended solids load after treatment j (kg m^{-3}).

Solids removal efficiencies are usually known for typical processes. They are part of the normal design requirements of a sewage treatment plant.

The radionuclide removal coefficient is the term used in SMART, this is equal to (1 – Removal Efficiency. The removal coefficient is the fraction of nuclide i per unit volume remaining in the effluent after sewage treatment j, or in the sludge after sludge treatment j:

$$E_{tot,i,j} = \frac{C_{ijk}}{C_{ij-1k}} = \frac{k_{Di} \cdot F_{j-1} \cdot \rho_e}{1+(k_{Di} \cdot F_{j-1} \cdot \rho_e)} \cdot E_{s,j} \qquad \text{Equation 23}$$

Where $E_{tot\ i,j}$ is the removal coefficient for treatment j, which is the amount of nuclide i remaining in the medium (Bq m^{-3} per Bq m^{-3}),
$k_{D,i}$ is the equilibrium distribution coefficient, representing the proportion of nuclide i becoming attached to solids in the medium (Bq kg^{-1} per Bq m^{-3}),
ρ_e is the density of effluent, approximately equal to that of water (ρ_e = 1000 kg m^{-3}), and other as defined above.

Radionuclide removal coefficients have been used preferably to K_ds as some are available in the literature, and more relevant to the present situation. Where removal coefficients were not available, they have been derived using K_ds and Fs. The best estimate values used in the model SMART are given in tables D10, D11, D12, and D13 of appendix D.

The amounts of solids and liquid involved can be used to work out suspended solids loads and removal efficiencies: the total amount of solids in the sewage before treatment is the sum of the amount of solids in the sludge and in the effluent.

$$(1000\text{-}F_{j-1}) \cdot Q_{j-1} = (1000\text{-}F_{js}) \cdot Q_{js} + (1000\text{-}F_{je}) \cdot Q_{je} \qquad \text{Equation 24}$$

where Q_{j-1} is the flow rate of the sewage before treatment j ($m^3\ d^{-1}$),
F_{j-1} is the suspended solids load of the sewage before treatment (kg m^{-3}),
Q_{js} is the production rate of sludge from treatment j ($m^3\ d^{-1}$),
F_{js} is the suspended solids load in the sludge (kg m^{-3}),
Q_{je} is the production rate of effluent from treatment j ($m^3\ d^{-1}$), and
F_{je} is the suspended solids load in the effluent (kg m^{-3}).

3.2 Preliminary treatment of the sewage (step 0)

The preliminary treatment consists of the removal of grit and bigger floating solids from the sewage. This is likely to have little influence on the suspended solids load, but it may remove the radionuclides that had sorbed on those solids. It is considered that compared with the total effluent flow, a negligible amount of liquid is removed with the solids by these processes.

The amount of solids produced by preliminary treatment is Q_{0s} ($m^3\ d^{-1}$), as presented in tables D23 and D30 of Appendix D. It does not undergo any further treatments and is disposed of in landfills separately from other sludges.

3.2.1 Effect on radionuclide concentrations

Taking into account radioactive decay, the activity concentration of nuclide i in the sewage effluent, at the end of preliminary treatment is:

$$C_{i0e} = E_{TOT\,i\,0} \cdot C_i \cdot e^{-\frac{\ln 2 * T_0}{T_{Hi}}} \qquad \text{Equation 25}$$

where C_{i0e} is the activity concentration in sewage effluent for nuclide i (Bq m^{-3}),
$E_{TOT\,i\,0}$ is the removal coefficient of radionuclide i for preliminary treatment (Bq m^{-3} per Bq m^{-3}), as presented in table D10 of Appendix D,
C_i is the activity concentration of nuclide i in sewage before preliminary treatment (Bq m^{-3}),
T_{Hi} is the half-life of nuclide i (h),
T_0 is the time spent by sewage in preliminary treatment (h), see table D2 of appendix D.

Taking into account radioactive decay, the activity concentration of nuclide i on the solids removed, at the end of preliminary treatment is:

$$C_{i0s} = (1 - E_{TOT\,i\,0}) \cdot Q_0/Q_{0s} \cdot C_i \cdot e^{-\frac{\ln 2 * T_{0S}}{T_{Hi}}} \qquad \text{Equation 26}$$

where C_{i0s} is the activity concentration of nuclide i on the solids (Bq m^{-3}), Q_0 is the effluent flow rate expressed in t/d, Q_{0s} is the amount of solids removed (t/d),
T_{0S} is the time spent by sludge in preliminary treatment (h), as presented in table D2 of Appendix D, and which may be longer that T_0, and all others as above.

3.2.2 Exposure pathways

Cleaning of the grid and grit channels with pressure-hoses may result in the re-suspension of some material and subsequent inhalation, as well as splashing and contact with the skin. Cleaning only happens approximately once every 4 years for each channel. Once the floating solids have been removed from the flow and shredded, they are dumped into skips. Workers may have to shovel spilled solids back into the container. During this operation, some material may become re-suspended or projected on workers. Finally, when the wind is strong, even within the house that protects the shredder, some material may get blown off the containers and therefore re-suspended.

Dose calculations will be presented in part 5 of this methodology, using activity concentration C_{i0e} for exposure to the effluent and C_{i0s} for exposure to the solids removed.

3.3 Primary treatment of sewage (step 1)

The primary treatment has been assumed to consist of sedimentation of the effluent. Any chemical treatments used are considered not to affect the activity concentrations in the products. As a result of the primary sedimentation, the sewage is separated into 2 parts, with

different suspended solids loads: the effluent which goes on to secondary treatment, and the primary or raw sludge which is pumped to settling tanks to undergo sludge treatment.

$$(1000\text{-}F_{0e}) \cdot Q_{0e} = (1000\text{-}F_{1s}) \cdot Q_{1s} + (1000\text{-}F_{1e}) \cdot Q_{1e} \qquad \text{Equation 27}$$

where F_{0e} is the suspended solids load in the incoming sewage (kg m^{-3}),
Q_{0e} is the sewage effluent flow rate before preliminary treatment (m^3 d^{-1}),
F_{1s} is the suspended solids load in the sludge removed during this stage (kg m^{-3}),
Q_{1s} is the rate of removal of sludge during primary treatment (m^3 d^{-1}),
F_{1e} is the suspended solids load in the sewage produced (kg m^{-3}), and
Q_{1e} is the flow rate of effluent from primary treatment (m^3 d^{-1}).
All are presented in tables D15, D23 and D30 of Appendix D.

3.3.1 Effect on radionuclide concentrations

The activity concentration of radionuclide i in the effluent at the end of the primary treatment is:

$$C_{i1e} = E_{TOT\,i\,1} \cdot C_{i0e} \cdot e^{-\frac{\ln 2 * T_1}{T_{Hi}}} \qquad \text{Equation 28}$$

where C_{i1e} is the activity concentration for nuclide i after primary treatment (Bq m^{-3}),
$E_{TOT\,i\,1}$ is the removal coefficient of radionuclide i for primary treatment (Bq m^{-3} per Bq m^{-3}), see table D11 of Appendix D,
T_1 is the time spent by sewage in primary treatment (h), see table D2 of Appendix D, and others as above.

The activity concentration of radionuclide i in the sludge at the end of the primary treatment can be calculated by:

$$C_{i1s} = (1 - E_{TOT\,i\,1}) \cdot Q_{0e}/Q_{1s} \cdot C_{i0e} \cdot e^{-\frac{\ln 2 * T_{1S}}{T_{Hi}}} \qquad \text{Equation 29}$$

where C_{i1s} is the activity concentration for nuclide i in sludge removed during primary treatment (Bq m^{-3}), $Q_{0e} \sim Q_0$ is the effluent flow rate expressed in t/d, Q_{2s} is the amount of sludge removed (t/d), T_{1S} is the time spent by sludge in primary treatment (h), as presented in table D2 of Appendix D and others as defined above.

3.3.2 Exposure pathways

Approximately once every 4 years the sedimentation tanks are emptied and cleaned out. As before this may result in the re-suspension of deposited material and its subsequent inhalation by the workers. At any one time, at a large works, there would probably be one tank being cleaned.

Dose calculations will be presented in part 5 of this methodology, using activity concentration C_{i1e} for exposure to the effluent and C_{i1s} for exposure to the solids removed.

3.4 Secondary treatment of sewage (step 2)

Secondary treatment consists of the aerobic bacterial digestion of the sewage, followed by a secondary sedimentation. The sewage is separated into 2 flows, the secondary sludge, which is mixed with primary sludge before treatment, and the effluent, which now has a low BOD and suspended solids load and may be discharged directly into a river or an estuary.

$$(1000-F_{1e}) \cdot Q_{1e} = (1000-F_{2e}) \cdot Q_{2e} + (1000-F_{2s}) \cdot Q_{2s} \qquad \text{Equation 30}$$

where
F_{2e} is the suspended solids load in the effluent after secondary treatment (kg m^{-3}),
Q_{2e} is the effluent flow rate from secondary treatment (m^3 d^{-1}), this is $\approx Q_0$.
F_{2s} is the suspended solids load in the sludge after secondary treatment (kg m^{-3}),
Q_{2s} is the sludge production rate from secondary treatment (m^3 d^{-1}), as presented in tables D23 and D30 of Appendix D.

3.4.1 Effect on radionuclide concentrations

The activity concentration of radionuclide i in effluent at the end of the secondary treatment is:

$$C_{i\,2e} = E_{TOT\,i\,2} \cdot C_{i1e} \cdot e^{-\frac{\ln 2 * T_2}{T_{Hi}}} \qquad \text{Equation 31}$$

where $C_{i\,2e}$ is the activity concentration of nuclide i in effluent after secondary treatment (Bq m^{-3}),
$E_{TOT\,i\,2}$ is the removal coefficient of radionuclide i for secondary treatment (Bq m^{-3} per Bq m^{-3}), as presented in table D12 of Appendix D,
T_2 is the time spent by sewage in secondary treatment (h), as presented in table D2 of Appendix D and others as above.

The activity concentration of radionuclide i in sludge removed at the end of the secondary treatment is:

$$C_{i2s} = (1- E_{TOT\,i\,2}) \cdot Q_{1e}/Q_{2s} \cdot C_{i1e} \cdot e^{-\frac{\ln 2 * T_{2S}}{T_{Hi}}} \qquad \text{Equation 32}$$

where C_{i2s} is the activity concentration of nuclide i in sludge removed during secondary treatment (Bq m^{-3}), $Q_{1e} \sim Q_0$ is the effluent flow rate expressed in t/d, Q_{2s} is the amount of sludge removed (t/d), T_{2S} is the time spent by sludge in secondary treatment (h), see table D2 of Appendix D and others as above.

Some of the secondary sludge (approximately 45 %) is pumped back into the digestion tanks to provide bacterial stock for the next batch.

3.4.2 Exposure pathways

In normal circumstances, there is very little need for direct intervention on contaminated material during the secondary treatment. As for other tanks, some maintenance and repair work must be carried out occasionally and the use of pressure-hoses may result in the re-suspension of some material. In the activated sludge tanks, aeration domes (which produce bubbles) occasionally break and this results in a large amount of sludge being sprayed in the air. This re-suspended material may be inhaled by outdoors workers standing next to these tanks.

Dose calculations will be presented in part 5 of this methodology, using activity concentration C_{i2e} for exposure to the effluent and C_{i2s} for exposure to the solids removed.

3.5 Tertiary treatment of sewage (step 3)

Tertiary treatment may be required depending on the type of water body the effluent is discharged into. Most plants do not have such advanced treatment. There are several types of tertiary treatments that may be carried out, depending on the specific problem at hand (heavy metals, nitrates, BOD, ...). Micro-straining, reed beds and filtration systems are discussed in the main part of this report. Micro-straining and sand filtration systems have to be flushed every so often to gather sedimented solids, which are usually pumped back into the secondary sedimentation tanks. Reeds take up pollutants from the effluent. Tertiary treatment is optional in this model.

3.5.1 Effect on radionuclide concentrations

In the case of micro-straining and filtration systems, activity sorbed on the remaining suspended solids will be retained with the tertiary sludge. In the case of reed beds, metallic radionuclides especially will be taken up and stored by the plants.

The activity concentration in the final effluent after tertiary treatment can be calculated as:

$$C_{i3e} = E_{TOT\,i\,3} \cdot C_{i2e} \cdot e^{-\frac{\ln 2 * T_3}{T_{Hi}}} \qquad \text{Equation 33}$$

where
C_{i3e} is the activity concentration of nuclide i in the effluent after tertiary treatment (Bq m^{-3}),
$E_{TOT\,i\,3}$ is the removal coefficient of tertiary treatment at removing nuclide i from the effluent (Bq m^{-3} per Bq m^{-3}), as presented in table D13 of Appendix D,
T_3 is the time spent by the sewage in tertiary treatment before being discharged (h), as presented in table D2 of Appendix D and all others as above.

The activity concentration of radionuclide i in sludge removed at the end of the tertiary treatment, or taken up by reeds is:

$$C_{i3s} = (1 - E_{TOT\,i\,3}) \cdot Q_{2e}/Q_{3s} \cdot C_{i2e} \cdot e^{-\frac{\ln 2 * T_{3S}}{T_{Hi}}} \qquad \text{Equation 34}$$

where C_{i3s} is the activity concentration of nuclide i in sludge removed during tertiary treatment (Bq m^{-3}), $Q_{2e} \sim Q_0$ is the effluent flow rate expressed in t/d, Q_{3s} is the amount of sludge removed (t/d), T_{3S} is the time spent by sludge in secondary treatment (h), see table D2 of Appendix D and others as above.

There is often very little sludge removed after tertiary treatment of sewage, and it is only removed after a fairly long functioning time (a few years in the case of reed beds).

Very few sewage treatment plants are equipped with tertiary treatment of the effluent at the present time in the UK. By setting T_{3S} and T_3 to zero, and the removal efficiencies to 1, one can effectively consider the secondary effluent as being discharged directly into the receiving water body. In the future, tertiary treatments may well become more common, and data may become available to update this model.

3.5.2 Exposure pathways

Depending on the type of treatment considered there is usually very little manual work to be carried out on a daily basis. There is also very little tertiary sludge produced, as the effluent should have a low suspended solids load before it goes through tertiary treatment. Reed beds can be left to grow for a few years before harvesting the reeds (which may subsequently be incinerated).

Dose calculations for workers will be presented in part 5 of this methodology, using activity concentration C_{i3e} for exposure to the effluent.

Note: when a particular treatment is not applied, in particular tertiary treatment of the effluent, the amount of sludge removed should be set to a very small amount (e.g. 1E-9 t/d) but not to zero so as not incur a 'division by 0' error.

4 SLUDGE TREATMENT AT THE WORKS

Sludges arise at three main stages:
1. from preliminary treatment (solids)
2. from primary treatment
3. from secondary treatment

Sludges from stage 1 go to landfill without further treatment. Sludges from stage 2 and 3 are combined for treatment. A small amount of tertiary sludge may also be mixed with them.

The treatment applied to sludge is dependent upon its final disposal route. Sludges to be disposed of at sea are settled, digested and may be left in lagoons for some time before being transported to the discharge point. Sludges to be incinerated are settled and de-watered before combustion. Sludges to be spread on agricultural land have to be digested. See figure 5 for illustration. This section gives equations for radionuclide concentrations at the end of each stage. Dose calculations are presented in chapter 5.

4.1 Effect of combining sludges on radionuclides concentrations

Sludges from primary and secondary treatments are usually mixed together before treatment. The average activity concentration of nuclide i in the mixed sludge, in Bq m^{-3}, is therefore:

$$C_{iS} = \frac{Q_{1s} \cdot C_{i1s} + Q_{2s} \cdot C_{i2s}}{Q_{1s} + Q_{2s}} \qquad \text{Equation 35}$$

where Q_{1s} is the amount of primary sludge produced per day (m^3 d^{-1}),
C_{i1s} is the activity concentration of nuclide i in primary sludge (Bq m^{-3}),
Q_{2s} is the amount of secondary sludge produced per day (m^3 d^{-1}),
C_{i2s} is the activity concentration of nuclide i in secondary sludge (Bq m^{-3}).

$$F_S = \frac{F_{1s} \cdot Q_{1s} + F_{2s} \cdot Q_{2s}}{Q_s} \qquad \text{Equation 36}$$

where F_S is the suspended solids load of the mixed sludge before any sludge treatment (kg m^{-3} per kg m^{-3}),
F_{1s} is the suspended solids load of the primary sludge (kg m^{-3}),
F_{2s} is the suspended solids load of the secondary sludge (kg m^{-3}),
Q_S is the production rate of mixed sludge: $Q_S = Q_{1s} + Q_{2s}$ (m^3 d^{-1}).
All are presented in tables D15, D23 and D30 of Appendix D.

4.2 Initial sludge processing (step 4)

The initial sludge processing may consist of pasteurisation followed by gravity settling in tanks to achieve a higher suspended solids load in the sludge. In some places, there is no pasteurisation. Some liquor may be removed at the end of the process and pumped back to the head of the works.

$$(1000\text{-}F_S) \cdot Q_S = (1000\text{-}F_{4s}) \cdot Q_{4s} + (1000\text{-}F_{4e}) \cdot Q_{4e} \qquad \text{Equation 37}$$

where F_{4s} is the suspended solids load in the sludge after initial processing (kg m^{-3}),
Q_{4s} is the amount of sludge removed from initial processing per day (m^3 d^{-1}),

$$Q_{4s} = Q_{1s} + Q_{2s}$$

These terms are presented in tables D15, D23 and D30 of Appendix D.

F_{4e} is the suspended solids load in the liquor after initial processing (kg m^{-3}),
Q_{4e} is the liquor flow rate after initial processing (m^3 d^{-1}), since no liquor is removed at either Beckton or Knostrop during initial sludge processing these terms are not included in appendix D.

4.2.1 Effect on radionuclides concentrations

The amount of liquor removed at the end of this process is small both compared to the total amount of sludge and to the main flow of effluent. It is therefore considered that the initial sludge processing has little effect on activity concentrations other than radioactive decay due to the time delay. The activity concentration of radionuclide i in the sludge after initial processing is calculated as:

$$C_{iS4} = C_{iS} . Q_S/Q_{4s} . e^{-\frac{\ln 2 * T_4}{T_{Hi}}} \quad \text{Equation 38}$$

where C_{iS4} is the activity concentration of nuclide i in the sludge after initial processing (Bq m^{-3}),
C_{iS} is the activity concentration of nuclide i in the sludge before any treatment (Bq m^{-3}),
T_4 is the time spent in initial processing (h), see table D2 of Appendix D and
T_{Hi} is the half-life of radionuclide i (h).

The activity concentrations in the liquor will be a negligible addition to the main flow of sewage, and will not be used in workers dose calculations.

The activity concentrations are used for calculating workers' doses. Total activity discharged are necessary to estimate doses to members of the public arising from the discharge of treated sludge:

$$B_{i4} = Q_S . C_{iS} . e^{-\frac{\ln 2 * T_4}{T_{Hi}}} \quad \text{Equation 39}$$

where
B_{i4} is the total amount of radionuclide i in sludge after initial processing (Bq d^{-1}), and others as above.

4.2.2 Exposure pathways

As above, maintenance and repair work may be carried out every so often by specialist workers. When emptied, the settling tanks have to be cleaned out.

Dose calculations for workers will be presented in part 5 of this methodology, using activity concentration C_{i4s} for exposure to the sludge removed.

4.3 Sludge stabilisation (step 5)

Sludge stabilisation may be achieved by thermophilic digestion, mesophilic digestion or composting. Stabilisation is not necessary and even disadvantageous in sludge destined to incineration. It is however required before spreading on agricultural land and usually carried out before disposal at sea or in landfills.

Anaerobic digestion will be considered here as it was found to be the dominant stabilisation treatment in the UK, at 96.6 %. Sludges are pumped to digestion tanks and stored for 20 to 30 days. Once digested, sludges may undergo a further settlement stage in an open tank or lagoon.

4.3.1 Effect on radionuclide concentrations

It will be assumed that digestion has no effect on the radionuclide contents of the sludge. However radioactive decay has a significant impact on activity concentrations in the sludge, and so the activity concentration of nuclide i at the end of sludge stabilisation is:

$$C_{i\,S5} = C_{iS4} \cdot Q_{4s}/Q_{5s} \cdot e^{-\frac{\ln 2 * T_5}{T_{Hi}}} \qquad \text{Equation 40}$$

where
$C_{i\,S5}$ is the activity concentration of nuclide i in the sludge after stabilisation (Bq m^{-3}),
Q_{5s} is the amount of sludge for disposal, at Knostrop this is either $Q_{s\,land}$ or $Q_{s\,agr}$ at Beckton it is $Q_{s\,sea}$ these values are presented in tables D23 and D30 of Appendix D.
T_5 is the total time spent by the sludge in stabilisation (h), including settling after digestion, and is presented in table D2 of Appendix D, and others as above.

$$B_{i5} = B_{i4} \cdot e^{-\frac{\ln 2 * T_5}{T_{Hi}}} \qquad \text{Equation 41}$$

where
B_{i5} is the total amount of radionuclide i in treated sludge (Bq d^{-1}), and others as above.

4.3.2 Exposure pathways

The only occasions when employees of the plant may come close to the sludge is during inspection visits and testing, or once the tanks are emptied, for cleaning and repairing.

Dose calculations for workers will be presented in part 5 of this methodology, using activity concentration C_{i5s} for exposure to the solids removed.

4.4 De-watering and drying of sludge (step 6)

De-watering and drying the sludge are necessary to reach the appropriate suspended solids load to allow ignition during incineration (minimum 30 % solids). There are several methods for de-watering and for drying sludge, as presented in this report. The liquor is usually pumped back to the head of the preliminary sewage treatment. The contribution of the returned liquor to the overall activity concentrations in the raw sewage will be considered negligible.

4.4.1 Effect on radionuclide concentrations

It is assumed that de-watering and drying have little direct effect on radionuclide activity concentrations. The sludge de-watered has not been digested. The activity concentration of nuclide i in the sludge after de-watering and drying is therefore dependent on the activity concentration in the sludge after initial processing (and not after stabilisation):

$$C_{i\,S6} = C_{iS4} \cdot Q_{4s}/Q_{6s} \cdot e^{-\frac{\ln 2 * T_6}{T_{Hi}}} \qquad \text{Equation 42}$$

where
$C_{i\,S6}$ is the activity concentration of nuclide i in the sludge after stabilisation (Bq m^{-3}),
T_6 is the time between the end of the digestion process and the end of the de-watering and drying processes (h), which may include some transport time as well as treatment time, and others as above. The value of T_6 is presented in table D2 of Appendix D.
Q_{6s} is the amount of outgoing sludge after de-watering and drying. It has been calculated using the suspended solids loads in the sludges before and after treatement, F_{4s} and F_{6s}. These values are presented in tables D15, D23 and D30 of appendix D.

$$B_{i6} = B_{i4} \cdot e^{-\frac{\ln 2 * T_6}{T_{Hi}}} \qquad \text{Equation 43}$$

where
B_{i6} is the total amount of radionuclide i in sludge after de-watering and drying (Bq d^{-1}), and others as above.

