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Intercomparison of Sewer Models

Report to the National Dose Assessments Working Group (NDAWG)

S Watson, M P Harvey, J G Titley and P Kennedy

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ABSTRACT

This report describes a project set up for the National Dose Assessment Working Group (NDAWG) to compare models used by the Environment Agency (EA), the Food Standards Agency (FSA) and the Health Protection Agency (HPA) to assess radiation doses from releases of radioactivity to sewer systems. The aim was to analyse estimated doses and the input data used in each model for a unit release to a sewage system, in order to determine how great the differences between models are and to identify the key issues that lead to these differences.

The HPA would like to thank the Environment Agency, the Food Standards Agency and the Scottish Environment Protection Agency for their contributions to this work.

Report Version 1

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1 INTRODUCTION

Radiation dose assessments are carried out as an input to the process of granting an environmental permit or authorisation to allow discharge of radioactive material to the environment. Some facilities are authorised to release liquids containing radionuclides directly to the municipal sewage system and these usually pass through sewage treatment works (STW). Assessments of doses from discharges to sewer generally include the calculation of doses to sewer pipe maintenance workers and workers at the sewage treatment works. Other groups of people may also be considered in the assessment such as farmers who may be exposed as a result of the application of sewage sludge to agricultural land.

In the spring of 2009 a project was started under the auspices of the National Dose Assessment Working Group (NDAWG) to compare sewer models used by different agencies: the Environment Agency (EA), the Food Standards Agency (FSA), and the Health Protection Agency (HPA). The aim was to look at the doses estimated by the models and the input data used in each model (EA, 2006a; Brownless and Round, 2000; Titley et al, 2000), in order to determine how great the differences were and to identify what the key issues are that lead to these differences. The Scottish Environmental Protection Agency (SEPA) also uses a model (SEPA, 2008) to calculate doses from discharges to sewers that is broadly similar to the EA model but was not able to participate in this comparison exercise.

The results of this intercomparison exercise were presented at the NDAWG meeting in November 2009. At that meeting members of NDAWG agreed that more work should be carried out. The three agencies agreed to compare results from new model runs for a small number of radionuclides. It was also agreed that the model developed by the HPA to assess doses from releases to sewers by small users (McDonnell, 2004), which had not been considered in the original exercise, should be included. This model will be referred to as the HPA W63 model in this report, from the serial number of the HPA report which describes it. The results of the new intercomparison exercise carried out in 2010 are presented in this report, together with a detailed analysis of the different models used to determine where the differences arose.

2 SUMMARY OF THE OUTCOME OF THE 2009 INTERCOMPARISON EXERCISE

For the 2009 intercomparison exercise, each participating agency was asked to perform an assessment of doses from a unit discharge (1 Bq y^{-1}) of a number of radionuclides, using a standard flow rate into the sewage treatment works (STW) of 1000 m^3 d^{-1}. Calculations were carried out for a range of radionuclides and results were compared for the 28 radionuclides common to all three models. Doses were estimated for three groups: maintenance workers of large sewer pipes, workers at the sewage treatment works and a farming family receiving doses from the application of sewage sludge to land. The pathways considered for the pipe maintenance workers and the STW workers were external exposure to, and inhalation and inadvertent ingestion of, sewage material. For the farming family, doses from external exposure, ingestion of food grown on land treated with sewage sludge and inadvertent ingestion and inhalation of soil were considered.

The results of this intercomparison exercise, presented at the NDAWG meeting in November 2009, showed that differences in the doses calculated were generally significant, although for some radionuclides and pathways the differences were quite small. It was difficult to determine why those differences arose, particularly because there had been some misunderstanding about what was required and how to set the model parameters to the agreed values. In addition some computational errors were made due to the low discharge rate assumed in the exercise. Therefore NDAWG agreed that a new intercomparison exercise, using a higher discharge rate, should be carried out in early 2010.

3 THE 2010 INTERCOMPARISON EXERCISE

In order to minimise the time required to carry out the new exercise it was agreed that only a limited set of radionuclides should be included. Six of the more commonly discharged radionuclides to public sewer were selected for the exercise. These radionuclides have a range of sewage partitioning factors and significant exposure pathways and are shown in Table 1.

Table 1: Radionuclides used in the January 2010 exercise

Radionuclide	Half life (h)	Dominant pathway	Partitioning
^{3}H	$1.08\ 10^{5}$	Inhalation/ingestion	Mostly stays in effluent
^{32}P	$3.43\ 10^{2}$	Inhalation/ingestion	Mostly goes to sludge
^{99m}Tc	6.02	External exposure	Mostly stays in effluent, good agreement in 2009 comparison
^{111}In	$6.79\ 10^{1}$	External exposure	Mostly goes to sludge
^{131}I	$1.93\ 10^{2}$	External exposure	Mostly stays in effluent
^{201}Tl	$7.30\ 10^{1}$	External exposure	Even split between effluent and sludge

It was agreed that an input flow rate of 1000 $m^3\ d^{-1}$ would be used and that the discharge rate would be increased from 1 Bq y^{-1} to 1 GBq y^{-1}. These are realistic values and ensured that computational errors due to handling very small numbers were avoided. All other parameters were to be set by the participating agencies as would normally be assumed in their assessments.

As in the 2009 exercise, doses were calculated for three groups of people: maintenance workers in large sewer pipes, workers at sewage treatment works and a farming family. Doses were calculated for the same exposure pathways as in the 2009 exercise. Doses from inhalation and ingestion of sewage material or soil were calculated separately but were combined for comparison purposes.

Not all the models included in the comparison exercise considered all these groups and exposure pathways. Table 2 shows the scenarios, groups and exposure pathways considered by the models, not all of which were included from the intercomparison exercise. The main reason why models consider different exposure pathways is the remit and statutory function of the agency. For example, the FSA has a primary goal of protecting the public from excessive exposure via the food chain. The FSA therefore ensures that ingestion of food is included in its model, but does not focus on pathways such as external exposure from river sediment. Additionally, while the EA and HPA models do include ingestion of food, they do not include the wide range of foodstuffs that are considered in the FSA model. The EA model does not estimate doses to sewer pipe workers because it considers that doses to these workers are lower than those to other groups of people.

Only the FSA model was able to provide estimates of doses from all exposure pathways to all groups and for all specified radionuclides. Therefore, in some cases the doses were compared between only two or three models and not all radionuclides were considered.

Table 2: Scenarios considered by the models

Scenario	Exposed group	Exposure pathway	EA Model	FSA Model	HPA SMART	HPA W63
Maintenance of large (2 m diameter) sewer pipe	Maintenance workers	External exposure	×	✓	✓	×
		Inhalation of sewage material	×	×[*]	✓	×
		Inadvertent ingestion of sewage material	×	✓	✓	×
STW	STW workers	External exposure	✓	✓	✓	✓
		Inhalation and inadvertent ingestion of sewage material	✓	✓	✓	✓
Application of sewage sludge to farmland	Farmer or farming family	External exposure	✓	✓	✓[#]	✓[†]
		Inhalation of soil	✓	✓	✓[#]	✓[†]
		Inadvertent ingestion of soil	✓	×	✓[#]	✓[†]
		Ingestion of food	✓	✓	✓[#]	✓[†]
Blockage of small sewer pipe[‡]	Maintenance workers	External exposure	×	×	×	✓
		Inhalation and inadvertent ingestion of sewage material	×	×	×	✓
Discharge of effluent to brook or river[‡]	Individual spending time by river and/or eating fish and/or eating produce irrigated with river water	External exposure	✓	×	✓	✓
		Inhalation and inadvertent ingestion of sediments	✓	×	×	×
		Ingestion of brook/river water	✓	×	✓	✓
		Ingestion of freshwater fish	✓	✓	✓	✓
		Ingestion of food irrigated by river water	✓	✓	✓	✓
		Inhalation of soil from land irrigated by river water	✓	✓	×	×
		External from land irrigated by river water	×	✓	×	×
Discharge of effluent to coastal water[‡]	Individual with high beach occupancy, who eats marine fish, crustaceans and molluscs.	External exposure from marine sediment	✓	×	✓	×
		External exposure from handling fishing gear	×	×	✓	×
		Ingestion of marine fish	✓	×	✓	×
		Ingestion of shellfish and seaweed	×	×	✓	×
		Inhalation of sea spray	×	×	✓	×

Notes:

[*]: The FSA model only considers inhalation of tritiated water in the sewer pipe

[#]: Doses to the farming family are calculated by the model SLUDLAND using activity concentrations in sewage sludge predicted by the SMART model

[†] The HPA W63 model only calculates a total dose to the farming family

[‡] Scenario not included in this comparison exercise

4 RESULTS AND DISCUSSION OF THE 2010 INTERCOMPARISON EXERCISE

This section presents the doses estimated in the 2010 comparison exercise, discusses differences in dose estimates to the three groups of people considered in this exercise, and examines how the methodologies used may affect the calculations. The influences of those parameters that directly affect the people exposed and are radionuclide independent, such as habit data and solids loading, is also investigated. Radionuclide specific parameters are discussed separately in Section 5.

4.1 Doses to sewer pipe maintenance workers

As shown in Table 2, only the FSA and HPA SMART models calculate doses to maintenance workers of large sewer pipes. Although it can be difficult to draw conclusions from comparing just two models, the doses are shown in Table 3. Doses for each exposure pathway as well as total doses are shown in Figures 1, 2 and 3.

The largest differences between models were found in the doses for ^{201}Tl and ^{111}In. The ratios of highest to lowest estimated doses are similar to those produced in the 2009 comparison exercise. Doses from external exposure provide the most significant contribution to the doses to sewer pipe maintenance workers, apart from ^{3}H, for which external doses were not estimated. The FSA model estimates significantly higher doses from external exposure to sewer pipe maintenance workers than the HPA SMART model. Doses calculated by the FSA model are greater by a factor of between 2 and 5 orders of magnitude than those calculated by the HPA SMART model. The HPA SMART model estimates higher doses to sewer pipe maintenance workers than the FSA model for inhalation and ingestion of sewage material for ^{3}H and ^{99m}Tc. For other radionuclides the FSA model gives higher values. As for external dose, the highest differences of almost three and four orders of magnitude between highest and lowest estimates occur for ^{111}In and ^{201}Tl. while for other radionuclides the ratios are between a factor of about 2 and less than 50.

Table 3: Doses per unit discharge (Sv y^{-1} per GBq y^{-1}) to sewer pipe maintenance workers working in large (2 m diameter) pipes (Feb 2010)*

Doses from external exposure (Sv y^{-1} per GBq y^{-1})			
Radionuclide	FSA model	HPA SMART model	Ratio high to low
^{3}H	0.0	0.0	N/A
^{32}P	0.0	$1.1\ 10^{-10}$	N/A
^{99m}Tc	**$1.5\ 10^{-9}$**	**$5.3\ 10^{-12}$**	$2.7\ 10^{2}$
^{111}In	**$8.1\ 10^{-6}$**	**$7.0\ 10^{-9}$**	$1.2\ 10^{3}$
^{131}I	**$5.7\ 10^{-7}$**	**$1.9\ 10^{-9}$**	$3.0\ 10^{2}$
^{201}Tl	**$1.6\ 10^{-5}$**	**$1.3\ 10^{-10}$**	$1.3\ 10^{5}$
Combined doses from inhalation and ingestion of sewage material (Sv y^{-1} per GBq y^{-1})			
Radionuclide	FSA model	HPA SMART model	Ratio high to low
^{3}H	**$8.8\ 10^{-15}$**	**$4.0\ 10^{-13}$**	$4.6\ 10^{1}$
^{32}P	**$1.8\ 10^{-9}$**	**$5.3\ 10^{-11}$**	$3.5\ 10^{1}$
^{99m}Tc	**$2.3\ 10^{-13}$**	**$4.9\ 10^{-13}$**	2.2
^{111}In	**$4.8\ 10^{-9}$**	**$6.5\ 10^{-12}$**	$7.5\ 10^{2}$
^{131}I	**$2.8\ 10^{-8}$**	**$4.9\ 10^{-10}$**	$5.7\ 10^{1}$
^{201}Tl	**$1.5\ 10^{-8}$**	**$2.1\ 10^{-12}$**	$6.9\ 10^{3}$
Total doses (Sv y^{-1} per GBq y^{-1})			
Radionuclide	FSA model	HPA SMART model	Ratio high to low
^{3}H	**$8.8\ 10^{-15}$**	**$4.0\ 10^{-13}$**	$4.6\ 10^{1}$
^{32}P	**$1.8\ 10^{-9}$**	**$1.7\ 10^{-10}$**	$1.1\ 10^{1}$
^{99m}Tc	**$1.5\ 10^{-9}$**	**$5.8\ 10^{-12}$**	$2.5\ 10^{2}$
^{111}In	**$8.1\ 10^{-6}$**	**$7.0\ 10^{-9}$**	$1.2\ 10^{3}$
^{131}I	**$6.0\ 10^{-7}$**	**$2.4\ 10^{-9}$**	$2.5\ 10^{2}$
^{201}Tl	**$1.6\ 10^{-5}$**	**$1.3\ 10^{-10}$**	$1.2\ 10^{5}$

Note:

*: Maximum doses are given in **bold**; minimum doses are given in **bold red**

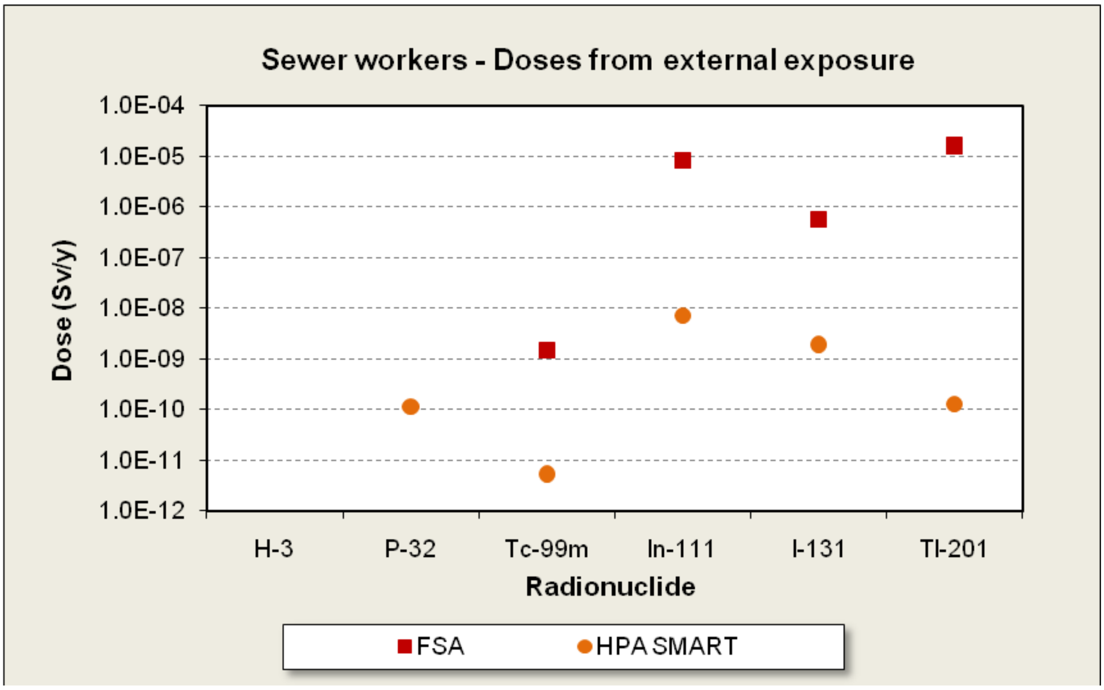

Figure 1. Doses to sewer pipe maintenance workers from external exposure

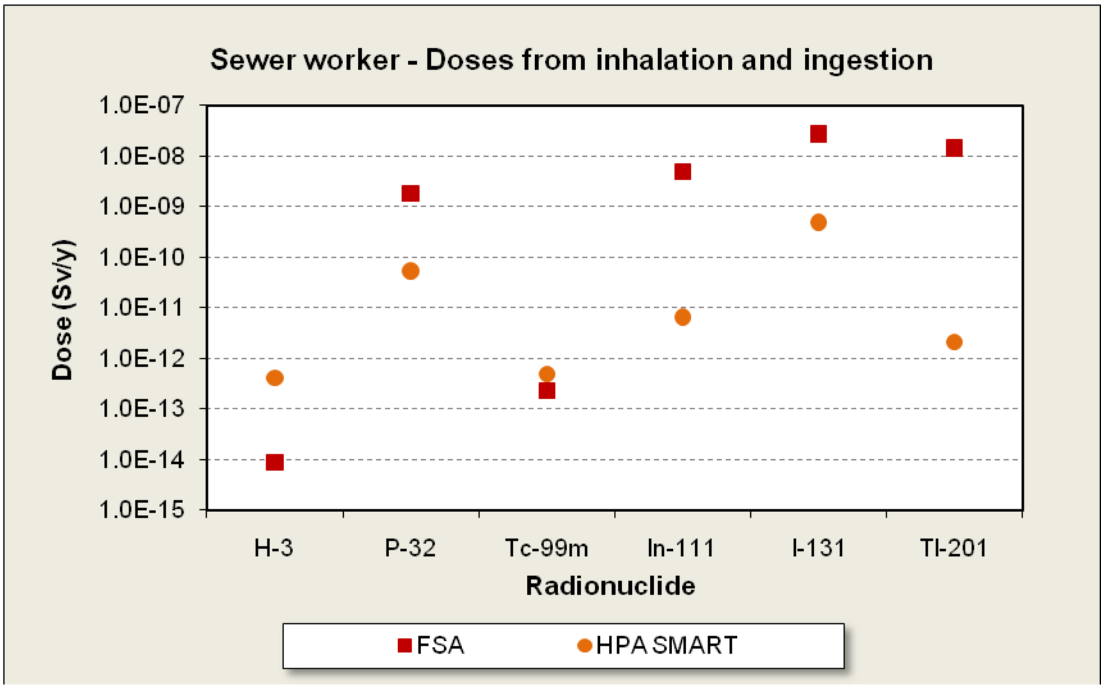

Figure 2. Combined doses to sewer pipe maintenance workers from inhalation and inadvertent ingestion doses of sewage material

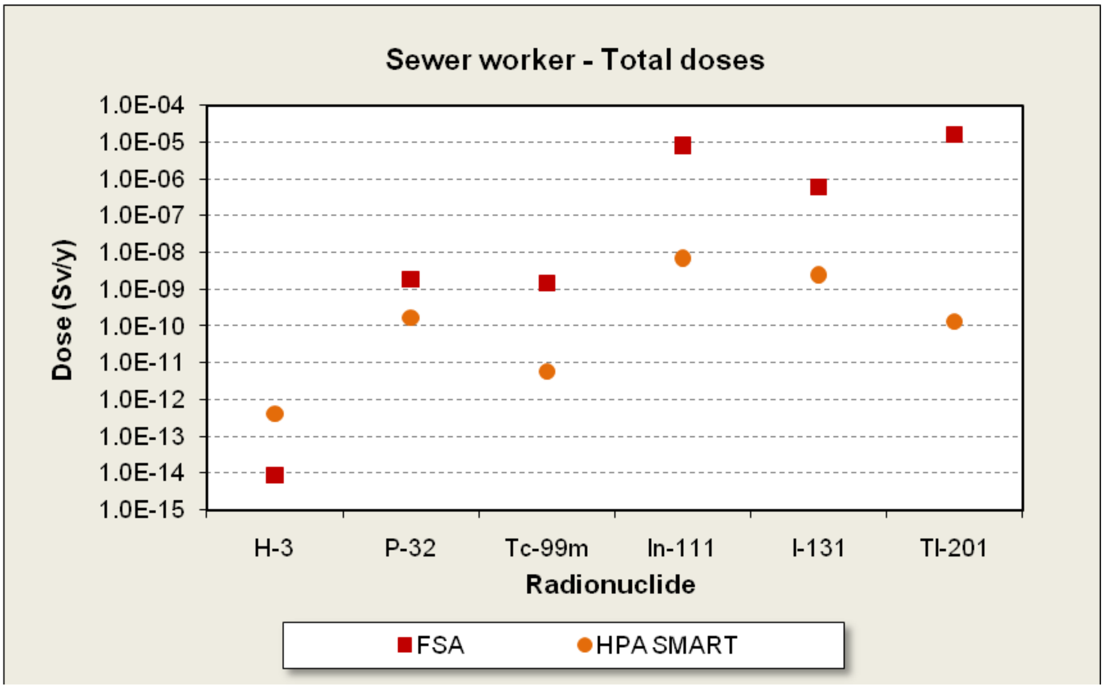

Figure 3. Total doses to sewer pipe maintenance workers

4.1.1 Analysis of the methodologies used in the calculation of doses to sewer pipe workers

In the HPA SMART model sediment is assumed to be evenly distributed, in a 1 cm layer, on all the internal surface of the pipe wall. The dose from external exposure to sewer pipe workers, E_{ext} (Sv y^{-1}) is calculated using the equation:

$$E_{ext} = C_{sed}\,DR\,T_{exp} = Q\frac{TF}{\rho_{sed}}DR\,T_{exp}$$

Where C_{sed} is the activity concentration in sediment (Bq g^{-1}), calculated from the discharge rate, Q, (Bq d^{-1}), a radionuclide dependent transfer coefficient of activity to the pipe wall, TF* (Bq m^{-2} per Bq d^{-1}) and the mass per unit area of the 1 cm sediment layer, ρ_{sed} (g m^{-2}). DR is the gamma dose rate per unit activity, (Sv h^{-1} per Bq g^{-1}) and T_{exp} is the exposure time (h y^{-1})

The FSA model treats the pipe as an infinite line source and calculates the activity per unit length of pipe or line strength, C_L (Bq m^{-1}). The line strength is the product of the cross-section area (m^2) of the sediment on the pipe and the activity concentration (Bq m^{-3}) in the sediment deposited on the pipe. A fuller explanation of how line strength is calculated is given in APPENDIX A. The line strength is used to estimate effective dose from external exposure, E_{ext} (Sv y^{-1}), to sewer pipe maintenance workers using the equation:

* SMART actually stores and uses a transfer coefficient given in terms of Bq g^{-1} per Bq d^{-1}, which is equal to TF/ρ_{sed} in the equation above.

$$E_{ext} = \frac{C_L}{4} DF\, T_{exp}\, 3.6 \times 10^3$$

Where DF is the effective dose per unit fluence (Sv m Bq^{-1} s^{-1}), T_{exp} is the exposure time (h y^{-1}) and 3.6 10^3 is the conversion factor from hours to seconds (s h^{-1}).

This equation has been taken from the report of a CEFAS study (Brownless and Round, 2000), which also provides values of effective doses per unit fluence. However, the values given in the report are in units of Sv m^{-1} per Bq cm^{-2}, rather than Sv m Bq^{-1} s^{-1}, as stated in the equation. This means that the resulting dose is in Sv s y^{-1}, rather than Sv y^{-1}, once a conversion from cm^2 to m^2 is included. This quantity can be interpreted as the effective dose delivered over a period of time per unit line strength. The CEFAS report does not specify over what period the dose is delivered; the doses from external exposure given in Table 3 seem to suggest that the integration time is an hour. Differences between the doses from external exposure to sewer pipe maintenance workers calculated by the FSA model and those calculated by the other models are likely to be due to the discrepancy between the equation for doses from external exposure and the values of the effective doses per unit fluence given in the CEFAS report (Brownless and Round, 2000). This conclusion is supported by the fact that the differences between doses from external exposure to STW workers calculated by the FSA model and other models are smaller than those to sewer pipe maintenance workers.

It is also unclear why the line strength is divided by four in the equation for doses from external exposure given in the CEFAS study report (Brownless and Round, 2000). It may be linked to the flux, J, from a point source emitting C_L being given as:

$$J = \frac{C_L}{4} \pi R^2$$

However no clear explanation is provided in the CEFAS report. In addition a mistake has been made when the equation to calculate the line strength was implemented in the FSA model (see APPENDIX A for more detail), which results in the FSA model underestimating by a factor of 4 the correct value of the line strength. As a result of these errors and inconsistencies it is very difficult to compare external doses to sewage pipe workers between the models.

With slight variations in units between models, all models calculate doses from inadvertent ingestion of sewage material, E_{ing} (Sv y^{-1}), as:

$$E_{ing} = C_{mat}\, R_{ing}\, DC_{ing}\, T_{exp}$$

where C_{mat} is the activity concentration in sewage effluent (HPA SMART) or sediment (FSA model) (Bq kg^{-1}), R_{ing} is the intake rate by inadvertent ingestion (kg h^{-1}), DC_{ing} is the dose coefficient for ingestion (Sv Bq^{-1}) and T_{exp} is the exposure time (h y^{-1}), while doses from inhalation of sewage material E_{inh} (Sv y^{-1}), are calculated as:

$$E_{inh} = C_{eff}\, DL\, R_{inh}\, DC_{inh}\, T_{exp}$$

where C_{eff} is the activity concentration in sewage effluent (in both models) (Bq kg^{-1}), DL is the dust loading per unit activity concentration in effluent (Bq m^{-3} per Bq kg^{-1}), R_{inh} is the intake rate by inhalation (m^3 h^{-1}), DC_{inh} is the dose coefficient for inhalation (Sv Bq^{-1}) and T_{exp} is the exposure time (h y^{-1}).

The FSA model does not consider inhalation in the sewer pipe except for tritiated water. The HPA SMART model indicates that, for most radionuclides, the dose to sewer pipe maintenance workers from inadvertent ingestion of sewage effluent is greater than the dose from inhalation by a factor of between 10 and 100. Therefore, the omission of inhalation from the calculation of doses is unlikely to make any significant difference to the total dose.

A difference that does affect estimated doses to sewer pipe maintenance workers is that the FSA model uses soil distribution coefficients (K_d) to separate the activity between the effluent and sediment within the sewer pipe, while the HPA SMART model assumes that it is unlikely that radionuclides would be partitioned at this stage between effluent and sediment, because of the lower suspended sediment load and the fast flow rate of the sewage. Section 5.2 provides more discussion about the use of soil distribution coefficient K_d to partition activity between effluent and sediment.

4.1.2 Influence of habit data on doses to sewer pipe workers

Tables 6 and 7 in Appendix B give the habit data used in the models. These include exposure times, intake rates for inhalation and inadvertent ingestion, and ingestion rates of different types of foodstuffs. For sewer pipe maintenance workers, the HPA SMART model uses exposure times that are eight times higher and intake rates for inadvertent ingestion that are 20% higher than those used by the FSA model. Therefore the HPA SMART model would be expected to calculate higher doses than the FSA model. However, most doses estimated by the FSA model are significantly greater than those estimated by the HPA SMART model. This indicates that the effects on doses due to exposure times and ingestion rates are less significant than those due to other differences in the methodologies.