4.4.2 Exposure pathways

Although the equipment is set up indoors, there may be some re-suspension occurring during this phase, particularly in bad weather and high wind. Otherwise there is very little intervention on the sludge by workers.

Dose calculations for workers will be presented in part 5 of this methodology, using C_{i6s} for exposure to the de-watered sludge.

4.5 Incineration of de-watered sludge (step 7)

The incineration process is becoming increasingly important as the 'disposal to sea' option is being phased out. Radionuclides may be released into the atmosphere under 2 forms: vaporised, as gases, if the incineration temperature is sufficient; and adhering to ash. The remaining ash is usually disposed of in landfills.

4.5.1 Effect on radionuclide concentrations

The total activity released into the atmosphere is the sum of the activity carried by the ash and that emitted with gases.

The amount of activity released to the atmosphere every day from the incineration of sludges is:

$$q_{iF} = V_i \cdot C_{iS6} \cdot Q_{6s} \cdot e^{-\frac{\ln 2 * T_7}{T_{Hi}}} \qquad \text{Equation 44}$$

where
q_{iF} is the amount of nuclide i released per day (Bq d^{-1}),
V_i is the volatised fraction of nuclide i (Bq released / Bq in sludge). A V_i of value 1 means that all the activity is volatised and released into the atmosphere, a V_i of 0 implies that all the activity remains in the sludge. Volatilised fractions are presented in table D14 of Appendix D.

T_7 is the time taken for the sludge to be completely incinerated (h), see table D2 of Appendix D.

The activity concentration of nuclide i in the remaining ash can be calculated as:

$$C_{i\,S7} = (1\text{-}V_i) \,.\, C_{I\,S6} \,.\, Q_{6s}/Q_{ash} \,.\, e^{-\frac{\ln 2 * T_7}{T_{Hi}}} \qquad \text{Equation 45}$$

where
$C_{i\,S7}$ is the activity concentration of nuclide i in the ash after incineration (Bq m^{-3}).
Q_{ash} is the amount of ash removed per day and is presented in tables D23 and D30 of Appendix D.

The total amount of activity disposed with the ash is:

$$B_{i7} = (1\text{-}V_i) \,.\, B_{i6} \,.\, e^{-\frac{\ln 2 * T_7}{T_{Hi}}} \qquad \text{Equation 46}$$

where
B_{i7} is the total amount of radionuclide i remaining in the ash after incineration (Bq d^{-1}).

Best estimate volatilised fractions have been chosen for each of the nuclides in the study from the ranges available in the literature, see notes with table D14 in Appendix D.

4.5.2 Exposure pathways

Workers may be exposed to the activity in the air as well as to large amounts of ash. Dose calculations for workers are treated in part 5 of this methodology. Members of the public may also incur doses from the atmospheric release of radioactivity, these are treated in part 6 (disposal).

Dose calculations for workers will be presented in part 5 of this methodology, using activity concentration C_{i7s} for exposure to the ash removed.

5 DOSES TO WORKERS AT SEWAGE TEATMENT PLANTS

In general, most parts of the sewage treatment plants are automated and do not require extensive manual handling. Furthermore, due to the bacterial and viral risks carried by sewage and sludge, Personal Protective Equipment (PPE) must be worn at all times when working on site. In many cases, the only occasion for an individual to be in contact with contaminated material would be during maintenance cleaning or repair work on parts of the installation.

Two different groups of people were identified as spending significant time at the works: Office workers, who spend most of the time indoors, and Outdoors workers, who may be contractors and who carry out maintenance and repair work.

As for sewage treatment, during normal operations most of the plant is automated and requires very little handling of material. At Beckton STW some workers were identified who would spend all of their time in the sludge press room, close to de-watered sludge. These workers were considered separately.

The number of hours spent exposed to each type of contaminated material by workers has been identified. The total dose for each pathway is calculated multiplying the number of hours spent close to or handling each type of sewage or effluent by the dose rate for each material.

5.1 Individual doses at the works

External irradiation
The individual dose from external irradiation is the sum of the dose from exposure to each type of material. Each external dose is the product of the dose rate corresponding to the situation considered by the activity concentration of the medium by the number of hours spent by a typical worker close to this source. Exposures during sewage treatment and sludge treatment have been considered separately:

$$_{Ext}D_{i,\,w} = H_{i,\,ext,\,w} \cdot \left(\sum_{j,k} T_{w,j,k} \cdot C_{i,j,k} + \sum_{j} T_{w,j} \cdot C_{i,S,j} \right) \qquad \text{Equation 47}$$

where $_{Ext}D_{i,\,w}$ is the individual external dose incurred by workers at the sewage treatment plant (Sv y^{-1}),
$H_{i,\,ext,\,w}$ is the external dose rate from standing 1m from sewage or sludge (Sv h^{-1} per Bq m^{-3}), as presented in table D5 of Appendix D. The dose rate assumes exposure to an infinite slab with the operator above the source.
$T_{w,\,j,\,k}$ is the number of hours per year spent next to effluent (k = e) or sludge (k = s) of activity concentration $C_{i,\,j,\,k}$ (h y^{-1}), during the sewage treatment processes (j = 0 to 3),
$T_{w,\,j}$ is the number of hours per year spent next to the sludge of activity concentration $C_{i,\,S,\,j}$ (h y^{-1}), during sludge treatment processes (j = 4 to 7 - includes ash) and i relates to nuclide i. Occupancy times are presented in table D16 of Appendix D.

Inhalation of re-suspended material
Similarly for the dose arising from the inhalation of re-suspended material:

$$_{inh}D_{i,\,w} = Brh \cdot H_{i,\,inh} \cdot \left(\sum_{j,k} T_{w,j,k} \cdot \frac{C_{i,j,k}}{\rho_e} \cdot L_{j,\,k} + \sum_{j} T_{w,j} \cdot \frac{C_{i,S,j}}{\rho_S} \cdot L_{S,\,j} \right) \qquad \text{Equation 48}$$

where $_{inh}D_{i,\,w}$ is the individual inhalation dose incurred by workers at the sewage treatment plant (Sv y^{-1}),
$L_{j,\,k}$ is the suspended solids in air concentration of activity concentration $C_{i,\,j,\,k}$ (kg m^{-3}), during sewage treatment processes (j = 0 to 3),
$L_{S,\,j}$ is the suspended solids in air concentration of activity concentration $C_{i,\,S,\,k}$ (kg m^{-3}) during the sludge treatment processes (j = 4 to 7), as presented in table D7 of Appendix D.
Brh is the breathing rate of a typical worker, see table D8 of Appendix D.
$H_{i,\,inh}$ is the inhalation dose coefficient (Sv Bq^{-1}), see table D6 of Appendix D.

ρ_e is the density of effluent, approximately equal to that of water (ρ_e = 1000 kg m^{-3}), and ρ_S is the density of sludge, assumed to be equal to that of water except for de-watered sludge (30 % suspended solids) and ash.

The suspended solids in air concentration may vary widely from one process to another. For example, near activated sludge digestion tanks, there may be a lot of spraying, and the presence of ash is likely to create a dusty environment, whereas during the sedimentation stages, there is very little splashing or spraying in quiet weather conditions.

Inadvertent ingestion
Similarly, the dose arising from the inadvertent ingestion of sewage or sludge can be calculated as:

$$_{ing}D_{i,\ w} = Ing\ .\ H_{i,\ ing,}\ .\ (\sum_{j,k} T_{w,j,k} \cdot \frac{C_{i,j,k}}{\rho_e} + \sum_{j} T_{w,j} \cdot \frac{C_{i,S,k}}{\rho_S}) \qquad \text{Equation 49}$$

where $_{ing}D_{i,\ w}$ is the individual inadvertent ingestion dose from nuclide i (Sv y^{-1}),
$H_{i,\ ing}$ is the ingestion dose coefficient (Sv Bq^{-1}),
Ing is the inadvertent ingestion rate, which is likely to be approximately constant whatever the material handled, and all others as above.

Total annual dose
For a typical worker, that total dose is the sum of doses over all nuclides and all pathways.

$$D_{ind,\ w} = \sum_{i=1}^{10} (_{ext} D_{i,w} +_{inh} D_{i,w} +_{ing} D_{i,w})) \qquad \text{Equation 50}$$

where $D_{ind,\ w}$ is the annual individual effective dose (Sv y^{-1}) for a typical worker,
10 is the total number of radionuclides considered, and all others as above.

5.2 Collective doses at the works

The collective doses to workers will be calculated from the individual annual doses and the number of workers in the catchment area:

$$_xD_{i,\ coll,\ w} = {}_xD_{i,\ w}\ .\ n_w \qquad \text{Equation 51}$$

where $_xD_{i,coll,\ w}$ is the overall collective dose for nuclide i and pathway x (x = ext, inh, ing) at the works (man Sv y^{-1}),
$_xD_{i,\ w}$ is the individual annual dose from pathway x (Sv y^{-1}),
and n_w is the number of workers at the works (man), as presented in tables D19 and D26 of Appendix D.

The total annual collective dose is:

$$D_{coll,\ w} = D_{ind,\ w}\ .\ n_w \qquad \text{Equation 52}$$

where $D_{coll,\ w}$ is the total annual collective dose at the works (man Sv y^{-1}), and all others as above.

6 DISPOSAL OF TREATED EFFLUENT AND SLUDGES

Both the final effluent and sludge are eventually disposed of. The sludge may be disposed of by 4 main routes: by disposal at sea, by incineration, by disposal to landfills and by spreading on agricultural land. Each of these routes may result in the exposure of members of the public. Critical groups will be identified for each disposal option and doses calculated using the appropriate models.

Doses to the public are calculated for members of the relevant critical groups, for average individuals and for the whole UK population. The calculations were performed for unit disposal and then used as a database in the model, to be scaled by the total activity discharged. The models used for calculating doses from unit disposal are presented in Appendix C.

6.1 Disposal of treated effluent

The treated effluent is usually disposed of in rivers or estuaries. Doses per unit discharge of each nuclide were calculated using standard models for 1st year of release and for 50th year of release.

The total activity discharged in effluent can be calculated from

$$B_{i\ out\ e} = C_{i\ 3\ e} \cdot 365 \cdot Q_{out\ e} \qquad \text{Equation 53}$$

where $B_{i\ out\ e}$ is the activity of nuclide i released per year in effluent (Bq y^{-1}),
$C_{i\ 3\ e}$ is the activity concentration of nuclide i in the final effluent (Bq m^{-3}), $C_{i\ 3\ e}$ may be equal to $C_{i\ 2\ e}$ where there is no tertiary treatment, and
$Q_{out\ e}$ is the effluent flow rate out of the works (m^3 d^{-1}), see tables D23 and D30 of Appendix D.

The exposures considered are described in Appendix C.

6.2 Disposal of sludge at sea

The total activity disposed per year in sludge can be estimated from:

If all the sludge from one site is disposed of at sea, the total activity disposed is:

$$B_{i\ S\ sea} = 365 \cdot B_{i5} \qquad \text{Equation 54}$$

where $B_{i\ S\ sea}$ is the activity of nuclide i disposed per year in sludge (Bq y^{-1}), and other as above.

If only part of the sludge is disposed of in the sea, then the total activity disposed per year in sludge can be estimated from:

$$B_{i\,S\,sea} = C_{i\,S\,5} \cdot 365 \cdot Q_{S\,sea} \qquad \text{Equation 55}$$

where $B_{i\,S\,sea}$ is the activity of nuclide i disposed per year in sludge (Bq y^{-1}),
$C_{i\,S\,5}$ is the activity concentration of nuclide i in the sludge considered (Bq m^{-3}) and
$Q_{S\,sea}$ is the amount of sludge disposed to sea per year (m^3 d^{-1}), see table 23 of Appendix D, (disposal at sea is not an option at Knostrop STW).

The exposures considered are described in Appendix C.

6.3 Disposal of sludge and ash in landfills

If all the sludge and/or ash from one site is disposed of in landfills, the total activity disposed is:

$$B_{i\,S\,land} = 365 \cdot B_{i5} + 365 \cdot B_{i7} \qquad \text{Equation 56}$$

where $B_{iS\,land}$ is the activity of nuclide i disposed per year in sludge (Bq y^{-1}), and other as above.

If only part of the sludge is disposed of in landfills, then the total activity disposed per year in sludge can be estimated from:

$$B_{i\,S\,land} = C_{i\,S\,5} \cdot 365 \cdot Q_{S\,land} + 365 \cdot B_{i7} \qquad \text{Equation 57}$$

where
$Q_{S\,land}$ is the amount of sludge disposed to landfills per year (m^3 y^{-1}), see tables D23 and D30 of Appendix D and others as defined in 4.5.1.

Total annual doses will then be calculated by summing over all nuclides.

6.4 Disposal of sludge on agricultural land

If all the sludge from one site is disposed of on agricultural land, the total activity disposed is:

$$B_{i\,S\,agr} = 365 \cdot B_{i5} \qquad \text{Equation 58}$$

where $B_{i\,S\,agr}$ is the activity of nuclide i disposed per year in sludge (Bq y^{-1}), and other as above.

If only part of the sludge is disposed of on agricultural land, then the total activity disposed per year in sludge can be estimated from:

$$B_{i\,S\,agr} = C_{i\,S\,5} \cdot 365 \cdot Q_{S\,agr} \qquad \text{Equation 59}$$

where
$C_{i\,S\,5}$ is the activity concentration of nuclide i in the sludge considered (Bq m^{-3}) and
$Q_{S\,agr}$ is the amount of sludge spread on agricultural land per year (m^3 d^{-1}), see table D23 and D30 of Appendix D.

Exposures considered are described in Appendix C.

6.5 Atmospheric discharge from incineration of sludges

The incineration process is fairly automated and the most important pathways of exposure to the public are likely to be linked to the gaseous and aerosols emissions from the stack, and the disposal of the ash. This affects both workers on-site and members of the public off-site.

Exposures considered are described in Appendix C.

6.6 Landfill disposal of sludges

Landfills may receive digested sludge itself or residual ash from incineration. Radionuclides in the sludge or ash may gradually migrate out of the site over 10 or more years and emerge into the biosphere. All radionuclides except carbon-14, tritium and americium-241 would have decayed away before migrating out of the site. The exposures considered from tritium, carbon-14 and americium-241 are described in Appendix C.

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APPENDIX C

MODELLING THE POST DISPOSAL RADIOLOGICAL IMPACT

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1. INTRODUCTION

During assessment of the radiological consequences of disposal of radionuclides to sewers it is necessary to consider both operational exposures during the management of contaminated sewage, and the exposures of members of the public arising from the radionuclides when they are disposed of or discharged into the environment. This Appendix describes the methodology used to make estimates of the dose to members of the public from disposals or discharges. Operational exposures are described in Appendices B and D. The methodologies described below have all been used to calculate doses arising from a discharge of unit activity. These doses may be scaled and summed together to provide an estimate for any disposal of interest. The routes of disposal that have been considered are:

(a) Treated effluent released into an estuary
(b) Treated effluent released into a river
(c) Atmospheric discharge following incineration of sludge
(d) Off-shore dispersal of sludge disposed at sea
(e) Disposal of sewage sludge or ash into a landfill
(f) The application of sewage sludge onto agricultural land

The models and general data used for each is described in turn in the following sections. In addition, data specific to the Beckton and Knostrop sewage treatment works in east London and Leeds respectively have been described. Individual and collective doses from disposal of seventeen radionuclides with effluent and sludges at a unit rate of 1GBq y^{-1} (except sewage sludge applied to land) have been calculated: H-3, C-14, P-32, S-35, Cr-51, Ga-67, Rb-83, Rb-84, Sr-89, Y-90, Tc-99m, In-111, I-123, I-125, I-131, Tl-201 and Am-241. For sewage sludge applied to land doses are expressed per Bq l^{-1} of sludge at 5% solid content.

2. DOSES CALCULATED

For each type of disposal or discharge, individual and collective doses assuming 1 and 50 years worth of disposals have been calculated, for a disposal a rate of 1 GBq y^{-1} of each radionuclide.

The individual doses calculated are the peak annual effective dose to members of hypothetical critical groups. The critical groups vary depending on the disposal route. For disposals to coastal waters and estuaries, a critical group consuming mollusca, crustacea and fish at critical levels (97.5th percentile consumption rates) caught in the location of the disposal has been assumed. This group is also assumed to have high beach occupancies. For disposals to rivers, the critical group is assumed to consume freshwater fish at critical levels from the river section and to take all drinking water from the same section. High river bank occupancies are also assumed. For incineration of sludges two exposed groups have been defined, one representing the nearest resident to the incinerator, the other the nearest farm to the incinerator. The nearest resident is assumed to grow sufficient green and root vegetables and fruit to provide 20% of critical intake, whilst other foods are assumed to be sourced elsewhere. The nearest farmer is assumed to produce sufficient of all food types to provide for a family consuming all foods at critical intakes.

Collective doses have been truncated at 500 years and are presented for the United Kingdom (UK) for atmospheric releases and application of sludge to farmland, and UK, European Union and world populations for other discharges or disposal routes.

3. DOSES FROM DISPOSAL OF EFFLUENT TO ESTUARY

3.1 Introduction

The doses from disposal of treated effluents into an estuary have been calculated using BIOS, a multi compartmental biosphere transport model to calculate the dose to man due to release of radionuclides into the environment from the disposal of radioactive waste. The model includes transfer through the marine environment, into marine foods and sediment, and sea to land transport of radionuclides and subsequent transfer into crops and animal products. Element dependent equilibrium transfer factors are used to represent the transfer of radionuclides through the marine and terrestrial environment. Some general data used in the model have been presented in Table C1.

3.2 Local marine compartment

Releases to the marine environment are modelled by creating a local marine compartment which sits within a regional marine compartment. The characteristics of the local compartment are determined using methods similar to that described in a study of the dumping of radionuclides in coastal waters, using appropriate data from other publications[4,5].

The volume of the local compartment can be estimated from the area and average depth. Exchange rates (fluxes) between the local compartment and regional compartment were calculated from tidal flow data and the cross sectional area of the local box. The average tidal flow rate was used and two tides per day assumed. The suspended sediment load and sedimentation rates were assumed to be the same as the regional marine compartment, and seafood catches from the regional compartment were scaled by the ratio of the regional compartment area to the local compartment area.

3.3 Data for Beckton

For the discharges of effluent from Beckton sewage works it has been assumed that the estuary compartment is the Thames tideway. This has been assumed to be the canalised tidal Thames between Teddington Lock and Shoeburyness. The parameter values describing the compartment have been presented in Table C2, and are described below. The mean depth of the section is 7.5 m, and the average width of the river is 194 m. The distance from Teddington Lock to the Shoeburyness is 96 km. Therefore the section has an area of $1.86\ 10^7$ m^2 and a volume of $1.4\ 10^8$ m^3.

The cross sectional area of the Thames was derived from the average width and depth of the channel (196 m x 7.5 m), giving a cross-sectional area of $1.47\ 10^3$ m^2. Taking a typical tidal flow rate (point J, reference), the estimated flux into the compartment is $1.218\ 10^{10}$ $m^3\ y^{-1}$. This estimate of flux suggests that the Thames from Shoeburyness to Teddington is flushed by tidal

water 87 times a year, just over three times a fortnight. Suspended sediment loads and the sedimentation rate have been assumed to have the same values as described in reference .

Individual doses to the critical group from the release of radionuclides into the estuary have been estimated. The doses assume that crustacea, mollusca and fish are caught locally and consumed at critical rates (97.5th percentile[1], Table C3) and that the exposed group has high beach occupancy. Sea to land transfer of radionuclides and exposure from the associated pathways is also considered. Table C4 presents the individual effective doses per unit disposal and the three most significant exposure pathways.

Collective doses were calculated using total catches of marine foods from the local compartment, shoreline occupancies and yields of terrestrial foods produced on the coastal strip of land, contaminated by sea to land transport of seaspray. The catches in the local compartment were estimated from the ratio of the area of the local compartment to regional compartment and regional compartment catches. The area of the tidal Thames local box is $1.86\ 10^{7}$ m^2, whilst the total area of the regional compartment (North Sea South West) is $1.45\ 10^{10}$ m^2. Therefore a fraction of $1.28\ 10^{-3}$ of the regional seafood catch has been allocated to the local box (4.62 t y^{-1} of fish, 0.552 t y^{-1} of crustacea and 10.4 t y^{-1} of mollusca, Table C2). The total length of coastline is approximately 192 km. Coastal occupancies of 50 man h y^{-1} m^{-1} were used, the collective occupancy of $9.6\ 10^{6}$ man h y^{-1} was used in the collective dose calculation. Table C5 shows the collective doses to the local population, UK, EU and world population from the discharges.

4 TREATED EFFLUENT TO RIVER

4.1 Introduction

The doses from disposal of treated effluents into rivers have been calculated using the biosphere transport model, BIOS[2]. For disposals of radionuclides to river, the model calculates doses from drinking river water, consumption of freshwater fish and external doses from river bank sediment. Doses from the ingestion of crops irrigated with river water were also modelled. Element dependant equilibrium transfer and concentration factors are used to predict the concentration of radionuclides in parts of the aquatic and terrestrial environment.

4.2 River model

The river water compartment considers activity dissolved in water plus that sorbed to sediments. Each river water compartment has three associated sediment compartments. The top layer of river bed sediment moves with river flow via viscous drag. Modelling is carried out assuming the river is divided up into successive downstream compartments linked together by water and sediment flow. The transfer of activity between compartments is modelled using the water velocity, bed sediment velocity and length of each compartment. The river compartments are defined by identifying the main parameters that describe the river section receiving the effluent, e.g. water velocity, bed sediment velocity, suspended sediment load, water depth, length of section, volume of water and area of farmed land either side of the river section.

4.3 Data used in this study

Treated effluent may be disposed into inland rivers. The river Aire in Yorkshire receives treated effluents from Knostrop sewage works, and the river Thames for example receives treated effluents from sewage works at Swindon, Oxford and Reading, and Thames tributary rivers receive treated effluent from a number of Greater London sewage treatment works.

The river model was set up with two river sections, one of length 500 m and with volumetric flow of 1 m^3 s^{-1} and a second downstream section of length 10 km and volumetric flow of 10 m^3 s^{-1}. The highest individual doses are predicted from the first section, whilst the collective dose was taken from the second section.

Individual doses were calculated assuming that freshwater fish were caught and consumed at critical rates from the first compartment (Table C3) and that drinking water for the critical group was sourced from the same compartment. It was assumed that the critical group spent time on the river bank. Ingestion of terrestrial foods irrigated with river water was included as appropriate. The parameters assumed for the generic river are given in Table C6 and the individual doses per unit disposal are given in Table C7.

Collective doses were calculated assuming that 0.1 and 10 t y^{-1} of fish were caught in the two river sections, that a fixed fraction of the river water was taken for drinking water and that a 100 m section of the river bank on either side was irrigated with river water. These parameters are shown in Table C6. The collective dose estimates per unit disposal are given in Table C8.

5 ATMOSPHERIC DISCHARGES FROM INCINERATOR

5.1 Introduction

Atmospheric discharges from sewage sludge incinerators have been assessed using the PC-Cream methodology developed by the European Commission (EC)[6]. PC-Cream is a suite of models, implemented as PC software, designed to preform radiological assessments of routine discharges of radionuclides. For atmospheric releases, the following exposure pathways are considered.

(a) Inhalation of radionuclides in the plume
(b) External irradiation by the plume
(c) Inhalation of resuspended radionuclides, integrated to a defined period
(d) External irradiation from deposited radionuclides, integrated to a defined period
(e) Ingestion of foodstuffs grown on land contaminated by deposited radionuclides.

The dispersion model and exposure pathway models are described in the following sections. Dispersion modelling specifically took into account the rise of the hot exhaust gases.

A final section describes the application of all models to the releases from the Beckton incinerator.

5.2 Plume rise model

The release of hot gases causes buoyancy of gases relative to the atmosphere, and plume rise must be considered. This can then be incorporated into dispersion models as an effective stack height from which the release takes place, (where the effective stack height is the stack height plus the plume rise).

It is known that the incinerator discharges gases at a high temperature. This data may be used to estimate the plume rise (Δh), using an equation derived by Briggs, and described in reference 6:

$$\Delta h \simeq 2.6 (F_B / [u_A \times s])^{1/3}$$

Where F_B = The initial buoyancy flux for the plume ($m^4 s^{-3}$)
u_A = The average windspeed through the height which the plume has risen ($m s^{-1}$), assumed to be 5 $m s^{-1}$
s = The stability parameter (s^{-2})

The initial buoyancy flux is determined by the heat emission rate, W (in Watts), calculated using:

$$W = c_P \times \Delta T \times v_D \times \rho_D$$

Where c_P = The specific heat of air at constant pressure, taken to be 1005 $J kg^{-1} K^{-1}$
ΔT = The temperature increase of the emissions relative to ambient temperature, K
v_D = The volumetric emission rate, $m^3 s^{-1}$
ρ_A = The density of the emissions, assumed to be 1.293 $kg m^{-3}$

The total heat emission rate from the stack is therefore 0.65 MW. This can be seen to be reasonable in comparison to the total power output of the incinerator, 9.5 MW. The initial buoyancy flux is now calculated using:

$$F_B = g \times W / (\pi \times c_P \times \rho_A \times T_A)$$

Where g = The acceleration due to gravity, 9.81 $m s^{-2}$
ρ_A = The density of air, assumed to be 1.293 $kg m^{-3}$
T_A = The temperature of the atmosphere, assumed to be 273 K

Finally, the stability parameter s requires calculation. This value of s is calculated using:

$$s = g \times \gamma / (s_2 \times \theta)$$

Where γ = The temperature gradient of the atmosphere, assumed to be 0.01 $K m^{-1}$
s_2 = A dimensionless parameter with value 2.3
θ = The potential temperature of the atmosphere (K). This value is the temperature that a parcel of air at temperature T_G would attain if brought adiabatically to a pressure of one standard atmosphere. It has been assumed that θ = 273 K

5.3 Dispersion and pathway models

A Gaussian plume atmospheric dispersion model which uses a source depletion model, and considers wet deposition using a washout coefficient was used to model dispersion from the incinerator stack. The model estimates air concentrations and deposition rates at a point for specified meteorological and terrain conditions. External irradiation from passing radionuclides in the plume is calculated using a finite cloud model for gamma irradiation and an infinite cloud model for beta irradiation.

For deposited radionuclides, three models were used to estimate the doses from deposited activity, (resuspension, 'RESUS', external gamma dose 'GRANIS' and food concentrations 'FARMLAND'). These three models are also described in reference 7. RESUS calculates integrated air concentrations of resuspended deposited activity, and is based on a semi-empirical model, as described in reference 7. GRANIS calculates doses from external irradiation from deposited radionuclides. It uses a soil model to estimate the migration of radionuclides through the soil column over time for either undisturbed or well mixed soil (i.e. either pasture or ploughed soil). The dose rate 1 m above the soil surface is then calculated taking into account the attenuation and build up properties of the soil. FARMLAND considers the transfer of radionuclides into foodstuffs. The model is dynamic and represents soils, crops and animal products by a series of compartmental models linked by transfer coefficients which describe key processes such as interception, translocation, downward migration, root uptake and processing losses.

5.4 Data for Beckton and Knostrop

The models have been used to assess the discharges from Beckton and Knostrop incinerators using the data described in this section. This data is broadly of two types: data describing the characteristics of the release of the radionuclides and their movement, and data describing the behaviour of people leading to exposure to these radionuclides ('habit data').

5.4.1 Atmospheric release data for Beckton and Knostrop

For atmospheric dispersion calculations the following assumptions have been modelled. The gas is assumed to be emitted at a temperature of 120°C and a rate of 5 m^3 s^{-1}. These data indicate a plume rise of about 50 m. The physical height of the Beckton stack is about 50 m and the Knostrop stack is about 40 m. Therefore the effective release height for Beckton and Knostrop were assumed to be 100 m and 90 m respectively. A uniform windrose has been assumed and the meteorological conditions have been chosen to be appropriate for the areas of the country. Therefore it has been assumed that neutral conditions (Pasquill stability class D) occur 60% of the time. The roughness length of the surrounding area has been assumed to be 1 m. This is the lower end of the range suitable for urban conditions (1 to 3 m), but reflects the mainly low rise buildings and derelict land in the vicinity of the incinerator. These data have been summarised in Table C9.

5.4.2 Habit data for Beckton and Knostrop

The characteristics of the most exposed groups have been based on habit surveys for the UK population[1]. The assumptions have been summarised in Table C10. Two exposed groups were considered. The first group represents residents living at a distance from the incinerator where ground level air concentrations were maximum and where housing was located. This was around 1.4 km from both the Knostrop and Beckton stacks. It was assumed that the resident spends all their time at that location, half of this outside, and obtains 20% of their intake of root vegetables, green vegetables and fruit from a back garden or allotment. The consumers are assumed to eat these locally produced foods at critical rates.

The second exposed group is located at the nearest farm. The nearest farm is 13 km from the Beckton stack and 5 km from the Knostrop stack. Here an adult is assumed to spend half of their time outside at that location. Each farm is assumed to produce 100% of all foods to support a family, and all foods are assumed to be consumed at critical rates. Collective doses from 1 y and 50 y releases truncated at 500 y have been calculated using UK population and agricultural grids centred on the Beckton and Knostrop plants.

5.5 Results for Beckton and Knostrop

The estimates of annual effective individual doses to exposed groups around Beckton are presented in Table C11, and for exposed groups around Knostrop in Table C12. In both cases the resident at 1.4 km is predicted to receive a higher individual doses than the farmer at 5 and 13 km, and is indicative of the critical group. Collective doses from incineration at Knostrop and Beckton are given in Table C13.

6 DISPOSAL OF SLUDGE OFF-SHORE

6.1 Introduction

The doses from the disposal of sludge by dispersion in coastal waters have also been calculated using the BIOS model[2].

6.2 Coastal marine model

A section of coastal water into which sludge may be dumped has been represented by an appropriate local marine compartment which sits within a predetermined regional compartment. The approach to setting up the local compartment is as described in section 3.2, and involves calculating the area, cross sectional area, volume and volumetric flow of water across the sections and total marine food catches.

6.3 Data for Beckton

Until the end of 1998, the sewage sludge generated at Beckton was disposed of at Barrow Deep, an area of coastal waters to the east of Foulness island. This was modelled by defining a local compartment bounded to the east by Sheerness, Shoeburyness, and the Essex Coastline

between the Thames Estuary and the river Blackwater, to the north by West Mersea, Clacton on Sea and Walton on the Naze and the to south by the North Kent Coast, as far east as Margate. The area approximately triangular with sides of 60 km (Shoeburyness to The Naze) and 49 km (Sheerness to Margate). The line joining Margate to the Naze is about 55 km, giving a total area of 1,265 km^2. The area includes mudflats, sand bars and trenches, and ranges from 1 m to 20 m deep. The average depth has been assumed to be 7 m. These values yield an overall volume of 8.86 10^9 m^3.

The flux of water across the defined local compartment was calculated from the tidal flow (along an east-west line) at the eastern edge of the compartment (55 km from 'the Naze' to the North of the compartment to Margate at the southern end). The depth along this section has been taken as 7 m. The cross sectional area is therefore 55 km × 7 m, ie. 3.85 10^5 m^2. The average flow rate into the compartment, over six hours, considering the spring and neap tides is 1.02 knots, or about 1890 m h^{-1}. This has been assumed to flow for 12 hours a day. This yields a flux of 8.73 10^9 m^3 d^{-1} or 3.19 10^{12} m^3 y^{-1}, equivalent to approximately 360 flushes of the local compartment a year or 1 flush per day. Suspended sediment loads and the sedimentation rate have been assumed to have the same values as the regional compartment. The characteristics of the local compartment are given in Table C14. The intake rates assumed for marine foods are given in Table C3.

Individual doses per GBq y^{-1} were calculated from the disposal of radionuclides in the sludges, estimates have been made for 1 year and 50 years of disposal. These are given in Table C15.

Collective doses were calculated using total catches of marine foods from the local compartment, shoreline occupancies and yields of terrestrial foods produced on the coastal strip of land. The catches in the local compartment were estimated from the ratio of the area of the local compartment to regional compartment and regional compartment catches. The local compartment has area 1.27 10^9 m^2, and represents 8.76% of the area of the regional compartment (North Sea South West, 1.45 10^{10} m^2). The seafood catches are therefore 315 t y^{-1} of fish, 37.7 t y^{-1} of crustacea and 709 t y^{-1} of mollusca. The total length of coastline is approximately 110 km. Assuming a coastal occupancy of 50 man h y^{-1} m^{-1}, the collective beach occupancy is 5.5 10^6 man h y^{-1} (Table C14).

Estimated collective doses per GBq y^{-1} from disposal of each radionuclide into coastal waters with sludges for 1 year and 50 years are presented in Table C16.

7 DISPOSAL OF SLUDGE AND ASH TO LANDFILL

7.1 Introduction

A proportion of sewage sludge produced in the UK is disposed of directly to landfill. In addition ash and residues from incineration may also be disposed. The doses from disposal of sludges and ash to landfill were estimated assuming that activity migrates out of the site over time.

7.2 Landfill site modelled

Reference reviewed the characteristics of a range of landfills. This work also determined typical parameters for landfills, deriving four generic types. The ash or sludge has been assumed to be disposed to the most basic, a 'dilute and disperse' landfill. Such a landfill is often used for inert industrial wastes that do not contain significant quantities of pollutants such as heavy metals.

The generic landfill modelled has no lining and limited capping (0.5 m of soil), meaning that water may enter the landfill relatively easily. There is good evidence that such landfills usually cover quite a substantial area (*eg.* 100 ha) as the trenches are rarely deep (more than about 5 m). The landfill has therefore been assumed to have a capacity of 5 10^6 m^3, to cover an area of 100 ha and have an average depth of 5 m. The exposed group are assumed to eat all terrestrial foodstuffs at mean consumer rates, produced in the area where radionuclides emerge into the biosphere from the landfill, a fast the ground water return time (95th percentile). The individual and collective doses per unit disposal are taken from reference 8.

7.3 Data used in this study

The methodology described in reference 8 has been used to estimate individual and collective doses from the disposal of radionuclides in sewage sludge to landfill. It was found that only tritium, carbon-14 and americium-241 gave rise to individual doses from landfill disposal and migration. Other radionuclides have half lives that were very short (< 6 months) in relation to the time which is taken for the radionuclide to leach from the waste, migrate through the soil and rock and enter the biosphere (>10 years). Individual and collective doses (per GBq y^{-1}) are presented in Table C17.

8 APPLICATION OF SLUDGE TO FARMLAND

8.1 Introduction

Sewage sludge can be applied to land as a soil conditioner. This is fairly common practice in areas where suitable agricultural land is available. This gives rise to possible exposure routes to man and therefore needs to be considered in the calculation of doses from sewage works. Sewage sludge may be applied to many types of agricultural land. However, controls are placed on the use of sludge on land used to produce crops that may be eaten raw, especially, fruit and vegetables. In these cases a time delay of 10 months between application of the sludge and harvesting of the crop is required. Further control is exercised by requiring the metal concentrations in the soil to be assessed before application of sludge is permitted. If levels are above certain limits application of sludge is not permitted. These controls mean that once suitable land is located, sewage sludge may be applied to it over a number of years. The assessment was therefore performed taking these factors into account. The dynamic food chain model Farmland was used to model the transfer of activity from the applied sludge into food products.

The application of sludge to land occurs where suitable land is available and transporting sewage sludge to the land is feasible. Sewage works on the outer edge of urban areas which fringe agricultural land are more likely to dispose of their sewage sludge to land. Sewage works

located in urban areas where agricultural land is not readily available use alternative disposal routes, either disposing to sea or incinerating. In Greater London, sewage works at Mogden, Long Reach, Maple Lodge, Beddington, Hogsmill, and Esher send sludge to land. Sewage works at Beckton, Crossness and Deephams either dispose to sea or incinerate sludge.

8.2 Modelling approach

In this work, the application of sewage sludge to land used for cattle meat, cattle offal, milk, sheep meat and sheep offal was considered. The radiological impact from cereals produced on conditioned soil was also calculated although cereals are not normally consumed locally. Therefore the inclusion of doses from cereals is conservative. Doses from fruit and vegetables produced on conditioned soil were not considered because of the controls placed on the use of sewage sludge to treat soil producing foods that may be eaten raw or unprocessed.

Animal products and cereals were the only foods considered. It should be noted that in the eastern parts of the UK, including the areas around London, the majority of farmland is arable. Sheep, beef and dairy farming occurs more to the west of the UK. This will need to be considered when the assessment results from animal products are included.

It was assumed that the sewage sludge was subjected to some treatment (pasteurisation or digestion) with processing times of around 30 days before application to land. In addition, sludges are only applied to land once or twice a year, this implies storage for an average of 6 months before application. Radionuclides with short half lives (< 15 days) would undergo significant radioactive decay during this time. Therefore, the modelling of radionuclide application to land in sewage sludge was not considered for radionuclides with a short half life.

For radionuclides applied in sewage sludge (except C-14 and H-3), the models used to estimate the doses from deposited activity were GRANIS (external gamma dose) and FARMLAND (food concentrations). These models are described in references 7 and 10. GRANIS calculates doses from external irradiation from deposited radionuclides. It uses a soil model to predict concentrations in either undisturbed or well mixed soil (i.e. either pasture or ploughed soil) after migration of radionuclides through the soil column over time. The dose rate 1 m above the soil surface is then calculated taking into account the attenuation and build up properties of the soil. FARMLAND[10] models the transfer of radionuclides into foodstuffs. The model is dynamic and represents soils, crops and animal products by a series of compartmental models linked by transfer coefficients which describe key processes such as downward migration, root uptake and processing losses. Interception and translocation processes were not modelled for sewage sludge application because direct deposition onto grass or crops does not occur (see section 8.3). For H-3, the model TRIF was used to predict soil and plant concentrations, whilst for C-14 the concentrations in food crops and soil were calculated using the model BIOS[2].

8.3 Data used in this study

Conditioning of pasture was assumed to occur annually in early spring, approximately one month before grazing animals were allowed onto the pasture. It was further assumed that the grass grew after the application and therefore there was no direct contamination of the grass by radionuclides in the sludge at the time when grazing of the pasture commenced. Conditioning of land used for cereals was assumed to occur before seeding. It was assumed that no direct

contamination of the plant by activity in sewage sludge occurred. Sludges were assumed to be 5% solid and 95% water at the time of application. The application rate for sewage sludge to land used was 8 kg m^{-2}.

Two sets of individual doses to a hypothetical farmer were calculated from ingestion of beef, milk, cow offal, sheep meat, sheep offal and cereal. One set assumed that the farmer produced all his own food and consumed them all at critical rates. The second set assumed the farmer consumed all foods, at mean consumer rates (cereals not consumed at all). External doses from the conditioned soil and doses from ingestion and inhalation of the soil were included. Individual doses for each nuclide expressed per Bq per kg^{-1} are given in Table C18.

Collective doses were calculated to the UK population from the total production of foods produced on the land treated with sludge. Collective dose is dependant on the activity concentration in the sludge and the size of the works, which influences the quantity of sludge produced, which in turn affects the area of land that can be treated and the total production of food. Collective doses expressed per Bq kg^{-1} in sludge per kg of sludge production are given in Table C19.

9 REFERENCES

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Table C1 Agricultural data for biosphere model

Parameter	Value
Fraction of land used for cattle	$7.30\ 10^{-2}$
Fraction of land used for sheep	$1.38\ 10^{-1}$
Fraction of land used for green vegetables	$5.00\ 10^{-2}$
Fraction of land used for root vegetables	$2.00\ 10^{-2}$
Fraction of land used for grain	$7.09\ 10^{-1}$
Fraction of land used for poultry	$1.00\ 10^{-2}$
Stocking density of cattle (km^{-2})	$4\ 10^{2}$
Stocking density of sheep (km^{-2})	$5\ 10^{2}$
Stocking density of poultry (km^{-2})	$3\ 10^{4}$
Yield of green vegetables (kg km^{-2})	$1\ 10^{6}$
Yield of grain (kg km^{-2})	$4\ 10^{5}$
Yield of root vegetables (kg km^{-2})	$3\ 10^{6}$
Yield of pasture (kg km^{-2})	$5\ 10^{5}$

Table C2 Local marine compartment for tidal Thames

Parameter	Value
Volume of Local Box (m^3)	$1.4\ 10^{8}$
Exchange Rate with Marine compartment ($m^3\ y^{-1}$)	$1.2\ 10^{10}$
Water Depth in Local Box (m)	7.5
Suspended Sediment Load in Local Box (t m^{-3})	$2.5\ 10^{-5}$
Sedimentation Rate in Local Box (t $m^{-2}\ y^{-1}$)	$1.0\ 10^{-4}$
Fish catch in Local Box (t y^{-1})	4.62
Crustacea catch in Local Box (t y^{-1})	0.552
Mollusc catch in Local Box (t y^{-1})	10.4
Seaweed harvest in Local Box (t y^{-1})	-
Collective beach occupancy in Local Box	$9.6\ 10^{6}$
Length of Coastline in Local Box (m)	$1.92\ 10^{5}$

Table C3 Exposed group habits assumed

Intakes	kg y^{-1}
Fruit	75
Green Veg	80
Root veg & potatoes	130
Cereals	100
Beef	45
Cow Offal	10
Mutton	25
Sheep Offal	10
Milk	300
Eggs	25
Chicken	30
Drinking Water	600
Freshwater fish	20
Marine Fish	20
Marine Crustacea	5
Marine Molluscs	5
Occupancies	**h y^{-1}**
Beach	300
Handling fishing gear	2000
Farmer ploughing	300

Table C4 Individual doses from effluent disposal to Thames Tideway

Nuclide	Time of peak dose	Dose from disposals, Sv y^{-1} per GBq y^{-1} disposed			
		Dose	Pathways		
1 y of disposals					
H-3	1	$4.9\ 10^{-14}$	Sea fish	Milk	Mollusc
C-14	1	$2.7\ 10^{-8}$	Sea fish	Crustacea	Mollusc
P-32	1	$8.1\ 10^{-8}$	Sea fish	Crustacea	Mollusc
S-35	1	$7.8\ 10^{-12}$	Mutton	Sea fish	Beef
Cr-51	1	$3.8\ 10^{-10}$	Fish gear	Sea fish	Crustacea
Ga-67	1	$1.0\ 10^{-9}$	Fish gear	Sea fish	Crustacea
Rb-83	1	$1.3\ 10^{-10}$	Fish gear	Sea fish	Crustacea
Rb-84	1	$2.0\ 10^{-10}$	Fish gear	Sea fish	Crustacea
Sr-89	1	$1.1\ 10^{-11}$	Sea fish	Crustacea	Mollusc
Y-90	1	$4.4\ 10^{-12}$	Crustacea	Mollusc	Sea fish
Tc-99m	1	$2.0\ 10^{-12}$	Crustacea	Mollusc	Fish Gear
In-111	1	$3.5\ 10^{-9}$	Fish Gear	Crustacea	Mollusc
I-123	1	$3.9\ 10^{-12}$	Sea fish	Fish Gear	Crustacea
I-125	1	$3.5\ 10^{-10}$	Sea fish	Crustacea	Mollusc
I-131	1	$3.9\ 10^{-10}$	Sea fish	Crustacea	Mollusc
Tl-201	1	$7.0\ 10^{-10}$	Fish gear	Sea fish	Mollusc
Am-241	1	$3.3\ 10^{-8}$	Mollusc	Crustacea	Fish Gear
50 y of disposals					
H-3	50	$4.9\ 10^{-14}$	Sea fish	Milk	Mollusc
C-14	50	$2.7\ 10^{-8}$	Sea fish	Crustacea	Mollusc
P-32	50	$8.1\ 10^{-8}$	Sea fish	Crustacea	Mollusc
S-35	50	$7.9\ 10^{-12}$	Mutton	Sea fish	Beef
Cr-51	50	$3.8\ 10^{-10}$	Fish gear	Sea fish	Crustacea
Ga-67	50	$1.0\ 10^{-9}$	Fish gear	Sea fish	Crustacea
Rb-83	50	$1.3\ 10^{-10}$	Fish gear	Sea fish	Crustacea
Rb-84	50	$2.0\ 10^{-10}$	Fish gear	Sea fish	Crustacea
Sr-89	50	$1.1\ 10^{-11}$	Sea fish	Crustacea	Mollusc
Y-90	50	$4.4\ 10^{-12}$	Crustacea	Mollusc	Sea fish
Tc-99m	50	$2.0\ 10^{-12}$	Crustacea	Mollusc	Fish Gear
In-111	50	$3.5\ 10^{-9}$	Fish Gear	Crustacea	Mollusc
I-123	50	$3.9\ 10^{-12}$	Sea fish	Fish Gear	Crustacea
I-125	50	$3.5\ 10^{-10}$	Sea fish	Crustacea	Mollusc
I-131	50	$3.9\ 10^{-10}$	Sea fish	Crustacea	Mollusc
Tl-201	50	$7.0\ 10^{-10}$	Fish gear	Sea fish	Mollusc
Am-241	50	3.310^{-8}	Mollusc	Crustacea	Fish Gear