4.2 Doses to workers at sewage treatment works (STW)

Doses to workers at sewage treatment facilities are shown in Table 4. Doses for each exposure pathway as well as total doses are shown in Figures 4, 5 and 6. The ratios of the highest to lowest doses were lower than those calculated in the 2009 comparison exercise for all of the radionuclides and pathways apart from total dose for ^{32}P. The higher ratio for this radionuclide was due to the inclusion of the HPA W63 model, which was not considered in the 2009 exercise. Estimates of total dose were relatively close for ^{99m}Tc and ^{131}I, though larger differences were found for ^{201}Tl and ^{111}In.

As for sewer pipe maintenance workers, the doses from external exposure contributed most significantly to the total dose estimates, apart for ^{3}H. Doses to STW workers from external exposure for ^{99m}Tc and ^{131}I were relatively close for all models; the ratio between highest and lowest doses is only around 2 for ^{99m}Tc, while it is 4.5 for ^{131}I. For

^{111}In the highest estimate of external dose was 9 times higher than the lowest estimate. A larger difference was found in doses for this pathway for ^{201}Tl, where the ratio of highest to lowest estimates was 150. Only the EA model calculated a dose from external exposure for ^{32}P. The inclusion of doses from external exposure for ^{32}P in the EA model led to a larger difference between maximum and minimum total dose estimated for this radionuclide of 2 orders of magnitude. The use of external dose factors, including the use of an external dose factor for ^{32}P is discussed in Section 5. The FSA model calculated the highest doses to STW workers from inhalation and inadvertent ingestion of sewage material for most radionuclides. However, for ^{3}H and ^{32}P, the FSA model gave the lowest estimated doses, with the HPA W63 model giving the highest estimates. While for most radionuclides the ratio between the highest and lowest dose estimates is quite small, greater differences between models for this pathway were for ^{3}H, ^{111}In and ^{201}Tl. For ^{111}In and ^{201}Tl this follows patterns seen with other pathways. In the case of ^{3}H, the lower estimate of ingestion and inhalation dose by the FSA model matches the fact that the FSA model estimates lower inhalation, ingestion and food doses to the farming family (see Section 4.3). These results are linked to the use of partitioning factors, as discussed in Section 5.2.

Table 4: Doses per unit discharge (Sv y^{-1} per GBq y^{-1}) to STW workers (Feb 2010)*

Doses from external exposure (Sv y^{-1} per GBq y^{-1})					
Radionuclide	EA model	FSA model	HPA SMART	HPA W63	Ratio high to low
^{3}H	0.0	0.0	0.0	0.0	N/A
^{32}P	4.0 10^{-8}	0.0	0.0	0.0	N/A
^{99m}Tc	3.3 10^{-8}	**2.2 10^{-8}**	2.3 10^{-8}	**4.7 10^{-8}**	2.1
^{111}In	1.3 10^{-6}	**8.3 10^{-6}**	1.9 10^{-6}	**9.0 10^{-7}**	9.2
^{131}I	9.0 10^{-7}	**6.5 10^{-7}**	1.0 10^{-6}	**2.9 10^{-6}**	4.5
^{201}Tl	**1.1 10^{-7}**	**1.7 10^{-5}**	4.3 10^{-7}	2.0 10^{-7}	150
Combined doses from inhalation and ingestion of sewage material (Sv y^{-1} per GBq y^{-1})					
Radionuclide	EA model	FSA model	HPA SMART	HPA W63	Ratio high to low
^{3}H	2.3 10^{-12}	**6.9 10^{-13}**	5.9 10^{-12}	**9.5 10^{-12}**	14
^{32}P	7.8 10^{-10}	**3.4 10^{-10}**	8.3 10^{-10}	**1.8 10^{-9}**	5.3
^{99m}Tc	2.4 10^{-13}	**4.0 10^{-13}**	**1.1 10^{-13}**	1.8 10^{-13}	3.6
^{111}In	3.2 10^{-11}	**7.2 10^{-10}**	7.0 10^{-11}	**1.4 10^{-11}**	51
^{131}I	**1.6 10^{-9}**	**4.1 10^{-9}**	3.4 10^{-9}	3.5 10^{-9}	2.6
^{201}Tl	7.2 10^{-12}	**2.0 10^{-9}**	2.3 10^{-11}	**4.3 10^{-12}**	470
Total doses (Sv y^{-1} per GBq y^{-1})					
Radionuclide	EA model	FSA model	HPA SMART	HPA W63	Ratio high to low
^{3}H	2.3 10^{-12}	**6.9 10^{-13}**	5.9 10^{-12}	**9.5 10^{-12}**	14
^{32}P	**4.1 10^{-8}**	**3.4 10^{-10}**	8.3 10^{-10}	1.8 10^{-9}	120
^{99m}Tc	3.3 10^{-8}	**2.2 10^{-8}**	2.3 10^{-8}	**4.7 10^{-8}**	2.1
^{111}In	1.3 10^{-6}	**8.3 10^{-6}**	1.9 10^{-6}	**9.0 10^{-7}**	9.2
^{131}I	9.0 10^{-7}	**6.5 10^{-7}**	1.0 10^{-6}	**2.9 10^{-6}**	4.5
^{201}Tl	**1.1 10^{-7}**	**1.7 10^{-5}**	4.3 10^{-7}	2.0 10^{-7}	150

Note:

*: Maximum doses are given in **bold**; minimum doses are given in **bold red**

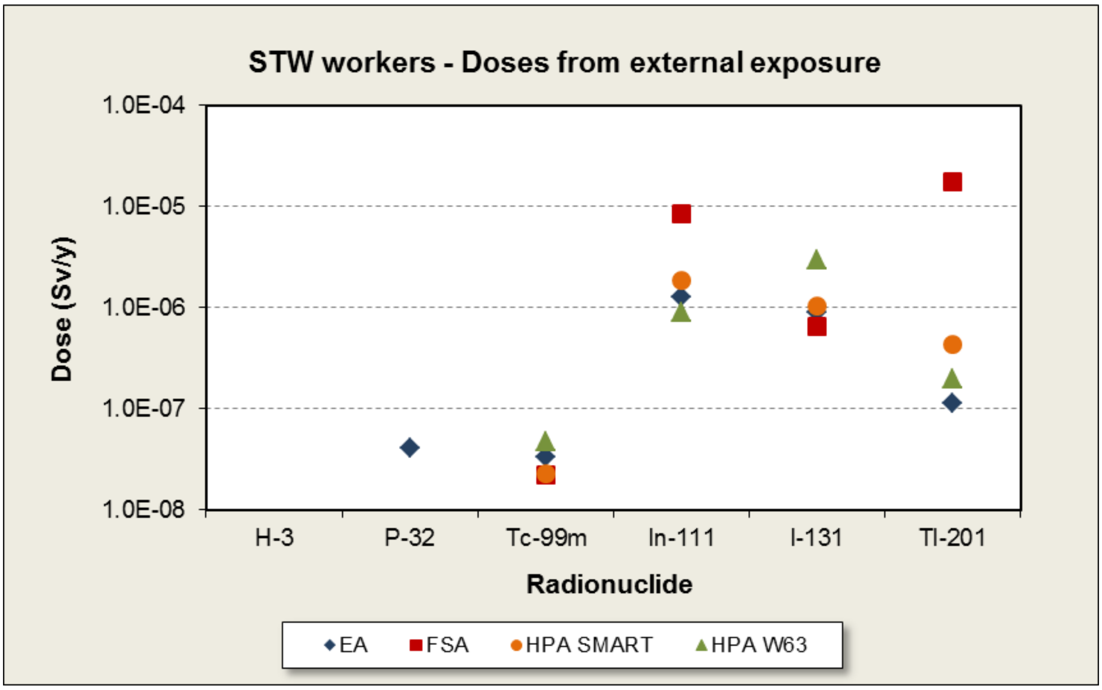

Figure 4. Doses to STW workers from external exposure

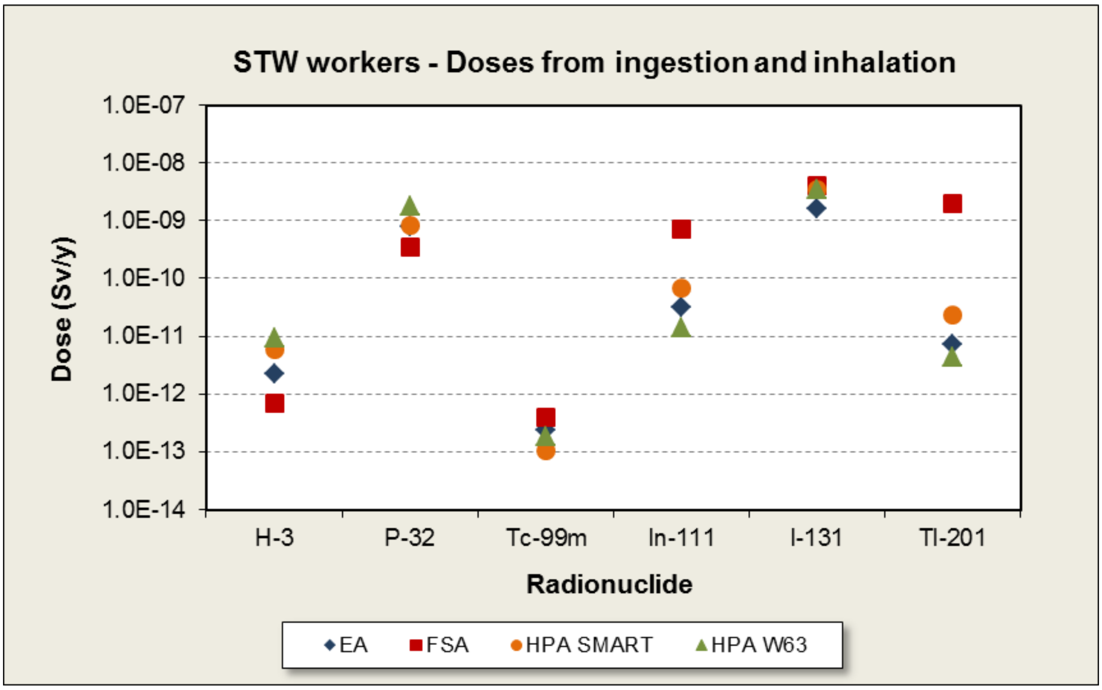

Figure 5. Doses to STW workers from inhalation and inadvertent ingestion (combined) of sewage material

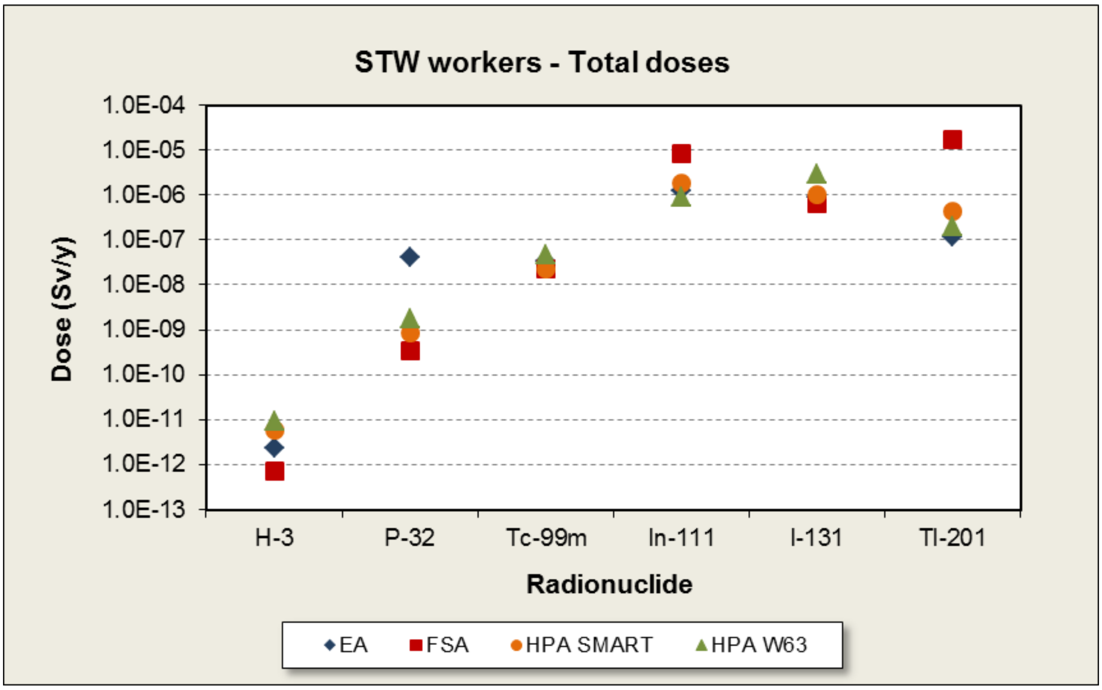

Figure 6. Total doses to STW workers

4.2.1 Analysis of the methodologies used in the calculation of doses to STW workers

The EA and both the HPA SMART and HPA W63 models calculate doses from external exposure to effluent and sludge separately, with the total external dose being the sum of the doses calculated for the two media. These three models calculate external doses to STW workers, E_{ext} (Sv y^{-1}), using the equation:

$$E_{ext} = C_{mat}\, DR_{\gamma}\, T_{exp}$$

Where C_{mat} is the activity concentration in either effluent or sewage sludge (Bq m^{-3}), DR_{γ} is the external gamma dose rate at 1 metre (Sv h^{-1} per Bq m^{-3}) and T_{exp} is the exposure time (h y^{-1}). The external gamma dose rate used in the EA model includes shielding from tanks, while the HPA SMART and W63 models do not.