Table C5 Collective doses truncated at 500 y from disposal of effluents to Thames Tideway

Nuclide	Collective dose, man Sv per GBq y^{-1}		
	Local terrestrial	Local marine	UK
1 y of disposals			
H-3	$2.1\ 10^{-9}$	$6.1\ 10^{-12}$	$2.2\ 10^{-9}$
C-14	$1.3\ 10^{-7}$	$3.7\ 10^{-6}$	$1.0\ 10^{-4}$
P-32	$2.2\ 10^{-7}$	$1.0\ 10^{-5}$	$2.2\ 10^{-5}$
S-35	$1.2\ 10^{-7}$	$6.8\ 10^{-10}$	$1.2\ 10^{-7}$
Cr-51	$2.5\ 10^{-10}$	$2.6\ 10^{-8}$	$3.4\ 10^{-8}$
Ga-67	$1.2\ 10^{-9}$	$1.5\ 10^{-8}$	$1.9\ 10^{-8}$
Rb-83	$2.1\ 10^{-7}$	$1.8\ 10^{-7}$	$5.0\ 10^{-7}$
Rb-84	$2.3\ 10^{-7}$	$2.2\ 10^{-8}$	$2.8\ 10^{-7}$
Sr-89	$3.8\ 10^{-8}$	$1.3\ 10^{-9}$	$4.4\ 10^{-8}$
Y-90	$1.5\ 10^{-8}$	$7.6\ 10^{-10}$	$1.6\ 10^{-8}$
Tc-99m	$1.5\ 10^{-10}$	$2.6\ 10^{-10}$	$4.2\ 10^{-10}$
In-111	$1.9\ 10^{-9}$	$8.8\ 10^{-8}$	$1.2\ 10^{-7}$
I-123	$1.1\ 10^{-8}$	$5.9\ 10^{-10}$	$1.2\ 10^{-8}$
I-125	$1.1\ 10^{-6}$	$4.8\ 10^{-8}$	$1.3\ 10^{-6}$
I-131	$1.3\ 10^{-6}$	$5.4\ 10^{-8}$	$1.4\ 10^{-6}$
Tl-201	$9.1\ 10^{-10}$	$5.4\ 10^{-8}$	$7.3\ 10^{-8}$
Am-241	$7.7\ 10^{-6}$	$5.2\ 10^{-5}$	$4.5\ 10^{-4}$
50 y of disposals			
H-3	$1.0\ 10^{-7}$	$3.0\ 10^{-10}$	$1.1\ 10^{-7}$
C-14	$6.4\ 10^{-5}$	$1.9\ 10^{-4}$	$5.2\ 10^{-3}$
P-32	$1.1\ 10^{-5}$	$5.2\ 10^{-4}$	$1.1\ 10^{-3}$
S-35	$5.9\ 10^{-6}$	$3.4\ 10^{-8}$	$6.1\ 10^{-6}$
Cr-51	$1.3\ 10^{-8}$	$1.3\ 10^{-6}$	$1.7\ 10^{-6}$
Ga-67	$6.0\ 10^{-8}$	$7.5\ 10^{-7}$	$9.7\ 10^{-7}$
Rb-83	$1.0\ 10^{-5}$	$8.8\ 10^{-6}$	$2.5\ 10^{-5}$
Rb-84	$1.2\ 10^{-5}$	$1.1\ 10^{-6}$	$1.4\ 10^{-5}$
Sr-89	$1.9\ 10^{-6}$	$6.5\ 10^{-8}$	$2.2\ 10^{-6}$
Y-90	$7.7\ 10^{-7}$	$3.8\ 10^{-8}$	$8.4\ 10^{-7}$
Tc-99m	$7.6\ 10^{-9}$	$1.3\ 10^{-8}$	$2.1\ 10^{-8}$
In-111	$9.3\ 10^{-8}$	$4.4\ 10^{-6}$	$6.2\ 10^{-6}$
I-123	$5.4\ 10^{-7}$	$2.9\ 10^{-8}$	$5.8\ 10^{-7}$
I-125	$5.7\ 10^{-5}$	$2.4\ 10^{-6}$	$6.9\ 10^{-5}$
I-131	$6.4\ 10^{-5}$	$2.7\ 10^{-6}$	$6.9\ 10^{-5}$
Tl-201	$4.5\ 10^{-8}$	$2.7\ 10^{-6}$	$3.7\ 10^{-6}$
Am-241	$3.9\ 10^{-4}$	$2.6\ 10^{-3}$	$2.2\ 10^{-2}$

Table C6 River model set up.

River Characteristic	Section 1	Section 2
Length m	500	10,000
Width m	5	25
Water depth m	1	2
Water volume m^3	2500	500,000
Bed sediment depth m	0.1	0.1
Bulk dry sediment density kg m^{-3}	1500	1500
River suspended sediment load kg m^{-3}	0.04	0.04
River water flows		
Velocity m s^{-1}	0.2	0.2
Volumetric flow m^3 s^{-1}	1	10
Bed sediment flow		
Velocity m s^{-1}	$2\ 10^{-5}$	$2\ 10^{-5}$
Volumetric flow m^3 s^{-1}	$1\ 10^{-5}$	$5\ 10^{-5}$
Local Marine Box receiving river section 2		
Volume m^3		$1\ 10^{9}$
Depth m		10
Exchange with regional compartment m^3 y^{-1}		$1\ 10^{11}$
		$1\ 10^{-4}$
Suspended sediment load t m^{-3}		$1\ 10^{-4}$
Sedimentation rate t m^{-2} y^{-1}		20
Fish catch t y^{-1}		2
Mollusc catch t y^{-1}		0.5
Crustacea catch t y^{-1}		$1\ 10^{4}$
Coastline length m		$5\ 10^{5}$
Coastline occupancy man hr y^{-1}		

Table C7 Individual doses from effluent disposal to a river of unit volumetric flow rate ($1\ m^3\ s^{-1}$)

Nuclide	Time of peak dose	Dose from disposals, Sv y^{-1} per GBq y^{-1} disposed: Dose	Pathways		
1 y of disposals					
H-3	1	$7.2\ 10^{-10}$	Drinking water	Milk	Chicken
C-14	1	$1.7\ 10^{-6}$	Fish	Chicken	Eggs
P-32	1	$7.3\ 10^{-5}$	Fish	Milk	Drinking water
S-35	1	$2.1\ 10^{-7}$	Fish	Mutton	Beef
Cr-51	1	$1.2\ 10^{-9}$	Fish	Drinking water	Pasture external
Ga-67	1	$3.1\ 10^{-8}$	Fish	Drinking water	Green veg
Rb-83	1	$7.1\ 10^{-7}$	Fish	Milk	Pasture external
Rb-84	1	$1.0\ 10^{-6}$	Fish	Milk	Drinking water
Sr-89	1	$1.4\ 10^{-7}$	Fish	Drinking water	Milk
Y-90	1	$8.2\ 10^{-8}$	Drinking water	Fish	Green veg
Tc-99m	1	$8.1\ 10^{-10}$	Drinking water	Milk	Fish
In-111	1	$1.3\ 10^{-6}$	Fish	Drinking water	Pasture external
I-123	1	$7.6\ 10^{-9}$	Drinking water	Fish	Milk
I-125	1	$5.5\ 10^{-7}$	Drinking water	Fish	Milk
I-131	1	$8.0\ 10^{-7}$	Drinking water	Fish	Milk
Tl-201	1	$5.5\ 10^{-7}$	Fish	Drinking water	Beef
Am-241	1	$5.5\ 10^{-7}$	Drinking water	Fish	Green veg
50 y of disposals					
H-3	50	$7.2\ 10^{-10}$	Drinking water	Milk	Chicken
C-14	50	$1.7\ 10^{-6}$	Fish	Chicken	Eggs
P-32	50	$7.3\ 10^{-5}$	Fish	Milk	Drinking water
S-35	50	$2.1\ 10^{-7}$	Fish	Mutton	Beef
Cr-51	50	$1.2\ 10^{-9}$	Fish	Drinking water	Pasture external
Ga-67	50	$3.1\ 10^{-8}$	Fish	Drinking water	Green veg
Rb-83	50	$7.6\ 10^{-7}$	Fish	Pasture external	Milk
Rb-84	50	$1.0\ 10^{-6}$	Fish	Milk	Drinking water
Sr-89	50	$1.4\ 10^{-7}$	Fish	Drinking water	Milk
Y-90	50	$8.2\ 10^{-8}$	Drinking water	Fish	Green veg
Tc-99m	50	$8.1\ 10^{-10}$	Drinking water	Milk	Fish
In-111	50	$1.3\ 10^{-6}$	Fish	Drinking water	Pasture external
I-123	50	$7.6\ 10^{-9}$	Drinking water	Fish	Milk
I-125	50	$5.5\ 10^{-7}$	Drinking water	Fish	Milk
I-131	50	$8.0\ 10^{-7}$	Drinking water	Fish	Milk
Tl-201	50	$5.5\ 10^{-7}$	Fish	Drinking water	Beef
Am-241	50	$1.0\ 10^{-6}$	Soil resuspension	Drinking water	Fish

Table C8 Collective doses truncated at 500 y from effluent disposal to river

Nuclide	Collective dose, man Sv per GBq y^{-1}		
	Local terrestrial	Local marine	UK
1 y of disposals			
H-3	$3.6\ 10^{-5}$	$2.0\ 10^{-12}$	$3.6\ 10^{-5}$
C-14	$1.1\ 10^{-3}$	$1.0\ 10^{-6}$	$1.2\ 10^{-3}$
P-32	$6.4\ 10^{-3}$	$4.0\ 10^{-6}$	$6.4\ 10^{-3}$
S-35	$1.4\ 10^{-3}$	$1.5\ 10^{-10}$	$1.4\ 10^{-3}$
Cr-51	$4.0\ 10^{-5}$	$1.6\ 10^{-9}$	$4.0\ 10^{-5}$
Ga-67	$2.5\ 10^{-4}$	$6.1\ 10^{-10}$	$2.5\ 10^{-4}$
Rb-83	$1.0\ 10^{-3}$	$5.6\ 10^{-7}$	$1.0\ 10^{-3}$
Rb-84	$1.5\ 10^{-3}$	$4.1\ 10^{-9}$	$1.5\ 10^{-3}$
Sr-89	$4.7\ 10^{-3}$	$4.7\ 10^{-10}$	$4.7\ 10^{-3}$
Y-90	$4.3\ 10^{-3}$	$1.1\ 10^{-11}$	$4.3\ 10^{-3}$
Tc-99m	$2.8\ 10^{-5}$	$7.5\ 10^{-12}$	$2.8\ 10^{-5}$
In-111	$4.1\ 10^{-4}$	$2.1\ 10^{-9}$	$4.1\ 10^{-4}$
I-123	$4.0\ 10^{-4}$	$1.5\ 10^{-10}$	$4.0\ 10^{-4}$
I-125	$2.9\ 10^{-2}$	$1.5\ 10^{-8}$	$2.9\ 10^{-2}$
I-131	$4.2\ 10^{-2}$	$1.7\ 10^{-8}$	$4.2\ 10^{-2}$
Tl-201	$1.8\ 10^{-4}$	$7.6\ 10^{-9}$	$1.8\ 10^{-4}$
Am-241	$2.2\ 10^{-2}$	$1.2\ 10^{-4}$	$2.2\ 10^{-2}$
50 y of disposals			
H-3	$1.8\ 10^{-3}$	$9.8\ 10^{-11}$	$1.8\ 10^{-3}$
C-14	$5.6\ 10^{-2}$	$5.2\ 10^{-5}$	$5.9\ 10^{-2}$
P-32	$3.2\ 10^{-1}$	$2.0\ 10^{-4}$	$3.2\ 10^{-1}$
S-35	$6.8\ 10^{-2}$	$7.7\ 10^{-9}$	$6.8\ 10^{-2}$
Cr-51	$2.0\ 10^{-3}$	$7.8\ 10^{-8}$	$2.0\ 10^{-3}$
Ga-67	$1.3\ 10^{-2}$	$3.1\ 10^{-8}$	$1.3\ 10^{-2}$
Rb-83	$5.1\ 10^{-2}$	$2.8\ 10^{-5}$	$5.2\ 10^{-2}$
Rb-84	$7.5\ 10^{-2}$	$2.1\ 10^{-7}$	$7.5\ 10^{-2}$
Sr-89	$2.4\ 10^{-1}$	$2.3\ 10^{-8}$	$2.4\ 10^{-1}$
Y-90	$2.1\ 10^{-1}$	$5.7\ 10^{-10}$	$2.1\ 10^{-1}$
Tc-99m	$1.4\ 10^{-3}$	$3.8\ 10^{-10}$	$1.4\ 10^{-3}$
In-111	$2.1\ 10^{-2}$	$1.1\ 10^{-7}$	$2.1\ 10^{-2}$
I-123	$2.0\ 10^{-2}$	$7.6\ 10^{-9}$	$2.0\ 10^{-2}$
I-125	$1.5\ 10^{0}$	$7.7\ 10^{-7}$	$1.5\ 10^{0}$
I-131	$2.1\ 10^{0}$	$8.6\ 10^{-7}$	$2.1\ 10^{0}$
Tl-201	$8.8\ 10^{-3}$	$3.8\ 10^{-7}$	$8.8\ 10^{-3}$
Am-241	$1.1\ 10^{0}$	5.810^{-3}	$1.1\ 10^{0}$

Table C9 Data for atmospheric dispersion model

Parameter	Value
Effective stack height	100 m (Beckton) 90 m (Knostrop)
Roughness length of terrain	1 m
Meteorological conditions	Uniform windrose, 60 % Pasquill stability class D (neutral)

Table C10 Habits for groups exposed to atmospheric releases

Parameter	Resident	Farmer
Distance from stack, km	1.4 (Knostrop & Beckton)	13 (Beckton) 5 (Knostrop)
Ingestion rate	Critical for root veg, green veg, fruit	Critical for all foods
Fraction of foods obtained locally	20 % of root veg, green veg, fruit	100 % of all foods
Indoors occupancy	50 %	50 %

Table C11 Individual doses for incinerator release for two exposed groups (Beckton)

Nuclides	Dose to resident at 1.4 km, Sv y^{-1}		Doses to farmer at 13 km, Sv y^{-1}	
	1 y release	50 y releases	1 y release	50 y releases
H-3	$5.3\ 10^{-13}$	$5.3\ 10^{-13}$	$2.7\ 10^{-13}$	$2.7\ 10^{-13}$
C-14	$3.6\ 10^{-11}$	$3.6\ 10^{-11}$	$6.4\ 10^{-11}$	$6.4\ 10^{-11}$
P-32	$2.8\ 10^{-10}$	$2.8\ 10^{-10}$	$5.7\ 10^{-10}$	$5.8\ 10^{-10}$
S-35	$3.2\ 10^{-11}$	$3.3\ 10^{-11}$	$5.3\ 10^{-10}$	$5.7\ 10^{-10}$
Cr-51	$5.1\ 10^{-12}$	$5.7\ 10^{-12}$	$1.3\ 10^{-12}$	$1.4\ 10^{-12}$
Ga-67	$5.9\ 10^{-12}$	$5.6\ 10^{-12}$	$7.7\ 10^{-12}$	$7.7\ 10^{-13}$
Rb-83	$2.2\ 10^{-10}$	$3.0\ 10^{-10}$	$3.0\ 10^{-10}$	$3.0\ 10^{-10}$
Rb-84	$2.3\ 10^{-10}$	$2.5\ 10^{-10}$	$5.1\ 10^{-10}$	$1.2\ 10^{-9}$
Sr-89	$1.9\ 10^{-10}$	$1.9\ 10^{-10}$	$5.0\ 10^{-11}$	$5.1\ 10^{-11}$
Y-90	$2.0\ 10^{-10}$	$2.0\ 10^{-10}$	$2.3\ 10^{-11}$	$2.3\ 10^{-11}$
Tc-99m	5.710^{-13}	$5.7\ 10^{-13}$	$5.5\ 10^{-14}$	$5.5\ 10^{-14}$
In-111	$9.6\ 10^{-12}$	$9.7\ 10^{-12}$	$1.3\ 10^{-12}$	$1.3\ 10^{-12}$
I-123	$3.0\ 10^{-12}$	$3.4\ 10^{-12}$	$3.1\ 10^{-13}$	$3.1\ 10^{-13}$
I-125	$1.2\ 10^{-9}$	$1.2\ 10^{-9}$	$4.2\ 10^{-9}$	$4.2\ 10^{-9}$
I-131	$7.0\ 10^{-10}$	$7.1\ 10^{-10}$	$2.3\ 10^{-9}$	$2.3\ 10^{-9}$
Tl-201	$2.0\ 10^{-12}$	$2.0\ 10^{-10}$	$8.6\ 10^{-13}$	$8.7\ 10^{-13}$
Am-241	$4.5\ 10^{-7}$	$4.5\ 10^{-7}$	$5.5\ 10^{-8}$	$5.6\ 10^{-8}$

Table C12 Individual doses from incinerator release for two exposed groups (Knostrop)

Nuclides	Dose to resident at 1.4 km, Sv y^{-1}		Doses to farmer at 5 km, Sv y^{-1}	
	1 y release	50 y releases	1 y release	50 y releases
H-3	$6.6\ 10^{-13}$	$6.6\ 10^{-13}$	$1.1\ 10^{-12}$	$1.1\ 10^{-12}$
C-14	$4.5\ 10^{-11}$	$4.5\ 10^{-11}$	$2.6\ 10^{-10}$	$2.6\ 10^{-10}$
P-32	$3.5\ 10^{-10}$	$3.5\ 10^{-10}$	$2.3\ 10^{-9}$	$2.3\ 10^{-9}$
S-35	$4.0\ 10^{-11}$	$4.0\ 10^{-11}$	$2.3\ 10^{-9}$	$2.3\ 10^{-9}$
Cr-51	$6.3\ 10^{-12}$	$7.1\ 10^{-12}$	$5.4\ 10^{-12}$	$5.7\ 10^{-12}$
Ga-67	$7.3\ 10^{-12}$	$7.3\ 10^{-12}$	$3.1\ 10^{-12}$	$3.1\ 10^{-12}$
Rb-83	$2.7\ 10^{-10}$	$3.7\ 10^{-10}$	$1.2\ 10^{-9}$	$1.2\ 10^{-9}$
Rb-84	$2.8\ 10^{-10}$	$3.1\ 10^{-10}$	$2.1\ 10^{-9}$	$4.7\ 10^{-9}$
Sr-89	$2.4\ 10^{-10}$	$2.4\ 10^{-10}$	$2.0\ 10^{-10}$	$2.1\ 10^{-10}$
Y-90	$2.5\ 10^{-10}$	$2.5\ 10^{-10}$	$9.2\ 10^{-11}$	$9.2\ 10^{-11}$
Tc-99m	$7.0\ 10^{-13}$	$7.0\ 10^{-13}$	$2.3\ 10^{-13}$	$2.3\ 10^{-13}$
In-111	$1.2\ 10^{-11}$	$1.2\ 10^{-11}$	$5.3\ 10^{-12}$	$5.3\ 10^{-12}$
I-123	$3.7\ 10^{-12}$	$3.7\ 10^{-12}$	$1.2\ 10^{-12}$	$1.2\ 10^{-12}$
I-125	$1.5\ 10^{-9}$	$1.5\ 10^{-9}$	$1.7\ 10^{-8}$	$1.7\ 10^{-8}$
I-131	$8.7\ 10^{-10}$	$8.7\ 10^{-10}$	$9.4\ 10^{-9}$	$9.4\ 10^{-9}$
Tl-201	$2.5\ 10^{-12}$	$2.5\ 10^{-12}$	$3.5\ 10^{-12}$	$3.5\ 10^{-12}$
Am-241	$5.6\ 10^{-7}$	$5.6\ 10^{-7}$	$2.2\ 10^{-7}$	$2.3\ 10^{-7}$

Table C13 Collective doses to UK population from Beckton and Knostrop incinerators

Nuclide	Collective dose from 1 GBq release over 1 y, truncated at 500 y (man Sv)	
	Beckton	Knostrop
H-3	$1.1\ 10^{-6}$	$1.1\ 10^{-6}$
C-14	$2.9\ 10^{-4}$	$3.1\ 10^{-4}$
P-32	$4.7\ 10^{-4}$	$5.1\ 10^{-4}$
S-35	$2.1\ 10^{-4}$	$3.5\ 10^{-4}$
Cr-51	$1.0\ 10^{-5}$	$6.1\ 10^{-6}$
Ga-67	$8.7\ 10^{-6}$	$5.0\ 10^{-6}$
Rb-83	$6.0\ 10^{-4}$	$4.8\ 10^{-4}$
Rb-84	$8.9\ 10^{-4}$	$1.1\ 10^{-3}$
Sr-89	$3.1\ 10^{-4}$	$2.0\ 10^{-4}$
Y-90	$3.8\ 10^{-4}$	$2.1\ 10^{-4}$
Tc-99m	$7.4\ 10^{-7}$	$3.8\ 10^{-7}$
In-111	$1.5\ 10^{-5}$	$8.7\ 10^{-6}$
I-123	$4.2\ 10^{-6}$	$2.1\ 10^{-6}$
I-125	$1.5\ 10^{-3}$	$2.4\ 10^{-3}$
I-131	$9.1\ 10^{-4}$	$1.3\ 10^{-3}$
Tl-201	$3.4\ 10^{-6}$	$2.2\ 10^{-6}$
Am-241	$5.7\ 10^{-1}$	$3.5\ 10^{-1}$

Footnote:
Collective dose from 50 years of release 50 times higher than those from 1 years release.