The FSA model, instead, only considers external exposure from sludge tanks, and uses the equation taken from the report of a CEFAS study (Brownless and Round, 2000):

$$E_{ext} = C_{sludge}\, DR_{f}\, \bar{E}_{\gamma}\, T_{exp}\, f_{geom}\, f_{D}$$

Where C_{sludge} is the activity concentration in the sewage sludge (Bq t^{-1}); DR_f is a dose rate factor per unit activity concentration (2.88 10^{-13} Gy h^{-1} per Mev Bq^{-1} per Bq t^{-1}); $\bar{E}_{\gamma}$ is the mean gamma energy per disintegration (MeV Bq^{-1}); T_{exp} is the exposure time (h y^{-1}); f_{geom} is a geometric factor to allow for the sludge being stored in tanks (0.5); and f_D is a factor used to convert absorbed dose in air to effective dose (0.85 Sv Gy^{-1}). This equation can also be written as:

$$E_{ext} = C_{sludge}\, DR_{\gamma}\, T_{exp}\, f_{geom}$$

Where:

$$DR_\gamma = DR_f \, \bar{E}_\gamma \, f_D$$

The effects of the different approaches used to calculate external dose are considered in Section B1. The use of a geometric factor of 0.5 in the FSA model would be expected to result in the HPA SMART model estimating external doses to STW workers that are about twice as high as those estimated by the FSA model. However, the FSA model assumes that exposure is due to sludge in sludge tanks, while the other models assumes that exposures are due to both sludge and effluent. As activity concentrations in sludge tend to be higher than those in the effluent, external doses estimated by the FSA are close to or greater than those estimated by the other models.

No significant differences were found between the models used to calculate doses from inhalation and ingestion of sewage material. For these pathways, all models use the methodologies outlined in Section 4.1.1.

4.2.2 Influence of habit data on estimates of doses to STW workers

Habit data used to calculate doses to STW workers, such as occupancy times, breathing and inadvertent ingestion rates are given in Table B6 of Appendix B. The EA and HPA W63 models use the same overall exposure times (2000 h y^{-1}), while the HPA SMART model uses a slightly lower exposure time (1800 h y^{-1}), and the FSA model uses a still lower exposure time (1000 h y^{-1}). The FSA model uses a slightly higher breathing rate, and a slightly lower inadvertent ingestion rate than the other models, but these would not be expected to lead to significant differences in estimated doses.

The FSA model calculates the lowest combined dose from inadvertent ingestion and inhalation of sewage material for ^{3}H and ^{32}P and the lowest external doses for ^{99m}Tc and ^{131}I. These results are consistent with the lower exposure time adopted in that model. This is not the case for the other radionuclides, however, and in particular the estimated doses for ^{111}In and ^{201}Tl, from both inadvertent ingestion and inhalation of sewage material and external exposure, are noticeably higher than those calculated by the other models. These results are probably due to higher activity concentrations in sewage sludge estimated by the FSA model because of the different approach to partitioning activity between effluent and sludge, as described in Section 5.2.

Despite similar exposure times for STW workers assumed by the two models, the HPA SMART model estimates higher external doses to STW workers than the EA model for all radionuclides apart from ^{99m}Tc. This is due to the fact that the models make different assumptions on the way this time is spent: the EA model assumes that a STW worker spends more time close to the effluent, while the HPA SMART model assumes that STW workers spend more time close to the sludge. With the exception of short-lived radionuclides such as ^{99m}Tc, which decay significantly by the time the sludge reaches the treatment stages, activity concentrations in the sludge are typically between 10 and 100 times greater than the activity concentrations in the effluent. This explains why the HPA SMART model calculates higher doses for the other radionuclides.

4.2.2.1 Influence of other parameters

Activity concentrations in sludge are based on the ratio of the solid contents of sludge to the solid contents of the incoming raw sewage. Although the EA and HPA SMART models use the same value for the solids content of raw sewage and similar element dependent partitioning coefficients, they estimate different activity concentrations in sludge because they use different values for solids contents of sludge, which affects external dose to STW workers.

4.3 Doses to the farming family

All models estimated doses to the farming family, though not for all radionuclides. This is partly because of a lack of data but mainly because the application of sewage sludge to land is not modelled for short lived radionuclides, with a half-life of up to 10 days, since these radionuclides decay considerably in the period between the treatment of sewage at the STW and its application to land, which is about a month. This approach is consistent with the one used for the calculation of Generalised Derived Limits (GDLs) (NRPB, 2005). The HPA W63 model only estimated a total dose, rather than doses by exposure pathway. The estimated doses are shown in Table 5. Doses for each exposure pathway as well as total doses are shown in Figures 7, 8, 9 and 10.

Of the radionuclides considered in this intercomparison exercise, total doses to the farming family could only be compared for ^{3}H, ^{32}P and ^{131}I. The estimates for ^{3}H and ^{131}I were fairly close, with the EA, HPA SMART and HPA W63 models showing very good agreement, while the estimates of the FSA model were within an order of magnitude. For ^{32}P, the EA, HPA SMART and HPA W63 models showed good agreement, but the total dose calculated by the FSA model was around two orders of magnitude lower. For the radionuclides included in this exercise, doses to the farming family calculated by different models were much closer than those calculated in the 2009 exercise.

Doses from ingestion of food grown on farmland treated with sewage sludge contribute the most to the doses to a farming family. The doses estimated by the models are fairly close to each other, with ratios of highest to lowest doses of less than one order of magnitude for ^{3}H and ^{131}I and two orders of magnitude for ^{32}P. Apart from ^{131}I, the FSA model calculated the lowest doses, but no model consistently gave the highest doses. Iodine-131 is the only radionuclide for which external doses can be compared. Dose from external exposure to the farming family from ^{131}I calculated by the FSA model was similar to those estimated by the HPA SMART models, while the dose calculated by the EA model was almost an order of magnitude higher. The approach used by the EA model to calculate external dose for ^{32}P is discussed in Section 5. Doses to the farming family from inadvertent ingestion and inhalation of soil could be compared between the EA, FSA and HPA SMART models only for ^{3}H, ^{32}P and ^{131}I. The HPA SMART model estimated the highest doses and the FSA model estimated the lowest doses for all three radionuclides. The ratios of the highest to the lowest doses were about three orders of magnitude for ^{3}H and about four orders of magnitude for ^{32}P and ^{131}I.