Table C14 Local marine compartment for disposal of sludge at Barrow Deeps

Parameter	Value
Volume of Local Box (m^3)	$8.86\ 10^{9}$
Exchange Rate with Marine compartment ($m^3\ y^{-1}$)	$3.19\ 1\ 0^{12}$
Water Depth in Local Box (m)	7
Suspended Sediment Load in Local Box ($t\ m^{-3}$)	$6.0\ 10^{-6}$
Sedimentation Rate in Local Box ($t\ m^{-2}\ y^{-1}$)	$1.0\ 10^{-4}$
Fish catch in Local Box ($t\ y^{-1}$)	315
Crustacea catch in Local Box ($t\ y^{-1}$)	37.7
Mollusc catch in Local Box ($t\ y^{-1}$)	709
Seaweed harvest in Local Box ($t\ y^{-1}$)	-
Collective beach occupancy in Local Box	$5.5\ 10^{6}$
Length of Coastline in Local Box (m)	$1.10\ 10^{5}$

Table C15 Individual doses for off-shore disposal of sludges at Barrow Deeps

Nuclide	Time of peak dose	Dose from disposals, Sv y^{-1} per GBq y^{-1} disposed			
		Dose	Pathways		
1 y of disposals					
H-3	1	$7.1\ 10^{-16}$	Sea fish	Milk	Crustacea
C-14	1	$4.2\ 10^{-10}$	Sea fish	Crustacea	Mollusc
P-32	1	$4.8\ 10^{-10}$	Sea fish	Crustacea	Mollusc
S-35	1	$7.2\ 10^{-14}$	Mutton	Sea fish	Beef
Cr-51	1	$4.3\ 10^{-12}$	Fish gear	Sea fish	Crustacea
Ga-67	1	$1.1\ 10^{-11}$	Fish gear	Sea fish	Crustacea
Rb-83	1	$1.7\ 10^{-12}$	Fish gear	Sea fish	Crustacea
Rb-84	1	$1.4\ 10^{-12}$	Fish gear	Sea fish	Crustacea
Sr-89	1	$8.7\ 10^{-14}$	Sea fish	Crustacea	Mollusc
Y-90	1	$1.2\ 10^{-13}$	Crustacea	Mollusc	Sea fish
In-111	1	$5.1\ 10^{-11}$	Fish Gear	Crustacea	Mollusc
I-123	1	$2.2\ 10^{-14}$	Sea fish	Fish Gear	Crustacea
I-125	1	$2.5\ 10^{-12}$	Sea fish	Crustacea	Mollusc
I-131	1	$2.3\ 10^{-12}$	Sea fish	Crustacea	Mollusc
Tl-201	1	$6.0\ 10^{-12}$	Fish gear	Sea fish	Mollusc
Am-241	1	$1.8\ 10^{-9}$	Mollusc	Crustacea	Fish Gear
50 y of disposals					
H-3	50	$7.9\ 10^{-16}$	Sea fish	Milk	Mollusc
C-14	50	$4.8\ 10^{-10}$	Sea fish	Crustacea	Mollusc
P-32	50	$4.8\ 10^{-10}$	Sea fish	Crustacea	Mollusc
S-35	50	$7.2\ 10^{-14}$	Mutton	Sea fish	Beef
Cr-51	50	$4.3\ 10^{-12}$	Fish gear	Sea fish	Crustacea
Ga-67	50	$1.1\ 10^{-11}$	Fish gear	Sea fish	Crustacea
Rb-83	50	$1.9\ 10^{-12}$	Fish gear	Sea fish	Crustacea
Rb-84	50	$1.4\ 10^{-12}$	Fish gear	Sea fish	Crustacea
Sr-89	50	$8.7\ 10^{-14}$	Sea fish	Crustacea	Mollusc
Y-90	50	$1.2\ 10^{-13}$	Crustacea	Mollusc	Sea fish
In-111	50	$5.1\ 10^{-11}$	Fish Gear	Crustacea	Mollusc
I-123	50	$2.2\ 10^{-14}$	Sea fish	Fish Gear	Crustacea
I-125	50	$2.5\ 10^{-12}$	Sea fish	Crustacea	Mollusc
I-131	50	$2.3\ 10^{-12}$	Sea fish	Crustacea	Mollusc
Tl-201	50	$6.0\ 10^{-12}$	Fish gear	Sea fish	Mollusc
Am-241	50	$1.9\ 10^{-9}$	Mollusc	Crustacea	Fish Gear

For Tc-99m, doses are zero.

Table C16 Collective doses for off-shore disposal of sludges at Barrow Deeps truncated at 500 y

Nuclide	Collective dose, man Sv per GBq y^{-1}		
	Local terrestrial	Local marine	UK
1 y of disposals			
H-3	$1.9\ 10^{-11}$	$6.8\ 10^{-12}$	$1.6\ 10^{-10}$
C-14	$1.3\ 10^{-9}$	$4.4\ 10^{-6}$	$1.1\ 10^{-4}$
P-32	$7.2\ 10^{-10}$	$4.2\ 10^{-6}$	$1.8\ 10^{-5}$
S-35	$6.2\ 10^{-10}$	$4.3\ 10^{-10}$	$4.5\ 10^{-9}$
Cr-51	$9.5\ 10^{-13}$	$1.9\ 10^{-9}$	$1.0\ 10^{-8}$
Ga-67	$4.4\ 10^{-12}$	$5.4\ 10^{-9}$	$1.0\ 10^{-8}$
Rb-83	$1.8\ 10^{-9}$	$9.3\ 10^{-9}$	$1.3\ 10^{-7}$
Rb-84	$9.2\ 10^{-10}$	$4.8\ 10^{-9}$	$3.1\ 10^{-8}$
Sr-89	$1.7\ 10^{-10}$	$7.0\ 10^{-10}$	$5.8\ 10^{-9}$
Y-90	$5.9\ 10^{-11}$	$1.4\ 10^{-9}$	$2.6\ 10^{-9}$
In-111	$7.0\ 10^{-12}$	$6.5\ 10^{-8}$	$1.2\ 10^{-7}$
I-123	$3.6\ 10^{-11}$	$1.8\ 10^{-10}$	$5.0\ 10^{-10}$
I-125	$5.4\ 10^{-9}$	$2.7\ 10^{-8}$	$2.3\ 10^{-7}$
I-131	$4.2\ 10^{-9}$	$2.1\ 10^{-8}$	$6.8\ 10^{-8}$
Tl-201	$3.4\ 10^{-12}$	$3.1\ 10^{-8}$	$5.8\ 10^{-8}$
Am-241	$7.7\ 10^{-8}$	$4.3\ 10^{-5}$	$4.3\ 10^{-4}$
50 y of disposals			
H-3	$9.7\ 10^{-10}$	$3.4\ 10^{-10}$	$8.1\ 10^{-9}$
C-14	$6.2\ 10^{-8}$	$2.2\ 10^{-4}$	$5.3\ 10^{-3}$
P-32	$3.6\ 10^{-8}$	$2.1\ 10^{-4}$	$8.8\ 10^{-4}$
S-35	$3.1\ 10^{-8}$	$2.1\ 10^{-8}$	$2.3\ 10^{-7}$
Cr-51	$4.8\ 10^{-11}$	$9.5\ 10^{-8}$	$5.2\ 10^{-7}$
Ga-67	$2.2\ 10^{-10}$	$2.7\ 10^{-7}$	$5.2\ 10^{-7}$
Rb-83	$8.9\ 10^{-8}$	$4.6\ 10^{-7}$	$6.5\ 10^{-6}$
Rb-84	$4.6\ 10^{-8}$	$2.4\ 10^{-7}$	$1.6\ 10^{-6}$
Sr-89	$8.6\ 10^{-9}$	$3.5\ 10^{-8}$	$2.9\ 10^{-7}$
Y-90	$2.9\ 10^{-9}$	$7.1\ 10^{-8}$	$1.3\ 10^{-7}$
In-111	$3.5\ 10^{-10}$	$3.3\ 10^{-6}$	$6.0\ 10^{-6}$
I-123	$1.8\ 10^{-9}$	$9.1\ 10^{-9}$	$2.5\ 10^{-8}$
I-125	$2.7\ 10^{-7}$	$1.3\ 10^{-6}$	$1.1\ 10^{-5}$
I-131	$2.1\ 10^{-7}$	$1.0\ 10^{-6}$	$3.4\ 10^{-6}$
Tl-201	$1.7\ 10^{-10}$	$1.5\ 10^{-6}$	$2.9\ 10^{-6}$
Am-241	$3.4\ 10^{-6}$	$2.1\ 10^{-3}$	$2.1\ 10^{-2}$

For Tc-99m, doses are zero.

Table C17 Individual and collective doses arising from landfill disposal

Nuclide	Individual dose per 1 GBq per y	Collective dose per 1 GBq per y truncated at 500 y
	Dose, Sv y^{-1}	Dose, man Sv
H-3	$6.3\ 10^{-11}$	$1.12\ 10^{-5}$
C-14	$1.0\ 10^{-8}$	$2.1\ 10^{-3}$
Am-241	$1.7\ 10^{-10}$	0

Table C18 Individual doses from application of sewage sludge to land

Nuclide	Time of peak dose	Dose from disposals, Sv y^{-1} per Bq kg^{-1} in sludge Critical	Average	Pathways
1 y of disposals				
H-3	1	$1.8\ 10^{-10}$	$4.4\ 10^{-11}$	Milk
C-14	1	$7.4\ 10^{-8}$	$2.3\ 10^{-8}$	Milk/Milk products
P-32	1	$8.4\ 10^{-8}$	$2.9\ 10^{-8}$	Milk/Milk products
S-35	1	$1.5\ 10^{-7}$	$5.0\ 10^{-8}$	Milk/Milk products
Cr-51	1	$2.8\ 10^{-10}$	$9.6\ 10^{-11}$	External
Rb-83	1	$3.1\ 10^{-8}$	$1.1\ 10^{-8}$	Milk/Milk products, External
Rb-84	1	$2.2\ 10^{-8}$	$7.4\ 10^{-9}$	External, Milk/Milk products
Sr-89	1	$2.3\ 10^{-9}$	$7.4\ 10^{-10}$	Milk/Milk products, Meat
I-125	1	$1.5\ 10^{-8}$	$5.0\ 10^{-9}$	Milk/Milk products
Am-241	1	$1.6\ 10^{-8}$	$1.1\ 10^{-8}$	Soil Inhalation, offal
50 y of disposals				
H-3	50	$1.8\ 10^{-10}$	$4.4\ 10^{-11}$	Milk
C-14	50	$7.6\ 10^{-8}$	$2.3\ 10^{-8}$	Milk/Milk products
P-32	50	$8.8\ 10^{-8}$	$3.0\ 10^{-8}$	Milk/Milk products
S-35	50	$1.6\ 10^{-7}$	$5.3\ 10^{-8}$	Milk/Milk products
Cr-51	50	$3.1\ 10^{-10}$	$1.1\ 10^{-10}$	External
Rb-83	50	$3.7\ 10^{-8}$	$1.3\ 10^{-8}$	Milk/Milk products, External
Rb-84	50	$2.3\ 10^{-8}$	$7.9\ 10^{-9}$	External, Milk/Milk products
Sr-89	50	$2.3\ 10^{-9}$	$7.4\ 10^{-10}$	Milk/Milk products, Meat
I-125	50	$1.5\ 10^{-8}$	$5.1\ 10^{-9}$	Milk/Milk products
Am-241	50	$2.0\ 10^{-7}$	$1.1\ 10^{-7}$	Soil Inhalation, offal

Doses from Ga-67, Y-90, Tc-99m, In-111, I-123, I-131, Tl-201 are predicted to be zero.

Table C19 Collective doses to UK from application of sewage sludge to land truncated at 500 y.

Nuclide	Collective dose, man Sv per Bq kg^{-1} per kg of sludge production
1 y of disposals	
H-3	$2.6\ 10^{-14}$
C-14	$3.9\ 10^{-12}$
P-32	$1.3\ 10^{-12}$
S-35	$1.9\ 10^{-12}$
Cr-51	$3.4\ 10^{-17}$
Rb-83	$3.5\ 10^{-13}$
Rb-84	$2.0\ 10^{-13}$
Sr-89	$9.7\ 10^{-14}$
I-125	$2.5\ 10^{-13}$
Am-241	$5.1\ 10^{-13}$

Doses from Ga-67, Y-90, Tc-99m, In-111, I-123, I-131, Tl-201 are predicted to be zero.

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APPENDIX D

DATA USED TO RUN SEWER MODEL FOR ASSESSING RADIONUCLIDE TRANSPORT (SMART)

This appendix is a list of all the input data required to run the Sewer Model for Assessing Radionuclide Transport. Some of the data are fixed, while others depend on the case studied. The section of the methodology to which the data refer is mentioned wherever appropriate. The methodology is presented in Appendix B of this report.

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1 General Data

Table D 1: Half-lives of nuclides (T_{Hi})

Nuclide	Half-life	Half-life in hours
H-3	12.35 y	1.081E+5
C-14	5730 y	5.019E+7
P-32	14.29 d	3.430E+2
S-35	87.44d	2.098E+3
Cr-51	27.7 d	6.648E+2
Ga-67	78.3 h	7.830E+1
Rb-83	86 d	2.064E+3
Rb-84	32.8 d	7.872E+2
Sr-89	50 d	1.200E+3
Y-90	64 h	6.400E+1
Tc-99m	6.02 h	6.020E+0
In-111	2.83 d	6.792E+1
I-123	13.2 h	1.320E+1
I-125	60 d	1.440E+3
I-131	8.04 d	1.930E+2
Tl-201	3.04 d	7.296E+1
Am-241	432 y	3.784E+6

Table D 2: Typical times involved in transport and treatment (T)

Definition	Section of Appendix D	Name	Range of Times /hours	Chosen Value /hours
Delay between blockage and repair of small pipes	2.2.1	T_d	2[a]	2
Delay to sewage during preliminary treatment	3.2.1	T_0	Fractions of an hour[b]	0.1
Retention time of gross floating solids during preliminary treatment	3.2.1	T_{0s}	Fractions of an hour[b]	0.5
Delay to sewage during primary treatment	3.3.1	T_1	2 – 6[c]	2
Retention time of sludge during primary treatment	3.3.1	T_{1s}	2 – 6[c]	4
Delay to sewage during secondary treatment	3.4.1	T_2	2.5 – 8[h,i]	2.5-Knostrop 8-Beckton
Retention time of sludge during secondary treatment	3.4.1	T_{2s}	4.5 – 20[j]	4.5 – Knostrop[k] 312-Beckton[k]
Retention time of sludge during initial processing	4.2.1	T_4	~24[d]	24
Retention time of sludge during stabilisation	4.3.1	T_5	At least 816[e]	816
Retention time of sludge during de-watering	4.4.1	T_6	Up to 72[f]	2[g]
Retention time of sludge during incineration	4.5.1	T_7	Up to 72[f]	0.5[g]

Notes:
[a] Assumed value.
[b] Reference 6.
[c] References 8, 13, 14 & 15.
[d] References 8 & 13.
[e] Reference 16.
[f] References 5 & 6.
[g] Pessimistic assumption.
[h] Reference 8, pessimistically assumed 2.5hours for Knostrop.
[i] Reference 17, 8hours required to remove most impurities.
[j] References 8 & 15.
[k] The difference in retention times between Beckton and Knostrop is confirmed by the measurement of Tc-99m in incinerator sludge at Knostrop but not at Beckton.

Table D 3: External dose rates for point sources ($H_{i, ext, d}$)
Source: RP 65

Nuclide	Point source gamma Sv/h per Bq	Point source beta Sv/h per Bq	Total Sv/h per Bq
H-3	0.00	0.00	0.00
C-14	0.00	0.00	0.00
P-32	0.00	2.70E-14	2.7E-14
S-35	0.00	0.00	0.00
Cr-51	3.57E-15	0.00	3.57E-15
Ga-67	1.70E-14	0.00	1.70E-14
Rb-83	5.04E-14	0.00	5.04E-14
Rb-84	9.18E-14	3.00E-17	9.18E-14
Sr-89	9.60E-18	2.20E-14	2.20E-14
Y-90	1.69E-19	3.74E-14	3.74E-14
Tc-99m	1.28E-14	0.00	1.28E-14
In-111	4.54E-14	0.00	4.54E-14
I-123	1.99E-14	0.00	1.99E-14
I-125	8.08E-15	0.00	8.08E-15
I-131	4.19E-14	8.44E-16	4.27E-14
Tl-201	9.49E-15	0.00	9.49E-15
Am-241	2.13E-15	1.00E-17	2.14E-15

Table D 4: External dose rates from inside a uniformly contaminated, 1m radius sewage pipe ($H_{i, ext, p}$)
Source: Reference 12.

Radionuclide	Sv h^{-1} per Bq g^{-1}
H-3	0.00
C-14	0.00
P-32	4.2E-11
S-35	0.00
Cr-51	9.3E-11
Ga-67	4.0E-10
Rb-83	1.4E-9
Rb-84	2.5E-9
Sr-89	2.0E-13
Y-90	7.3E-11
Tc-99m	3.1E-10
In-111	1.2E-09
I-123	3.5E-10
I-125	6.7E-10
I-131	1.1E-9
Tl-201	2.0E-10
Am-241	8.2E-10

Table D 5: Effective gamma dose rates 1m above sewage or sludge ($H_{i, ext, w}$)
Source: Reference 1.

Radionuclide	Sv h^{-1} per Bq m^{-3}
H-3	0.00
C-14	0.00
P-32	0.00
S-35	0.00
Cr-51	6.81E-15
Ga-67	3.49E-14
Rb-83	1.04E-13
Rb-84	1.93E-13
Sr-89	1.79E-17
Y-90	6.65E-21
Tc-99m	2.98E-14
In-111	8.76E-14
I-123	3.51E-14
I-125	1.72E-15
I-131	8.13E-14
Tl-201	2.01E-14
Am-241	4.40E-15

Table D 6: Inhalation ($H_{i, inh}$) and Ingestion ($H_{i, ing}$) effective dose coefficients for adult members of the public
Source: Reference 2.

Radionuclide	Inhalation Dose Coefficient Sv Bq^{-1}	Ingestion Dose Coefficient Sv Bq^{-1}
H-3 (M tritiated water)	4.5E-11	1.8E-11
H-3 (M OBT)	4.1E-11	4.2E-11
C-14 (M)	2.0E-9	5.8E-10
P-32 M	3.4E-9	2.4E-9
S-35 organic	1.1E-10	7.7E-10
Cr-51 M	3.2E-11	3.8E-11
Ga-67 M	2.4E-10	1.9E-10
Rb-83 F	6.9E-10	1.9E-9
Rb-84 (F)	1.0E-9	2.8E-9
Sr-89 (M)	6.1E-9	2.6E-9
Y-90 M	1.4E-9	2.7E-9
Tc-99m (M)	1.9E-11	2.2E-11
In-111 M	2.3E-10	2.9E-10
I-123 (F)	7.4E-11	2.1E-10
I-125 (F)	5.1E-9	1.5E-8
I-131 (F)	7.4E-9	2.2E-8
Tl-201 (F)	4.4E-11	9.5E-11
Am-241 (M)	4.2E-5	2.0E-7

Table D 7: Suspended solids in air concentration (L).
Value: 1E-7 kg m^{-3} for ambient conditions, 1E-5 kg m^{-3} for dusty environments or conditions where aerosols are likely to occur. Source: Reference 3.

Definition	Section of Appendix D	Name	kg m^{-3}
From small pipes	2.2.1	L_d	1E-7
From large pipes	2.4.1	L_p	1E-7
From preliminary effluent	5.1	$L_{0,e}$	1E-7
From raw solids	5.1	$L_{0,s}$	1E-5
From primary effluent	5.1	$L_{1,e}$	1E-5
From primary sludge	5.1	$L_{1,s}$	1E-7
From secondary effluent	5.1	$L_{2,e}$	1E-7
From secondary sludge	5.1	$L_{2,s}$	1E-7
From tertiary/final effluent	5.1	$L_{3,e}$	1E-7
From processed sludge	5.1	$L_{S,4}$	1E-7
From stabilised sludge	5.1	$L_{S,5}$	1E-5
From de-watered and dried sludge	5.1	$L_{S,6}$	1E-5
From ash	5.1	$L_{S,7}$	1E-5

Table D 8: Workers' habit data.
Source: Habit Data report, reference 4.

Habit data	
Inadvertent ingestion rate	1.25E-6 kg h^{-1}
Working adult breathing rate	1.18 m^3 h^{-1}

Table D 9: Transfer to pipe walls coefficient (G_i)
Refer to section 2.4.1 of appendix B, peak values.

Radionuclide	Bq m^{-2} per Bq d^{-1}
H-3	1.177E-5
C-14	1.180E-5
P-32	6.042E-6
S-35	1.046E-5
Cr-51	8.189E-6
Ga-67	1.575E-6
Rb-83	1.045E-5
Rb-84	8.642E-6
Sr-89	9.606E-6
Y-90	1.241E-6
Tc-99m	3.909E-8
In-111	1.334E-6
I-123	1.429E-7
I-125	9.919E-6
I-131	3.988E-6
Tl-201	1.454E-6
Am-241	1.180E-5

Table D 10: Assumed preliminary treatment removal coefficients ($E_{TOT\,i\,0}$).
Refer to section 3.2.1 of appendix B.

Radionuclide	Bq m^{-3} per Bq m^{-3}
H-3	1
C-14	1
P-32	1
S-35	1
Cr-51	1
Ga-67	1
Rb-83	1
Rb-84	1
Sr-89	1
Y-90	1
Tc-99m	1
In-111	1
I-123	1
I-125	1
I-131	1
Tl-201	1
Am-241	1

Notes:
Any losses of activity at this stage are assumed to be minimal and are ignored. A removal efficiency of 1 implies that all activity is transferred to primary treatment.

Table D 11: Default primary treatment removal coefficients ($E_{TOT\,i\,1}$).

Radionuclide	Bq m^{-3} per Bq m^{-3}
H-3	1[b]
C-14	1[a]
P-32	1[a]
S-35	1[a]
Cr-51	1[b]
Ga-67	1[b]
Rb-83	1[b]
Rb-84	1[b]
Sr-89	1[b]
Y-90	1[b]
Tc-99m	1[a]
In-111	1[b]
I-123	0.93[a]
I-125	0.93[a]
I-131	0.93[a]
Tl-201	1[b]
Am-241	1[b]

Notes:
[a] Based on measured data.
[b] Extrapolated from measured data.

Table D 12: Best estimates of secondary treatment removal coefficients ($E_{TOT\,i\,2}$)
Refer to section 3.4 of Appendix B.

Radionuclide	Activated sludge Bq m^{-3} per Bq m^{-3}	Percolating filters Bq m^{-3} per Bq m^{-3}
H-3	0.90[a]	0.90[a]
C-14	0.70[a]	0.70[a]
P-32	0.15[b]	0.25[b]
S-35	0.90[a]	0.90[a]
Cr-51	0.20[a]	0.40[a]
Ga-67	0.20[c]	0.20[c]
Rb-83	0.20[c]	0.20[c]
Rb-84	0.20[c]	0.20[c]
Sr-89	0.20[c]	0.20[c]
Y-90	0.20[c]	0.20[c]
Tc-99m	0.90[a]	0.90[a]
In-111	0.20[c]	0.20[c]
I-123	0.90[b]	0.90[b]
I-125	0.90[b]	0.90[b]
I-131	0.90[b]	0.90[b]
Tl-201	0.20[c]	0.20[c]
Am-241	0.20[c]	0.20[c]

Notes:
[a] Derived from literature values, ref. 20 and results from measurements.
[b] Estimated from literature and measured values, refs 20-22, conservative best estimates have been taken from ranges of values.
[c] Extrapolated from literature values for other metallic elements. 0.2 is a conservative estimate because it will result in higher activity concentrations in sludges.

Table D 13: Default tertiary treatments removal coefficients ($E_{TOT\,i\,3}$).

Radionuclide	Tertiary treatment Bq m^{-3} per Bq m^{-3}
H-3	1
C-14	1
P-32	1
S-35	1
Cr-51	1
Ga-67	1
Rb-83	1
Rb-84	1
Sr-89	1
Y-90	1
Tc-99m	1
In-111	1
I-123	1
I-125	1
I-131	1
Tl-201	1
Am-241	1

Note:
Tertiary treatment does not take place at the STW's studied, so there is no measured data available.

Table D 14: Assumed volatilised fraction for sludge incineration (V_i)

Radionuclide	Best estimate[a] Bq m^{-3} per Bq m^{-3}	Range Bq m^{-3} per Bq m^{-3}
H-3	0.99	0.99 – 1[g]
C-14	0.99	0.97 – 0.99[d]
P-32	0.01	0.00 – 0.34[d]
S-35	0.10	0.10 – 1.00[d]
Cr-51	0.01	0.00 – 0.29[d]
Ga-67	0.23[b]	0.23 – 0.36[h]
Rb-83	0.01[f]	0.00 – 0.18[i]
Rb-84	0.01[f]	0.00 – 0.18[i]
Sr-89	0.20[I]	0.19 – 0.21[i]
Y-90	0.20[I]	0.19 – 0.21[i]
Tc-99m	0.95	0.93 – 0.95[e]
In-111	0.61[b]	0.61 – 0.65[h]
I-123	0.95	0.90 – 0.99[c]
I-125	0.95	0.90 – 0.99[c]
I-131	0.95	0.90 – 0.99[c]
Tl-201	0.95[I]	0.95 – 0.998[i]
Am-241	0.05[j]	0.01 – 0.30[j]

Notes:

[a] Best estimate values were taken from ref. 9 unless stated otherwise.

[b] Measurement taken from Beckton sampling, where results available for incinerator cake and incinerator ash.

[c] Ranges estimated using Beckton and Knostrop sampling results and data from refs. 9, 10 & 11.

[d] Ranges estimated using data from refs. 9,10 & 19.

[e] Range estimated using Knostrop sampling results and data from ref. 9.

[f] Estimated on the basis of results for Cs-137.

[g] Range estimated using data from refs. 9 & 19.

[h] Range estimated using Beckton sampling results and data from ref.19.

[i] Based on information from ref. 19.

[j] Extrapolated from values for other heavy metals from ref.19.

Table D 15: Suspended solids loads in effluents and sludges (F).
Based on references 5, 6, 7 & 8.

Medium	Section of Appendix B	Name	kg m^{-3}
Incoming raw sewage	3.3	$F_0 = F_{0e}$	0.5
Effluent after primary treatment	3.3	F_{1e}	0.15
Sludge from primary and secondary treatment combined	3.3	F_s	15 Beckton 50 Knostrop
Effluent after secondary treatment	3.4	F_{2e}	0.02
Sludge after initial sludge processing	4.2	F_{4s}	35
Sludge after stabilisation	4.2	F_{5s}	35
Sludge after de-watering and drying	4.2	F_{6s}	300

Table D 16: Occupancy times for workers (T).

Definition	Section of Appendix B	Name	h y^{-1}
Transport of sewage to the works			
Hours spent repairing small pipes blocked with monthly radionuclide discharge	2.2.1	T_w	2
Hours per year spent in man-accessible pipes [c]	2.4.1	T_p	1600
Treatment of sewage at the works [b]			
Hours spent close to preliminary effluent	5.1	$T_{w,0,e}$	100
Hours spent close to preliminary solids	5.1	$T_{w,0,s}$	200
Hours spent close to primary effluent	5.1	$T_{w,1,e}$	100
Hours spent close to primary sludge	5.1	$T_{w,1,s}$	200
Hours spent close to secondary effluent	5.1	$T_{w,2,e}$	100
Hours spent close to secondary sludge	5.1	$T_{w,2,s}$	200 (Beckton) 0 (Knostrop)
Hours spent close to tertiary/final effluent	5.1	$T_{w,3,e}$	100
Treatment of sludge at the works [a, b]			
Hours spent close to processed sludge	5.1	$T_{w,4}$	400
Hours spent close to stabilised sludge	5.1	$T_{w,5}$	600
Hours spent close to de-watered sludge	5.1	$T_{w,6}$	200
Hours spent close to ash / incinerator	5.1	$T_{w,7}$	400
Hours spent close to de-watered sludge - sludge press workers	5.1	$T_{w,8}$	2000

Notes:

[a] Option disposal at sea: use $T_{w,4}$, and $T_{w,5}$; option incinerator: use $T_{w,4}$, $T_{w,6}$ and $T_{w,7}$.

[b] 1000 h/y divided between different areas.

[c] Reference 5.

2 Data specific to Beckton Sewage Works (London)

Table D17: Average monthly amount of activity authorised for discharge per disposer ($B_{i,d}$ in Bq)

NB. Disposers are listed in order of ranking, refer to Table 23 of main report which gives the ranking of the disposers.

Disposer	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
H-3														2.0E+10	1.0E+11					1.0E+09
C-14		5.0E+09						2.4E+09		1.5E+09	4.1E+09					4.0E+08	2.0E+09	1.5E+09		1.0E+08
P-32					5.0E+08	7.4E+08					1.0E+10				1.0E+09					2.0E+09
S-35											1.5E+10				3.4E+09			2.1E+09		2.5E+09
Cr-51																				7.0E+08
Ga-67					2.5E+09								2.4E+09							4.0E+09
Rb-84																				
Sr-89					1.0E+09															
Y-90						7.4E+08														
Tc-99m	8.0E+10		5.0E+10	5.0E+10	7.5E+10	3.0E+10	2.5E+10	8.0E+10	6.0E+10			1.5E+10	1.2E+11						3.5E+10	
In-111					1.0E+09	5.0E+08														
I-123					2.0E+09			1.0E+10												
I-125	9.5E+10		3.7E+10		1.0E+08	3.8E+10		2.0E+10		5.0E+08	2.5E+09					5.0E+07	7.0E+08	9.0E+08	6.0E+07	1.0E+09
I-131		1.0E+10		1.0E+10	3.0E+10		1.7E+10		1.5E+10			1.0E+09								2.0E+08
Tl-201					1.0E+09								1.2E+10							
α emitters (Am-241)														4.0E+05						
Other β/γ	9.0E+09	5.0E+10	2.0E+10	2.5E+10	2.0E+08	2.0E+08	3.5E+09	5.0E+09	4.0E+09	7.5E+09	4.5E+09	6.0E+09	5.9E+09	5.0E+09	5.0E+08	3.0E+09	2.3E+09	2.1E+09	2.2E+09	1.5E+09

Table D 18: Average distance from dischargers to the works
Refer to section 2.5 of Appendix B.

	Value in m
Average distance	20,000

Table D 19: Number of workers involved in sewage and sewers works (man).

Definition	Section of Appendix B	Name	man
Workers involved in small pipes repairs	2.2.2	n_d	2
Workers involved in large pipes repairs	2.4.2	n_p	2
Workers at the sewage treatment plant	5.2	n_w	27

Table D 20: Total authorised monthly radionuclide disposal rates (q_i in Bq) to Beckton (sewage).

Radionuclide	Monthly q_i (Bq)	Daily[a] q_I (Bq)
H-3	2.1E+10	7.0E+8
C-14	1.2E+11	4.0E+9
P-32	1.4E+10	4.67E+9
S-35	2.3E+10	7.67E+8
Cr-51	7.0E+8	2.33E+8
Ga-67	4.9E+9	1.63E+8
Rb-84	4.9E+9	1.63E+8
Sr-89	1.0E+9	3.33E+7
Y-90	7.4E+8	2.47E+7
Tc-99m	6.2E+11	2.07E+10
In-111	1.5E+9	5.0E+7
I-123	1.2E+10	4.0E+8
I-125[b]	1.6E+11	5.33E+9
I-131	8.3E+10	2.77E+9
Tl-201	1.3E+10	4.33E+8
Am-241	4.0E+5	1.33E+4
Other $\beta\gamma$[c]	1.61E+11	5.37E+9

Notes:
[a] Monthly amount divided by 30.
[b] Includes 'Iodines' or 'I-131/I-125' types of authorisations.
[c] Considered with I-131.

Table D 21: Total actual monthly radionuclide disposal rates (q_i in Bq) to Beckton (sewage).

Radionuclide	Monthly q_i (Bq)	Daily[a] q_I (Bq)
H-3	2.35E+09	7.83E+07
C-14	1.02E+11	3.40E+09
P-32	4.13E+09	1.38E+08
S-35	1.02E+10	3.40E+08
Cr-51	1.47E+09	4.90E+07
Ga-67	2.91E+09	9.70E+07
Rb-84	4.0E+09	1.33E+08
Sr-89	1.63E+08	5.43E+06
Y-90	2.04E+09	6.80E+07
Tc-99m	4.05E+11	1.35E+10
In-111	1.42E+07	4.73E+05
I-123	7.37E+08	2.46E+07
I-125[b]	4.10E+10	1.37E+09
I-131	5.63E+10	1.88E+09
Tl-201	1.38E+10	4.6E+08
Am-241	0	0
Other $\beta\gamma$[c]	4.15E+10	1.38E+09

Notes:
[a] Monthly amount divided by 30.
[b] Includes 'Iodines' or 'I-131/I-125' types of authorisations.
[c] Considered with I-131.

Table D 22: Volumetric flow rates in large sewers (Q_0 in $m^3\ d^{-1}$)
Refer to section 2.3 of Appendix B. Source: Thames Water.

Case	$m^3\ d^{-1}$
Maximum rate of flow	2 700 000
Average flow	1 080 000

Table D 23: Flow rates and amounts at various points (Q)
Source: Thames Water.

Definition	Section of Appendix B	Name	$m^3\ d^{-1}$
Effluent flow rate after preliminary treatment	3.2	Q_{0e}	Q_0
Amount of solids removed after preliminary treatment	3.2	Q_{0s}	~1365
Effluent flow rate after primary treatment	3.3	Q_{1e}	Q_0
Amount of sludge removed from primary treatment	3.3	Q_{1s}	10000[a]
Amount of sludge removed from secondary treatment	3.4	Q_{2s}	6048[a]
Amount of outgoing sludge after de-watering	4.4	Q_{6s}	752
Final effluent flow rate	6.1	$Q_{out\ e}$	Q_0
Amount of ash removed per day (t/d)	4.5.1	Q_{ash}	48
Amount of sludge disposed at sea per day	4.3.1 and 6.2	$Q_{S\ sea}$	7260
Amount of sludge disposed to landfill per day	6.3	$Q_{S\ land}$	0
Amount of sludge disposed to agricultural land	6.4	$Q_{S\ agr}$	0

Note:
[a] Reference 18.

3 Data specific to Knostrop Sewage Works (Leeds)

Table D 24: Average monthly amount of activity authorised for discharge per disposer ($B_{i,d}$ in Bq)

NB. Disposers are listed in order of ranking, refer to table 31 of main report which gives the ranking of the disposers.

Disposer	1	2	3	4	5	6
H-3	0	0	0	0	0	1.0E+9
C-14	0	0	0	8.0E+10[a]	3.0E+7[a]	0
P-32	0	0	0	0	0	0
S-35	0	0	0	0	0	0
Cr-51	0	0	0	0	0	0
Ga-67	0	0	0	0	0	0
Rb-83	0	0	0	0	0	0
Rb-84	0	0	0	0	0	0
Sr-89	0	0	0	0	0	0
Y-90	0	0	0	0	0	0
Tc-99m	4.0E+10	2.0E+11	2.0E+11	0	0	0
In-111	0	0	0	0	0	0
I-123	0	0	0	0	0	0
I-125	2.8E+10[b]	1.5E+10[b]	0	2.0E+9[c]	1.9E+8[c]	0
I-131	0	0	0	0	0	0
Tl-201	0	0	0	0	0	0
α emitters	0	0	0	1.0E+7	0	5.0E+5
Other β / γ	2.0E+9	1.0E+10	2.0E+10	1.2E+10	1.9E+8	1.0E+6

Notes:

[a] These values are for C-14 and H-3 discharges, and not for C-14 only.

[b] These values are for all iodines.

[c] This value is for I-125/I-131.

Table D 25: Average distance from dischargers to the works in m

Refer to section 2.5 of Appendix B.

	Value in m
Average distance	5,500

Table D 26: Number of workers involved in sewage and sewers works (man).

Definition	Section of Appendix B	Name	man
Workers involved in small pipes repairs	2.2.2	n_d	2
Workers involved in large pipes repairs	2.4.2	n_p	2
Workers at the sewage treatment plant	5.2	n_w	9

Table D 27: Total monthly authorised disposal rates (q_i in Bq) to Knostrop.

Radionuclide	Monthly q_i (Bq)	Daily[a] Q_i (Bq)
H-3	1.0E+9	3.33E+7
C-14	8.0E+10[b]	2.67E+9[b]
P-32	0	0
S-35	0	0
Cr-51	0	0
Ga-67	0	0
Rb-83	0	0
Rb-84	0	0
Sr-89	0	0
Y-90	0	0
Tc-99m	4.4E+11	1.46E+10
In-111	0	0
I-123	0	0
I-125	4.5E+10[c]	1.50E+9[c]
I-131	0	0
Tl-201	0	0
Am-241 (α's)	1.1E+7	3.67E+5
Other β / γ	4.4E+10[d]	1.40E+9[d]

Notes:
[a] Monthly amount divided by 30.
[b] These values are for C-14 and H-3 discharges, and not for C-14 only.
[c] Includes 'Iodines' and 'I-131/I-125' types of authorisations.
[d] Considered with I-131.

Table D 28: Total monthly actual disposal rates (q_i in Bq) to Knostrop.

Radionuclide	Monthly q_i (Bq)	Daily[a] Q_i (Bq)
H-3	3.7E+09	1.23E+08
C-14	2.7E+07[b]	8.9E+05[b]
P-32	2.5E+08	8.33E+06
S-35	5.8E+07	1.93E+06
Cr-51	6.7E+07	2.23E+06
Ga-67	0	0
Rb-84	0	0
Sr-89	2.0E+08	6.67E+06
Y-90	0	0
Tc-99m	1.5E+11	5.0E+09
In-111	0	0
I-123	2.0E+08	6.67E+06
I-125	2.08E+09[c]	6.94E+07[c]
I-131	1.3E+10	4.33E+08
Tl-201	0	0
Am-241 (α's)	0	0
Other β / γ	2.3E+09[d]	7.67E+07[d]

Notes:
[a] Monthly amount divided by 30.
[b] These values are for C-14 and H-3 discharges, and not for C-14 only.
[c] Includes 'Iodines' and 'I-131/I-125' types of discharges.
[d] Considered with I-131.

Table D 29: Volumetric flow rates in large sewers (Q_0 in $m^3 d^{-1}$)
Refer to section 2.3 of appendix B.

Case	$m^3 d^{-1}$
Dry weather flow	79,000
Average flow	97,600
Maximum flow	205,000

Table D 30: Flow rates and amounts at various points (Q)
Source: Knostrop data, refs. 7 & 8.

Definition	Section of Appendix B	Name	$m^3 d^{-1}$
Effluent flow rate after preliminary treatment	3.2	Q_{0e}	Q_0
Amount of solids removed after preliminary treatment	3.2	Q_{0s}	49
Effluent flow rate after primary treatment	3.3	Q_{1e}	Q_0
Amount of sludge removed from primary treatment	3.3	Q_{1s}	702
Amount of sludge removed from secondary treatment	3.4	Q_{2s}	0
Amount of outgoing sludge after de-watering	4.4	Q_{6s}	159
Final effluent flow rate	6.1	$Q_{out\ e}$	Q_0
Amount of ash removed per day (t/d)	4.5.1	Q_{ash}	11
Amount of sludge disposed to landfill per day[a]	6.3	$Q_{S\ land}$	702
Amount of sludge disposed to agricultural land[a]	6.4	$Q_{S\ agr}$	702

Notes:
[a] Either disposal to landfill or disposal to agricultural land can be used.

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APPENDIX E

SAMPLING AT WORCESTER PARK SEWAGE TREATMENT WORKS

1 INTRODUCTION

Worcester Park sewage works in London was subject to a programme of monitoring for discharged radionuclides. This work was carried out as an extension to the main study. Sampling was carried out at this works because it is a small urban sewage works which receives discharges of radionuclides almost entirely from one disposer (a hospital specialising in cancer therapy). Studying Worcester Park should provide measured activity concentrations that can be related to the actual discharges and the variation with time observed.

1.1 Worcester Park Works

Worcester Park carries out primary and secondary treatment of sewage and discharges treated effluents to a brook which flows to the Thames Estuary. The treatment process is fairly complex. Raw effluent arrives at the works from Sutton sewer, Cheam Sewer and Worcester Park sewer, is combined and then divided between four primary sedimentation tanks. Separated primary solids are recovered, and transferred to Mogden Works for treatment. Primary effluent passes forward to secondary treatment, where it is divided into 16 streams and subjected to biological filtration. The secondary effluent from biological filtration is then settled in humus tanks. Effluents emerging from humus tanks are divided into two streams, one stream is discharged to the environment via outfall B, stream 2 is retained and divided into two. Final effluent from stream 2a is subject to further treatment: aerobic digestion, final settling and composting lakes before discharge via outfall A. Stream 2b goes straight to composting lakes before discharge to the environment, also via outfall A. Some of the recovered solids from aerobic digestion are used as feedstock for the aeration process, the remainder are returned to Worcester Park sewer and reenter the STW for complete retreatment.

Treated sewage sludges at 2% solid are exported to Mogden Works for treatment. Worcester Park STW closed early in 1999 and all sewage treated there is now diverted to Hogsmill.

1.2 Radionuclide discharges to Worcester Park

Information on the hospital disposal practices indicated that the principal radionuclides discharged were ^{99m}Tc and ^{131}I. The radiological assessment carried out as part of the present study indicated that these are the radionuclides of most importance in terms of doses to the workforce at STW at Beckton and Knostrop (see chapters 7 and 12 of main report). Current disposal practice at the hospital in Worcester Park catchment is for ^{99m}Tc administration and discharge more or less continuously throughout the week. Iodine-131 is used in thyroid ablation and is administered on a single day, usually Tuesday, with the resulting discharges occurring over the next 48 h or so. This practice therefore provided an opportunity to study the temporal variability of a pulse of ^{131}I as it passed through the sewage treatment process, which in turn could enable current modelling assumptions to be verified.

2 MONITORING STRATEGY

Following discussions with the disposer, it was decided to monitor Worcester Park STW on 9 and 10 December 1998. This timing was chosen because 20 GBq of ^{131}I was due to be administered to a patient at 15:00 on 8 December. Previous measurements had shown that the peak activity in the outfall of the STW occurred approximately 30 hours after injection i.e. at about 21:00 on 9 December. Samples of effluent and sludges from the primary and secondary treatment stages were taken at various times.

Samples of the liquid effluent from the primary settlement tank were taken at 11:30, 14:00 and 17:30 on 9 December and 9:15 and 13:40 on the following day. It was originally intended to take samples of the sludge from the primary settlement tank at the same times. However, the pump removing the sludge from the tanks was only working intermittently and so sampling was not possible on all occasions. Samples of sludge were taken at 11:30 and 18:00 on 9 December and 09:30 on the following morning. Although this was unfortunate, the sludge was likely to be of secondary importance because most of the ^{131}I entering the STW was expected to remain in the liquid phase (see main report).

The activated (secondary) sludge was sampled at 11:30 and 13:15 on 9 December and 09:15 on the following day. The treated effluent from the outfall of the STW (outfall B) was sampled at 12:00, 13:15, 14:30, 15:30 and 17:30 on 9 December. A composite sample was then taken by auto sampler from 18:30 on 9 December to 08:30 on the following day. Manual sampling then resumed and a sample was taken at 09:45, 10:45, 11:45, 12:45 and 13:45 on 10 December. The operators reported that the plant flow during that period was 30,000 m^3 per day with 200 m^3 of sludge produced.

3 SAMPLE PREPARATION AND ANALYSIS

For gamma-ray spectrometry, reliable results can only be obtained if the sample remains homogeneous during the entire counting period, which may be up to 16 hours. For the reasons given in Chapters 8 and 13 of the main text, this is a particularly important factor in the present study. For this reason, all liquid effluent samples were filtered on site and the residue and filter paper discarded. On return to the laboratory all sludge samples were concentrated to a stable paste by centrifugation. All samples were then packed into standard containers and gamma-ray emitting radionuclides determined using high-purity Ge detectors housed in a purpose built facility and appropriately calibrated. These measurements took place at the Board's Chilton laboratory. These analyses are within the scope of the laboratory's formal United Kingdom Accreditation Service (UKAS) accreditation.

Given the objective of this part of the study, analytical work was confined to gamma-ray spectrometry.

4 RESULTS AND DISCUSSION

Only one of the primary sludge and two of the primary fluid samples contained measurable amounts of ^{99m}Tc and these had high associated counting uncertainties. Discussion will therefore be confined to the measurements of ^{131}I. Table E1 lists the ^{131}I results expressed in terms of Bq kg^{-1} of the sample as collected. These results are expressed graphically in Figure E1. It is important to note that the liquids and sludges are plotted against different scales.

Activity concentrations in the final effluent were observed to start to increase at about 1500 h on 9 December, with the peak occurring at about 1800 h, about 27 h after the activity had been administered. This was in good agreement with the earlier estimates of transit time. From Figure E1, it is not possible to be certain whether sampling started early enough to catch the peak concentrations in the effluents in the primary settlement tank. The transit time for material to pass through the STW is about 8 - 10 h. From the trend observed in the final liquid effluent, this would imply that activity concentrations in the liquid in the primary settlement tank would start to increase between about 0500 - 0700 on 9 December. Sampling did not start until 11.30 am, so it is possible that the peak may have been missed. Any future studies of this type should therefore begin sampling the primary effluents at the STW in the early hours of the morning after the activity had been administered. In contrast, the activity concentrations in the primary sludges continued to increase over the duration of the monitoring. This was to be expected since the sludge removed via the pump would represent the cumulative effect of activity entering the STW over the previous few hours.

The initial peak of activity in liquids from the primary settlement tanks passed in no more than about 12h, whereas the activity concentrations in the final liquid effluent declined more slowly: only after about 20 h did the activity concentrations approach the values observed at in the primary effluent at the start of the sampling programme. These results suggest that a considerable degree of mixing occurs during the transport of liquid through the STW.

The hospital is the only contributor of ^{131}I to the Worcester Park STW. Consequently, concentrations in the effluents from the primary settlement tanks should have been below detection limits of around 1 Bq l^{-1} or less prior to the arrival of the pulse of activity. Activity concentrations would then have increased by around two orders of magnitude between about 0600 h and 1200 h on 9 December. Future studies of this type should therefore use relatively short time intervals between sampling during this period. Patients undergoing thyroid oblation therapy generally excrete about 50% of the administered activity within 24 h of intake, a total of 90% being excreted within 48h. This implies 10GBq discharged in the first 24 h and 8 GBq in the second 24 h. The implied pattern is broadly consistent with the temporal variability of activity concentrations observed in liquid from the primary settlement tanks. This showed an initial sharp peak in over the first 6 hours of sampling, followed by a decline with activity concentrations to around half the initial peak at the end of the sampling period. The implications for future studies are therefore that transport of activity through the plant should be started earlier and followed for longer than the 27 h period used in the present

study.

4.1 Estimated iodine budget

The measured activity concentrations in the primary and secondary liquid effluents were used to estimate the total amount of ^{131}I passing through the plant during the present study. Using the activity concentrations from primary effluents at various times and the average flow of effluent, (1,250 m^3 h^{-1}) the estimated throughput of iodine-131 during the 27 hour sampling cycle was approximately 4.5GBq. It should be noted that 20 hours had elapsed since administration to the patient and first sampling and so a significant amount of activity (perhaps 2-5 GBq) may already have passed through the primary effluents at the STW before sampling began. This implies that between 10-13.5 GBq had still to arrive at the works after sampling ended. An estimate of the amount of iodine expected over the 24 hours following sampling was made. Assuming that the decline in concentration in the primary effluents at the same rate continued gives an approximate average concentration in primary effluent over this period of around 100 Bq kg^{-1}. Over the 24 h period a further 3 GBq would pass through primary effluent. Thus over the first 70 hours or so after administration, 10-13 GBq of the administered iodine had been estimated to have passed through the primary effluent. Allowing for radioactive decay on the remaining iodine (accounting for a further 1-2 GBq of the iodine remaining over the first 70 h), a further 5-8 GBq of the administered iodine was outstanding. If the patient was sent to a hospital nearer to home after 72 hours, the remaining disposals may occur to another sewage works elsewhere.

Using the measured activity concentrations for the final effluent in Table E1, the total amount of activity that had been discharged by the plant was about 1.3 GBq over the first 27 hour of sampling. This is a small fraction of the total ^{131}I administered. This suggests that activity concentrations in the final liquid effluent may remain elevated for some considerable time after the administration has been carried out, which reinforces the idea that future studies such as this should carry out sampling for longer periods than those employed here.

5 CONCLUSIONS

This study showed that a discrete set of I-131 discharges can be traced through a sewage system. However, it was not possible to account for all the I-131 introduced into the system. This may be because sampling did not commence soon enough after the administration of the I-131 to the patient or significant amounts (50%) of the administered I-131 was still to be excreted by the patient after smpling ceased and would be discharged later (i.e. after 48h). The latter may have implications for patients who are sent either home or to smaller local hospitals in rural areas 48 h after thyroid ablation treatment.

Table E1, Worcester Park ^{131}I results.

	Bq kg^{-1}, as sampled			
Date and Time	Primary Solids	Primary Fluids	Activated Sludge	Final effluent
09/12/98 11:30	61	234	42	-
09/12/98 12:00	-	-	-	20
09/12/98 13:15	-	-	-	21
09/12/98 14:00	-	224	60	-
09/12/98 14:30	-	-	-	20
09/12/98 15:30	-	-	-	27
09/12/98 17:30	-	129	-	51
09/12/98 18:00	680	-	-	-
09/12/98 18:30 to 10/12/98 08:30	-	-	-	48
10/12/98 09:15	-	117	114	-
10/12/98 09:30	1,305	-	-	-
10/12/98 09:45	-	-	-	35
10/12/98 10:45	-	-	-	31
10/12/98 11:45	-	-	-	28
10/12/98 12:45	-	-	-	28
10/12/98 13:45	-	112	-	25

APPENDIX F

Comparison of SMART predicted activity concentrations with measured results from Beckton and Knostrop

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1. INTRODUCTION

Raw sewage effluent, sludges, treated effluents, incinerator cake, ash and off gas scrubbers at Beckton and Knostrop sewage treatment works were analysed 11 discharged radionuclides. Where radionuclides were detected, the results were converted to activity concentrations and were compared with those predicted by the SMART model.

Activity concentrations predicted by SMART were compared with measured values were compared for five media: raw incoming effluent, secondary sludge, final effluent, incinerator cake and residual ash. Activity concentrations of any nuclides that were above the limits of detection are presented. Where radionuclide concentrations were below the limit of detection, the detection limit is given instead.

As a result of the conservative assumptions made in the model it is to be expected that, in the main, the predicted concentrations will be above the measured values. Two sets of activity concentrations have been predicted, one on the basis that the disposal authorisations are being used to their fullest extent, and the second using the estimated typical discharges. Both assume that discharges are continuous, so the predicted concentrations may be below the measured values if the discharges occur as pulses and the sampling occurs when a pulse passes through.

2. INCOMING SEWAGE EFFLUENT

Table F1 shows the activity concentrations in filtered incoming raw sewage effluent at Beckton for four radionuclides that were above limits of detection. Measured activity concentrations have been compared with the activity concentration after preliminary treatment predicted by SMART. SMART assumes that the time taken to pass through preliminary treatment is very short, so there is practically no decay and only large floating items are removed from the flow.

The raw incoming sewage was sampled at Beckton only, on one day in the summer. Two samples were taken, one collected between 10am and 3pm and the other from 3pm to 8pm. C-14, P-32, Tc-99m and I-131 were above the limits of detection, both sets of measurements are shown in Table F1. The predicted activity concentrations are in good agreement with measurement for Tc-99m and I-131 and an order of magnitude too high for C-14. However, The activity concentration of P-32 was several predicted orders of magnitude lower than the measured value. The difference for P-32 suggests that either.

1) P-32 was discharged as a pulse and that the sampling occurred when the pulse passed through, or:
2) discharges of P-32 are one or two orders of magnitude higher than expected from the authorised discharges, or:
3) There is an error in the measured result.

The normal uses of P-32 do not suggest that discharges would occur as a pulse. P-32 is normally used in diagnosis or research, which suggests that the P-32 should be discharged fairly gradually. Another reason for the higher than expected measured value may arise from higher

discharges than permitted as a named discharge nuclide on a disposal authorisation. The amount of P-32 disposed has been estimated by assuming maximum authorised discharges, however only named P-32 authorisations were considered. P-32 may be discharged by premises without a named P-32 authorisation under an other βγ authorisation, therefore the total amount of P-32 disposed under authorisations may be higher than that assumed in the model. If it is assumed that all the other βγ authorisation is P-32, then the predicted activity concentration in raw sewage becomes 3.5 Bq/kg. This is still considerably lower than the measured result, and it is highly unlikely that all the other βγ authorisations of the major disposers in central London are used to dispose P-32. Hence the reason for high P-32 result is not clear.

3. SECONDARY SLUDGE

Table F2 shows the measured and predicted activity concentration in secondary sludge for Beckton and Knostrop. All of the nuclides that were detected are shown and limits of detection are given for nuclides that could not be measured. At Beckton, for all but Tc-99m and I-131, predicted concentrations from typical are greater than measured concentrations. Tc-99m measured is significantly higher than predicted. This could be explained if the sampled sludge had in fact only been stored for a short time, not the 3 weeks assumed in the model. The I-131 measurement was slightly larger than the predicted result for the authorised limited. Again, the same reason may apply. However, it does confirm that the concentrations predicted by the model are not significant over estimates for I-131. The measured C-14 concentration is a factor of 100 smaller than the predicted value, whereas it was only a factor of 10 smaller in the raw sewage. I-125 is also a factor of 100 small than predicted.

At Knostrop, all the predicted concentrations are significantly higher than those measured. For I-131 the measurements are about a factor of 10 smaller than predicted.

4. FINAL EFFLUENT

Table F3 shows the measured and predicted concentrations for final effluent for Beckton and Knostrop. At Beckton, all predicted concentrations are greater than measured concentrations, but only by a factor of 2 to 20. The model predicts a C-14 concentration a factor of 20 greater than the measurements and hence may overestimate doses from effluent disposal.

At Knostrop, predicted concentrations for all except I-131 are greater than measured concentrations. The I-131 measurements are up to a factor of 2 greater than predicted. This, together with the sludge results indicates that the model is not quite right.

5. INCINERATOR CAKE AND ASH

Tables F4 and F5 show the activity concentrations in the incinerator cake and ash. For Beckton most of the predictions of activity in incinerator cake are in good agreement, only C-14, Rb-84 and I-125 have been over estimated significantly. The predicted activities in ash are also in reasonable agreement with measurement, except for C-14 and Rb-84 which again have been over estimated. The activity concentration of Rb-84 has been over estimated throughout all stages of treatment, it can therefore be assumed that the authorisation to dispose Rb-84 is no longer used. A comparison of the activity concentrations in the cake and ash should give an

indication of the accuracy of the volatilising fractions used in SMART. They suggest that the volatilising fractions are quite accurate. Again, the measurements confirm that the model does not significantly over estimate the concentration of I-131 in cake or ash.

At Knostrop the predicted activity concentrations for all the nuclides are approximately 2 orders of magnitude higher than the measured values, in both the incinerator sludge and ash.

6. CONCLUSIONS

Agreement between the measured and predicted values appears good for most of the radionuclides. In particular, the measured concentrations of I-131 in sludge, cake and ash are very similar to predicted values, so supporting the dose calculations. Conversely, the model appears to overestimate the C-14 concentration in final effluent, so over estimating the dose to the public.

Table F1 Predicted and measured activity concentrations in Raw Sewage at Beckton

Radionuclide	Measured Activity Concentration Bq kg^{-1}	SMART Predicted Activity Concentration Bq kg^{-1}	
		Authorised disposals	Typical Disposals
C-14	0.22±0.09 , 0.2±0.1	3.70	3.15
P-32	11.1±4 , 191±16	0.42	0.13
Tc-99m	0.35±0.14 , 4.9±0.3	5.27	3.44
I-131	1.1±0.4 , 0.35±0.02	7.24	2.90

Table F2 Predicted and measured activity concentrations in secondary sludges at Beckton and Knostrop

(A)

Radionuclide	Beckton			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	1.2±0.4	-	198	169
P-32	8 ± 2	-	34.0	10.0
S-35	1.3±0.1	-	11.4	5.0
Cr-51	<1.2	<3	2.2	4.6
Ga-67	<0.9	-	1.2	0.7
Rb-84	<0.2	-	16.2	13.2
Tc-99m	1.6±0.2	<4	$1.9\ 10^{-14}$	$1.2\ 10^{-14}$
In-111	<0.1	<0.4	0.24	$2.3\ 10^{-3}$
I-123	<0.6	<1	$2.4\ 10^{-7}$	$1.4\ 10^{-8}$
I-125	0.13±0.01	-	70.0	18.0
I-131	43 ± 3	55 ± 7	38.9	15.6

(B)

Radionuclide	Knostrop			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	<9	-	810000	270
P-32	57 ± 22	-	a	6100
S-35	<30	-	a	192
Cr-51	0	<100	a	1300
Tc-99m	189± 24	1071±126	493000	160000
I-123	<3	<3	a	374
I-125	-	-	139000	6400
I-131	880±103	2363±276	135000 (b)	46000

Notes:

(a) Modelled as I-131 since it is included under 'other' radionuclides

(b) Includes 'other' radionuclides

Table F3 Predicted and measured activity concentrations in final effluent at Beckton and Knostrop

(A)

Radionuclide	Beckton			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	0.14± 0.08	-	2.6	2.2
P-32	<15	-	0.1	0.02
S-35	0.17± 0.06	-	0.6	0.3
Cr-51	<1	<1	0.004	$8.9\ 10^{-3}$
Ga-67	<0.8	-	0.03	0.015
Rb-84	<0.2	-	0.03	0.02
Tc-99m	0.4± 0.1	2 ± 1	1.5	1.0
In-111	<0.2	<0.1 , <0.2	$7.5\ 10^{-3}$	$7.1\ 10^{-5}$
I-123	<0.3	<0.2 , <0.6	0.1	$6.2\ 10^{-3}$
I-125	<0.05	-	4.1	1.1
I-131	0.52± 0.06	0.4±0.1 , 0.6±0.2	5.8	2.3

(B)

Radionuclide	Knostrop			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	<0.3	<0.1	23.9	0.008
P-32	<1	<0.2		0.03
S-35	<0.3	0.3 ± 0.2		0.02
Cr-51	<0.4-<1	0.6 ± 0.2		0.01
Tc-99m	3.6± 1.0	3± 1	70	23.3
I-123	<0.06-<0.3	<0.1-<0.2		0.05
I-125	<0.01	<0.06	15.8	0.7
I-131	18±2 , 31± 4	3.1±0.4 , 3.4±0.5	15.5	5.3

Table F4 Predicted and measured activity concentrations in incinerator cake at Beckton and Knostrop

(A)

Radionuclide	Beckton			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	50± 11	-	$1.6\ 10^{3}$	$1.4\ 10^{3}$
P-32	155± 60	-	260	76.7
S-35	-	-	90.8	40.3
Cr-51	10± 2	<13	17.2	36.1
Ga-67	5.6± 0.9	-	7.7	4.6
Rb-84	<0.4	-	127	104
Tc-99m	-	<20	$7.5\ 10^{-15}$	$4.9\ 10^{-15}$
In-111	1.0 ± 0.3	<2	1.5	0.01
I-123	<25	<6	4.3	0.3
I-125	0.33± 0.04	-	$1.0\ 10^{3}$	268
I-131	417± 21	326± 37	939	376

(B)

Radionuclide	Knostrop			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	1.0 ± 0.1	16± 8	$5.1\ 10^{3}$	1.7
P-32	<0.1	4.0 ± 0.2		36.6
S-35	<0.06	<4		1.2
Cr-51	<1	2 ± 2		8.1
Tc-99m	0.8± 0.5	<0.2	155	51.8
I-123	<0.2	<0.1		1.1
I-125	<0.1	<0.2	$1.5\ 10^{3}$	70
I-131	7± 1	13± 2	$1.4\ 10^{3}$	462

Table F5 Predicted and measured activity concentrations in incinerator ash at Beckton and Knostrop

(A)

Radionuclide	Beckton			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	<3	-	250	213
P-32	Lost	-	$4.0\ 10^{3}$	$1.2\ 10^{3}$
S-35	-	-	$1.3\ 10^{3}$	568
Cr-51	250 ± 17	48 ± 9.6	266	560
Ga-67	70 ± 6	-	93	55.3
Rb-84	<0.8	-	$2.0\ 10^{3}$	$1.6\ 10^{3}$
Tc-99m	-	-	$5.6\ 10^{-15}$	$3.6\ 10^{-15}$
In-111	6.5 ± 0.7	<3	9.0	0.08
I-123	<34	<359	3.3	0.2
I-125	Lost	-	817	210
I-131	608± 30	723± 40	734	294

(B)

Radionuclide	Knostrop			
	Measured Act. Conc. Bq/kg summer	Measured Act. Conc. Bq/kg winter	SMART Predicted Act. Conc. Bq/kg	
			Authorised disposals	Typical disposals
C-14	<8	<7	736	0.2
P-32	Lost	12 ± 1		523
S-35	<2	-		15.6
Cr-51	31 ± 9	47 ± 19		116
Tc-99m	3 ± 2	5 ± 1	106	35.3
I-123	<0.5	<0.7		0.8
I-125	<0.01	<0.06	$1.1\ 10^{3}$	50.6
I-131	7± 2	3± 2	980	333

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APPENDIX G

Studies of external dose rates at Beckton and Knostrop incinerators.

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1 INTRODUCTION

The estimated doses based on model predictions and measurements on feedstocks and waste streams indicated that the irradiation of workers at STWs could be an exposure pathway of particular importance. However, the measurement data was derived from two limited sampling programmes. Predictions of external doses based on relatively few experimental data inevitably contain uncertainties because of variations in activity concentrations in the feedstock over time. In order to determine if significant additional doses from sewage materials are occurring, an additional work programme was agreed, in which external dose rates were measured using thermoluminescent dosemeters (TLDs) at selected points within Beckton and Knostrop sewage works

2 MONITORING

Initial calculations in the study indicated that the areas where workers were likely to receive the highest external doses were in the incinerator facilities near to the sludge cake feedstock and residual ash. It was decided, therefore, to place ten sets of two TLDs in these areas at each site. One set was used to measure local background radiation, placed in the control room which was relatively remote from the sludge and ash. The other sets were placed in locations where operators spend time close to the sludge and ash. The incinerator facilities are described in Chapters 8 and 12 of the main report.

The TLDs detect external gamma dose from all radionuclides including natural background and cosmic rays. Background dose rates are expected to be significantly higher than any additional dose from sewage materials. The study was intended to indicate if significant external doses in addition to background were recorded, rather than a detailed study of excess dose.

TLDs were used because they are robust and can be left in the working environment for long periods. However, TLDs do not give a precise measure of external dose, the detection threshold for TLDs is 25 μSv, the standard deviation on a recorded dose of 250 is +- 25%. The response fade is less than 5% over 6 months. The excess dose over background can only be established by subtracting a measurement of background from the measurement of total dose. The measurements of total dose and background dose are both likely to be subject to an uncertainty in the TLD response (+- 25%), plus uncertainty in the background itself, which is related to the variability in the dose rate from different building materials (Table G1) that may be seen over the two measuring periods (2800 h at Beckton and 4500 h at Knostrop). The uncertainty in the background was assumed to be half the absolute range of background dose rate measurements.

The total uncertainty on the control TLD or the measuring TLDs can be calculated from

$$[\text{Totuncert}]^2 = [\text{TLDuncert}]^2 + [\text{BGuncert}]^2$$

The uncertainty on the excess dose, established by subtracting the dose measured by the control and the dose measured by any TLD was established from:

$$[\text{Exuncert}]^2 = [\text{Totuncert}]^2 + [\text{Totuncert}]^2$$

The excess dose must exceed the estimate of uncertainty to be considered measurable. Therefore the error on the excess dose has been assumed to be the limit of detection for the excess dose.

3 KNOSTROP

TLDs were placed at various accessible points around the incinerator plant: on the guard rails surrounding the belt presses (one metre from the edge of the sludge layer), on the cover of the conveyor that takes the thickened sludge to the incinerator, on the ash conveyor from the incinerator and under the ash hoppers. The TLDs were installed on 19th November 1998 and removed for measurement in March and May 1999. The maximum duration of exposure was 189 days.

The average dose detected by the ten detectors left in place for 189 days (4500 h), was 280 μSv, (+-200μSv) whilst the control TLD gave a reading of 390 μSv (+- 210 μSv) (Table G2). The difference between the total dose and background was –100 μSv (+-290 μSv).

This suggests that the doses recorded by the TLDs in place around the Knostrop incinerator were not significantly different to that recorded by the control. The results suggest that the TLD control was placed in a part of the works where background was lower. Table G1 indicates that variations in dose rate that can occur between different building materials.

Therefore, at Knostrop sewage works, excess dose from sewage material over 4500 h was less than the estimate of error (i.e. less than approximately 290 μSv), or, for a 2000 h exposure time, less than 129 μSv.

4 BECKTON

TLDs were attached to the guard rails which surround each filter press at about 0.5 m from the edge of the frame, near the main sludge pumps and near the cake conveyor under the incinerator. To estimate external dose from ash TLDs were installed on the control platform under the two ash hoppers. The TLDs were installed on 28th January 1999 and were removed in March and May 1999. The maximum duration was 117 days.

The average dose detected by the ten detectors left in place for 117 days, (2800 h) was 275 μSv (+-135μSv). The control TLD, which was not exposed to any radiation from radionuclides in sewage recorded 250 μSv (+-132 μSv) (Table G3). The difference between the control and the average was 25 (+- 134 μSv). The highest recorded dose was 380 μSv (+- 150 μSv), the highest excess dose being 130 μSv (+-200 μSv).

Therefore, at Beckton sewage works, the highest excess dose from sewage material (over 2800 h) was less than the estimate of error (i.e. less than approximately 200 μSv), or,

over a 2000 h exposure time, less than 142 μSv.

5 CONCLUSIONS

The TLD study shows that no doses could be measured at either works that were above the estimate of uncertainty on the results. Using the estimate of uncertainty as the limit of detection, the measured excess dose from authorised discharges of radionuclides are less than approximately 100 to 150 μSv at both Knostrop and Beckton.

6 REFERENCES

1. UNSCEAR, Sources and Effects of Ionising Radiations. 1977 Report to the General Assembly.

Table G1: Dose rates from various UK building materials[1]

Material type	Absorbed dose in air nGy/h	Effective dose rate nSv/h	Effective dose μSv per 2800 h	Effective dose μSv per 4500 h
Clay Brick	160	112	313	504
Concrete	150	105	294	473
Plaster	40	28	78	126
Absolute Range	120	84	235	378
Adopted Range			117	189

Table G2: Knostrop Sewage Works TLDs

Serial Number	Total Dose mSv (Knostrop)	Control corrected (mSv)
2939253 (c)	0.391	N/A
2939255	0.283	-0.108
2939257	0.273	-0.118
2939259	0.376	-0.015
2939261	0.337	-0.054
2939264	0.210	-0.181
2939267	0.206	-0.185
2939269	0.304	-0.087
2939271	0.264	-0.127
Average	0.282	N/A
1*Stdev	0.058	N/A
2*Stdev	0.116	N/A
2* St dev range	0.166 – 0.398	N/A
25% range	0.212 – 0.352	N/A

Notes:
(c) = Control TLD

Table G3: Beckton Sewage Works TLD results

Serial Number	Total Dose mSv (Beckton)	Control corrected (mSv)
2939481 (c)	0.250	N/A
2939465	0.213	-0.037
2939467	0.380	0.130
2939469	0.230	-0.020
2939471	0.274	0.024
2939473	0.280	0.030
2939475	0.298	0.048
2939476	0.254	0.004
2939478	0.271	0.021
Average	0.275	
1* Stdev	0.051	
2*Stdev	0.102	
2 Stdev range	0.173 - 0.377	
+-25% average	0.206 - 0.344	

Notes:
(c) = Control TLD

APPENDIX H

DESCRIPTION OF THE GREATER LONDON SEWAGE SYSTEM

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1 INTRODUCTION

This appendix provides a description of the general characteristics of the sewage collection, treatment and disposal systems of the major sewage works in London. This information was used to select one STW for further study. The criteria for selection were: the location and catchment area of the works and the throughput and disposal routes of effluents. In addition the number of establishments with authorisations to dispose radionuclides to sewers in the catchment of the sewage works was considered.

2 SEWAGE COLLECTION

Reference 1 describes the main features of the central London sewage collection system. The population covered by this part of the sewer network is about 50% of the total population of Greater London. These sewers are the largest and most densely packed. The run-off and effluents from central London are transferred to low level trunk sewers which run principally from east to west. Pumping stations assist the flow in these pipes. The principal pumping stations are at Abbey Mills, Greenwich and Western. Smaller outstations are controlled from these main pumping stations remotely.

Abbey Mills pumping station serves the 135 km^2 of the low lying districts north of central London, pumping sewage to the Beckton STW. This area includes Camden, Ealing, Islington Hackney, Hammersmith, Kensington and Chelsea, Newham, Tower Hamlets, Waltham Forest and the cities of London and Westminster. It has a capacity of 3.4 10^6 m^3 d^{-1}, and in storms excess flow is discharged directly into the Thames. Four trunk sewers, ranging in size from 2.7 to 3.5 m diameter, gravitate the sewage to the station. The pumping station raises the effluent by 12 m and it then travels by gravity the remaining 6 km to Beckton STW.

Greenwich pumping station serves central London south of the river, and feeds the Crossness STW. The catchment covers Wimbledon, Wandle Valley, Merton, and areas of Wandsworth, Lambeth, Southwark and Greenwich adjacent to the river. Two trunk sewers (about 2 m in diameter) feed the station by gravity. At the pumping station the sewage is raised 6 m into a single outfall which carries the sewage to Crossness. In storms excess flow is discharged to Deptford creek.

Western pumping station serves about 17 km^2 of west London including Hammersmith and Kensington and Chelsea. The sewage arrives in a trunk sewer 2.1 m in diameter and is raised 6 m by pumps with a capacity of 1.2 10^6 m^3 d^{-1}. The effluent then flows by gravity to Abbey Mills.

3 SEWAGE TREATMENT

3.1 Overview

A total of 13 sewage treatment works (STW) of varying size serve 9.6 million population equivalent (p.e.)[1] in the Greater London area. Twelve of the works are described in detail below. One of the works, Kew STW which is due to close is not described. The catchments of the twelve works are shown in Figure 8 in the main report. The smallest works (Blackbirds in North West London) serves 100,000 p.e., whilst Beckton, which covers North

1 Population equivalents include notional population to account for industrial discharges.

Central London, serves 3 million p.e. Information from References 2 and 3 have been used to show the important characteristics of these sewage treatment works. A summary of these data have been presented in Table 20 and Figures 9 - 11 in the main text.

The following sections are based on information supplied by Thames Water Utilities.

3.2 Beckton

Beckton is the largest sewage works in London and the UK, and amongst the largest in Europe. It is situated on the western edge of Barking, next to the Thames, and serves a catchment of about 3 10^6 p.e. (31 % of the total served by Thames Water) north of the river Thames in central London. The catchment covers Hammersmith and Fulham and extends through Islington to Woodford and Ilford. The present consented flow rate of Beckton is 2.7 10^6 m^3 d^{-1}, which is considerably larger than the consented flow in 1978. The sewage collects at the Abbey Mills pumping station and then enters Beckton, where it passes through sedimentation tanks and activated sludge plant. The final effluent is discharged to the Thames tideway at about 90 % of the influent flow rate. The peak outfall flow rate is in the early afternoon. Typical suspended solids loads in out flowing sewage are about 17 mg l^{-1}, with a maximum of 130 mg l^{-1}.

Historically the sewage sludges were thickened to about 4.5 % solids and mixed with surplus activated sludge (about 6 % solids).These sludges were then processed by anaerobic digestion for about 3 weeks. The sludges were then passed to lagoons for secondary digestion (a tertiary treatment) and then disposed of in coastal waters at Barrow Deeps, in the outer Thames Estuary. On average there were four sailings per day, each carrying about 2200 wet tonnes with about 2 % solids. However, these sludge conditioning processes were replaced by incineration in 1998.

For incineration, the sewage sludge is thickened to about 32 % and incinerated. Normally, the time between the start of the sewage treatment process and incineration is 2-4 days. The incinerator is expected to handle 7.0 10^4 tonnes of dry solid (tds) y^{-1} of sludge. Around 20% of the sludge remains as ash, which will be sent to a landfill at Beddington.

3.3 Mogden

West London from Sunbury and Staines to Uxbridge, Harrow and Edgeware is served by the Mogden works. The population served, 1.7 10^6 p.e., is second only to Beckton and represents some 17% of the population of London. Mogden has not grown as much as some of the other London works: the consented flow of 4.2 10^5 m^3 d^{-1} is slightly less than that in 1978. Mogden discharges around 88% of this flow into the Thames Tideway. The average suspended solids load of the discharge is 18 mg l^{-1}, with a maximum of 70 mg l^{-1}.

Historically, sewage sludge from Mogden was disposed of on land and at sea. In 1978 5,538 tonnes of dry solid (tds) y^{-1} was used in agriculture and 9,205 tds y^{-1} was dumped in the sea. Currently all sludges are dewatered by centrifuging to a cake (23% solids) following digestion at 30-35 °C for about 20 days, and then disposed of to land.

3.4 Crossness

The Crossness works has a catchment that includes a substantial area of London south of the river Thames. It includes areas such as Putney and Wimbledon in the west and Catford and Eltham in the east by way of Wandsworth, Lambeth and Lewisham. It serves 16 % of the population of London, some 1.5 million p.e. Since 1978, the consented flow has increased by 1.6 times to 9.8 10^5 m^3 d^{-1}. The incoming sewage has a typical quantity of suspended solids

of about 290 mg l^{-1} which is reduced to about 28 mg l^{-1} before discharge at a rate of 89 % the inflow rate into the Thames tideway.

In the past the resulting sludge has been discharged to sea. However, a sewage sludge incinerator became operational in 1998 to burn the sludge. It is estimated that the throughput will be around 5.0 10^4 tds y^{-1} of sludge with a solids content of 32%. It is expected that the incinerator will produce about 13,505 t y^{-1} of ash, which will be disposed of to landfill.

3.5 Long Reach

Covering in the south east of London, the catchment of Long Reach STW includes Dartford and Blexley and runs south to Orpington and Sevenoaks. It is amongst the larger works in London, serving 8.4 10^5 p.e., which is about 9 % of London's population. This station has not significantly increased throughput since 1978. The final effluent (around 1.9 10^5 m^3 d^{-1}, 93 % of the influent) is discharged into the Thames tideway. The average suspended solids at the outflow of the works was around 34 mg l^{-1} for 1997/1998, the maximum recorded values being 400 - 690 mg l^{-1}.

Because of its position on the edge of London, the sewage sludge produced is sent to agricultural land. These sludges are first treated by mesophilic digestion and then disposed (at a rate of 2.1 10^6 wet t y^{-1}) to agricultural land. The solids content on disposal is 3 - 4 %.

3.6 Deephams

The Deephams works serves a significant area North of the Beckton catchment. It is bordered by Barnet, Tottenham and Enfield. A small finger of the catchment extends north to include Cheshunt. Deephams processes sewage from 8.0 10^5 p.e., 8% of the London population, and has a consented flow of 2.0 10^5 m^3 d^{-1}. This flow is roughly the same as that in 1978 indicating that there has not been significant change of catchment. The effluent is discharged at a typical rate of 2.0 10^5 m^3 d^{-1} to the river Lee. The EA consented suspended solids content of the discharges is currently 30 mg l^{-1}. The average suspended solids content for discharges in 1997-1998 was 9.6 mg l^{-1}, the maximum recorded value being 67.6 mg l^{-1}.

Deephams STW used to send its sludge to Beckton, however it is now digested, pressed and sent to land. Around 655,200 wet t y^{-1} (or about 52,000 tds y^{-1}) are processed for 18 days followed by a further 8 weeks storage. The solids content is increased to 18 - 22 % before the sludge is the sent to land.

3.7 Maple Lodge and Blackbirds

The large catchment area in the North West of London is served by two STWs, Maple Lodge and Blackbirds. The two works are connected and some of the sewage flow to Blackbirds STW can be diverted to Maple Lodge STW for operational reasons if required. The catchment includes Watford and Borehamwood in the south, and Hemel Hempstead and St Albans in the north. The Maple lodge works has about 5 times the capacity of Blackbirds, and the two combined serve 6.0 10^5 p.e. Both Maple Lodge, with a consented flow of 1.3 10^5 m^3 d^{-1} and Blackbirds (1.2 10^5 m^3 d^{-1}) discharge into the river Colne, a tributary of the Thames. For Maple Lodge, the consented suspended solids of the effluent is 25 mg l^{-1}, which will be reduced to 15 mg l^{-1}. The discharges in 1997-1998 were on average 9.8 mg l^{-1}, peaking at 29.3 mg l^{-1}. Currently, the consented suspended solids content of discharges from the Blackbirds works is 10 mg l^{-1}. In 1997-1998 the recorded discharges averaged 3.6 mg l^{-1} with a maximum of 10.2 mg l^{-1}.

All sludge produced by Blackbirds is treated at the Maple Lodge works. Here it is

digested for 22 days and stored for a further 14. The total throughput is around 42,000 tds y^{-1} and the resulting sludge (20-21% solids) is sent to land.

3.8 Riverside

This is the easternmost catchment in London and includes areas of Dagenham, Romford and Rainham. The Riverside STW is fairly large and serves 4.3 10^5 p.e. customers. The consented flow rate is 2.5 10^5 m^3 d^{-1}, which is around 1.4 times the flow rate in 1978. A significant amount of the sewage is industrial effluent, therefore the inflowing effluent has a higher suspended solids load than is typical (about 400 mg l^{-1} against 300 mg l^{-1}).The effluent is discharged to the Tideway at about 93% the flow rate of the inflowing effluent. The suspended solids load is reduced to around 10-20 mg l^{-1} (average for 1997-1998) on discharge. However, one peak value for this period exceed 800 mg l^{-1}, although other high recorded values were less than about 330 mg l^{-1}.

Most of the sludge produced was previously sent to Beckton for disposal at sea. It is now sent to Beckton for incineration, at a rate of 358,430 wet t y^{-1}.

3.9 Beddington

On the southern edge of London, Beddington STW has its northern edge at southern Stretham, and extends through part of Croydon to Caterham. Beddington is relatively small and serves 3.2 10^5 p.e. The throughput of the works has increased by over 250% since 1978 to reach a consented flow of 8.6 10^4 m^3 d^{-1} (DWF). After processing, 7.6 10^4 m^3 d^{-1} of final effluent is discharged into the river Wandle, a tributary of the Thames. The consented suspended solids load for the effluent is 30 mg l^{-1}. In 1997-1998 the average concentration discharged was 5-11 mg l^{-1}, with peak values ranging from 19 to 30 mg l^{-1}. In a single instance the discharged solids was recorded at 180 mg l^{-1}.

Sludge is produced at a rate of 237,250 wet t y^{-1} at 3 - 4 % dry solids. It is digested for 24 days and then currently stored in lagoons for 3 - 6 months. In the future the lagoon storage will be removed and replaced by dewatering. Sludges are currently disposed of to farmland around Beddington at 6% dry solids. In the future, the dewatered cake (24 % solids) will be sent to the same general area.

3.10 Worcester Park

To the west of Beddington, Worcester Park STW serves Sutton and Worcester Park. The catchment is long and thin in the North-South direction. For London, Worcester Park is a small works which only serves about 1 % of the population (1.2 10^5 p.e.). However the quantity of sewage handled by the station has increased dramatically from a consented maximum flow of 1.6 10^4 to 7.1 10^4 m^3 d^{-1}. Around 80 % (by volume) of this sewage is discharged into Beverly Brook having been treated to reduce the suspended solids to an average 23 mg l^{-1} (with a peak value of 95 mg l^{-1}) in 1997 - 1998. The works produces 65,700 wet t y^{-1} at 2 % solids content. This raw sludge is taken to the Mogden works for treatment. Worcester Park STW closed in early 1999 and the sewage is diverted to the Hogsmill works. Some treated effluent is pumped back into Beverly Brook as a way of compensation flow.

3.11 Hogsmill

The Hogsmill STW catchment meets the western edge of Worcester Park's catchment. It runs from Kingston down through New Malden and Ewell to Epsom. The works has about twice the capacity of Worcester Park but the throughput has not grown significantly since

1978. At 8.6 10^4 m^3 d^{-1} the consented maximum flow is now larger than in 1978. Only 70 % of the inflowing volume remains to be discharged into the Hogsmill river with an average suspended sediment load of around 9 mg l^{-1} and a peak value of 68 mg l^{-1} (for 1997-1998). The consented limits for suspended solids in effluent is currently 50 mg l^{-1}, but is likely to reduce to 25 mg l^{-1}.

The total quantity of sludge produced amounted to 219,000 wet t y^{-1} in 1997 - 1998. This value is likely to drop slightly in the future. After processing, Hogsmill sends its sewage sludge to agricultural land as its position is amenable to this route. It is digested for 20 days and dewatered to a final solids content of 20 %.

3.12 Esher

The catchment for Esher STW loops around the Island Barn and Queen Elizabeth II reservoirs. It includes Walton on Thames and Esher and is bordered by Hampton and Teddington. It is a small works, by London standards, covering a population of 1.1 10^5 p.e., about 1 % of the total population served in Greater London. The consented flow is 3.5 10^4 m^3 d^{-1}, which is discharged into the river Mole with a suspended solids content of on average 10 mg l^{-1} for 1997 - 1998, with a maximum concentration of 17 mg l^{-1}. The consented suspended solids content of effluent is 25 mg l^{-1}.

The Esher works produces about 2,700 tds y^{-1} of sewage sludge. This is retained for 14 days without treatment. It is then disposed of to land at a solids content of 4 - 6%.

4 CHOICE OF STW FOR FURTHER STUDY

The review has shown that all sewage treatment works in London are large. However, the sewage collection and treatment system in London is dominated by three works, Beckton, Crossness and Mogden. These together serve ⅔ of the population, account for ¾ of the total sewage processing capacity and produce ⅔ of the sewage sludge. Of the three, Beckton is the largest, serving a third of the population of greater London.

The review has also shown that there are significant differences in the sewage sludge treatment and disposal methods. However, two main options are apparent: the sludge is anaerobically digested and sent to land, or it is anaerobically digested and incinerated. The former option is dependant on the availability of farmland, the sewage works on the edges of London have easier access to land and can therefore exploit this disposal route. The incineration option replaces the sea dumping option which was used by the more central works in the past: the relatively easy access to the North Sea via the Thames Estuary was used by Crossness, Beckton, Riverside and Deephams STW. However, disposals of sludge at sea were phased out by the end of 1998 and this has been replaced by on site incineration of the sludges. The trend towards incineration appears to be common to most large sewage treatment works that previously sent sewage sludge to sea.

Beckton STW dominates in terms of population served and quantity of sewage treated. This sewage works serves a large section of Central London and has a number of research establishments and hospitals using radionuclides for medical diagnosis and treatment. Beckton STW is also fully urban, and the works has recently switched from disposing sewage sludge to the North Sea to incineration of sludges. Beckton works was therefore selected as the most appropriate works for more detailed study because of its size, location, method of disposal of sludges and the number of organisations with authorisations

to dispose radionuclides in its catchment area.

5 REFERENCES

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APPENDIX I

GLOSSARY

Glossary

Absorbed dose: Quantity of energy imparted by ionising radiation to unit mass of matter such as tissue. Unit Gray. Symbol Gy. 1Gy = 1 Joule per Kilogram.

Activity: Attribute of an amount of a radionuclide. Describes the rate at which transformations occur in it. Unit Becquerel. Symbol Bq. 1Bq = 1 transformation per second.

Adsorption: Covers a variety of reactions of which particles of another substance are held onto the surface of a solid.

Alpha particle: A particle consisting of two protons and two neutrons.

Authorised limits: The maximum quantity of a particular radionuclide that a premises has consent to discharge into the sewage system.

Beta particle: An electron emitted by the nucleus of a radionuclide. The electric charge may be positive, in which case the beta particle is called a positron.

Biochemical Oxygen Demand (BOD): A standard water-treatment test for the presence of organic pollutants. A measure of the oxygen required by the microbes which reduce the wastes to simple compounds.

Collective effective dose: The quantity obtained by multiplying the average effective dose by the number of people exposed to a given source of ionising radiation. Unit man Sievert. Symbol man Sv. Frequently abbreviated to collective dose.

Combined sewer: Sewers dating back to the last century where foul water and industrial effluents are carried together with storm run-off, in a single sewer. They are inefficient at dealing with run-off during storms and may discharge untreated wastewater into the environment.

Disposal: In relation to radioactive waste, dispersal or emplacement in any medium without the intention of retrieval.

Dose: General term for quantity of ionising radiation. Frequently used for effective dose.

Dry weather flow: Usual measure of the average volumetric flow rate of a sewer which does not include any contribution from surface run-off.

Effective dose: The quantity obtained by multiplying the equivalent dose to various tissues and organs by a weighting factor appropriate to each and summing the products. Unit Sievert, symbol Sv.

Electron capture: The process in which an electron from an inner orbit is captured by the nucleus, resulting in the conversion of a proton to a neutron and the emission of X-rays.

Equivalent dose: The quantity obtained by multiplying the absorbed dose by a factor to allow for the effectiveness of the various ionising radiations in causing harm to tissue. Unit Sievert, symbol Sv.

Gamma ray: A discrete quantity of electromagnetic energy without mass or charge.

Generalised Derived Constraints (GDC): Estimates of the amount of activity discharged into sewers which, if not exceeded, indicate that nobody should receive an effective dose above the maximum dose constraint of $0.3mSvy^{-1}$.

Gray: See absorbed dose.

Half-life: The time taken for the activity of a radionuclide to lose half its value by decay. Symbol $t_{1/2}$.

Ionisation: The process by which a neutral atom or molecule acquires or loses an electrical charge. The production of ions.

Ionising radiation: Radiation that produces ionisation in matter. Examples are alpha particles and beta and gamma rays.

Isotope: Nuclides with the same number of protons but different numbers of neutrons.

K_d: Equilibrium distribution coefficient, a measure of the sorption of radionuclides onto solids. Represents the activity concentration on the solid phase (in $Bq\ t^{-1}$) divided by the concentration in the aqueous phase (in $Bq\ m^{-3}$), at equilibrium. Units of $m^3\ t^{-1}$

Man Sievert: See collective effective dose.

Maximum permissible exposure: The irradiance likely to cause detectable damage to the human eye or skin from exposure. Units of Watt per square metre, symbol Wm^{-2}.

MOIRA: Model developed for radionuclide migration in urban environments and drainage systems by the Spanish team of EC MOIRA.

Nuclear medicine: Term usually applied to the use of radionuclides for diagnosing or treating disease in patients.

Preliminary sewage treatment: This involves screening and grit removal. Sewage arriving at the works is passed through coarse and fine screens to remove large debris such as sticks, cans, rags and bricks. It is then passed through channels at a speed that allows any grit suspended to sink to the bottom and be taken out.

Primary sewage treatment: This occurs after preliminary sewage treatment, the purpose of this treatment is to remove as much of the solid matter from the sewage as possible. The sewage flows into large tanks where the solids (crude sludge) settle to the bottom. The crude sludge is removed regularly for treatment and disposal.

Radiation: The process of emitting energy as waves or particles. The energy thus radiated.

Radioactive: Possessing the property of radioactivity.

Radioactive waste: Useless material containing radionuclides.

Radioactivity: The property of radionuclides of spontaneously emitting ionising radiation.

Radionuclide: An unstable nuclide that emits ionising radiation.

Removal efficiencies: The removal of radionuclides from sewage during treatment processes. It is a partitioning coefficient giving the fraction of a radionuclide's activity that is removed with sludge and the fraction remaining in the effluent.

Risk factor: The probability of cancer and leukaemia or hereditary damage per unit effective. Unit Sv^{-1}.

Secondary sewage treatment: The further reduction of BOD in the supernatant liquor from primary sewage treatment, by bacterial action.

Separate sewer: Modern sewers where foul water and industrial effluent is carried separately from storm run-off. Run-off can be discharged directly into rivers without potentially serious effects.

Sewage effluent: Wastewater with a low concentration of suspended solids originating from domestic homes, factories, hospitals etc. and road run-off.

Sewage sludge: All solid matter removed from sewage effluent, excluding that which is removed during preliminary treatment.

Sievert: See effective dose.

SMART: Sewer Model for the Assessment of Radioactive Transport.

Stabilisation: Treatment of sludge to destroy pathogenic organisms and reduce the water content.

Tertiary sewage treatment: The removal of nutrients, remaining suspended matter, heavy metals and micro-organisms.

Typical disposals: The actual quantity of a particular radionuclide that was disposed of into the sewage system in 1997.

Volatilised fractions: Partitioning coefficients used to describe the behaviour of radionuclides during incineration. This is the fraction of a radionuclide that is released to atmosphere, the remainder is left in the ash.

Wet weather flow: Measure of volumetric flow including surface run-off during storm episodes.