Table 5: Doses per unit discharge (Sv y^{-1} per GBq y^{-1}) to a farming family[*] (Feb 2010)

Doses from external exposure (Sv y^{-1} per GBq y^{-1})

Radionuclide	EA model	FSA model	HPA SMART	HPA W63[#‡]	Ratio high to low
^{3}H	–[†]	–	–	–	N/A
^{32}P	$6.0\ 10^{-9}$	–	–	–	N/A
^{99m}Tc	–	–	–	–	N/A
^{111}In	–	$2.0\ 10^{-8}$	–	–	N/A
^{131}I	**$1.1\ 10^{-8}$**	**$1.6\ 10^{-9}$**	$2.4\ 10^{-9}$	–	$7.0\ 10^{0}$
^{201}Tl	–	$4.2\ 10^{-8}$	–	–	N/A

Combined doses from inhalation and ingestion of soil (Sv y^{-1} per GBq y^{-1})[‡]

Radionuclide	EA model	FSA model	HPA SMART	HPA W63[‡]	Ratio high to low
^{3}H	$3.9\ 10^{-16}$	**$7.7\ 10^{-17}$**	**$1.3\ 10^{-13}$**	–	$1.7\ 10^{3}$
^{32}P	$9.0\ 10^{-12}$	**$5.8\ 10^{-15}$**	**$2.3\ 10^{-10}$**	–	$4.0\ 10^{4}$
^{99m}Tc	–	–	–	–	N/A
^{111}In	–	$6.4\ 10^{-15}$	–	–	N/A
^{131}I	$1.5\ 10^{-12}$	**$1.7\ 10^{-14}$**	**$1.0\ 10^{-9}$**	–	$5.9\ 10^{4}$
^{201}Tl	–	$1.1\ 10^{-14}$	–	–	N/A

Doses from ingestion of food (Sv y^{-1} per GBq y^{-1})

Radionuclide	EA model	FSA model[§]	HPA SMART	HPA W63[‡]	Ratio high to low
^{3}H	**$3.1\ 10^{-9}$**	**$6.3\ 10^{-10}$**	$2.6\ 10^{-9}$	–	4.8
^{32}P	$6.0\ 10^{-6}$	**$5.3\ 10^{-8}$**	**$1.4\ 10^{-5}$**	–	$2.7\ 10^{2}$
^{99m}Tc	–	–	–	–	N/A
^{111}In	–	$1.8\ 10^{-8}$	–	–	N/A
^{131}I	**$3.5\ 10^{-8}$**	**$2.7\ 10^{-7}$**	$6.5\ 10^{-8}$	–	$7.6\ 10^{0}$
^{201}Tl	–	$2.6\ 10^{-9}$	–	–	N/A

Total doses (Sv y^{-1} per GBq y^{-1})[¶]

Radionuclide	EA model	FSA model	HPA SMART	HPA W63[‡]	Ratio high to low
^{3}H	$3.1\ 10^{-9}$ (Infant)	**$6.3\ 10^{-10}$ (Adult)**	$2.6\ 10^{-9}$ (Infant)	**$3.1\ 10^{-9}$ (Adult)**	$4.9\ 10^{0}$
^{32}P	$6.0\ 10^{-6}$ (Offspring)	**$5.3\ 10^{-8}$ (Adult)**	**$1.4\ 10^{-5}$ (Offspring)**	$1.2\ 10^{-5}$ (Adult)	$2.7\ 10^{2}$
^{99m}Tc	–	–	–	–	N/A
^{111}In	–	$3.8\ 10^{-8}$ (Adult)	–	–	N/A
^{131}I	**$4.6\ 10^{-8}$ (Infant)**	**$2.7\ 10^{-7}$ (Adult)**	$6.8\ 10^{-8}$ (Infant)	–	$5.9\ 10^{0}$
^{201}Tl	–	$4.5\ 10^{-8}$ (Adult)	–	–	N/A

Notes:

[*]: Maximum doses are given in **bold**; minimum doses are given in **bold red**

[#]: The HPA W63 model only estimates a total dose to the farming family

[†]: – = Not included in the model

[‡]: The FSA model does not include inadvertent ingestion of soil

[§]: The FSA provided separate doses from ingestion of terrestrial food following irrigation, spreading of sludge on land, and animal consumption of fodder. For the purpose of the intercomparison exercise these doses were summed together

[¶]: The age group that receives the highest total dose is given in brackets.

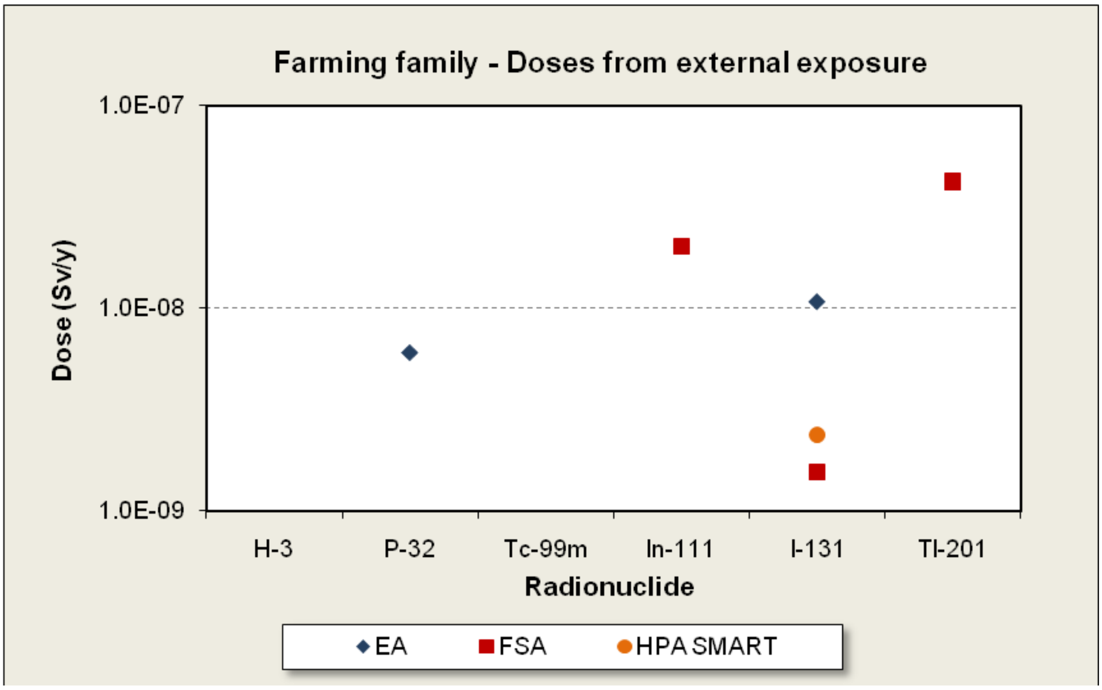

Figure 7. Doses to the farming family from external exposure

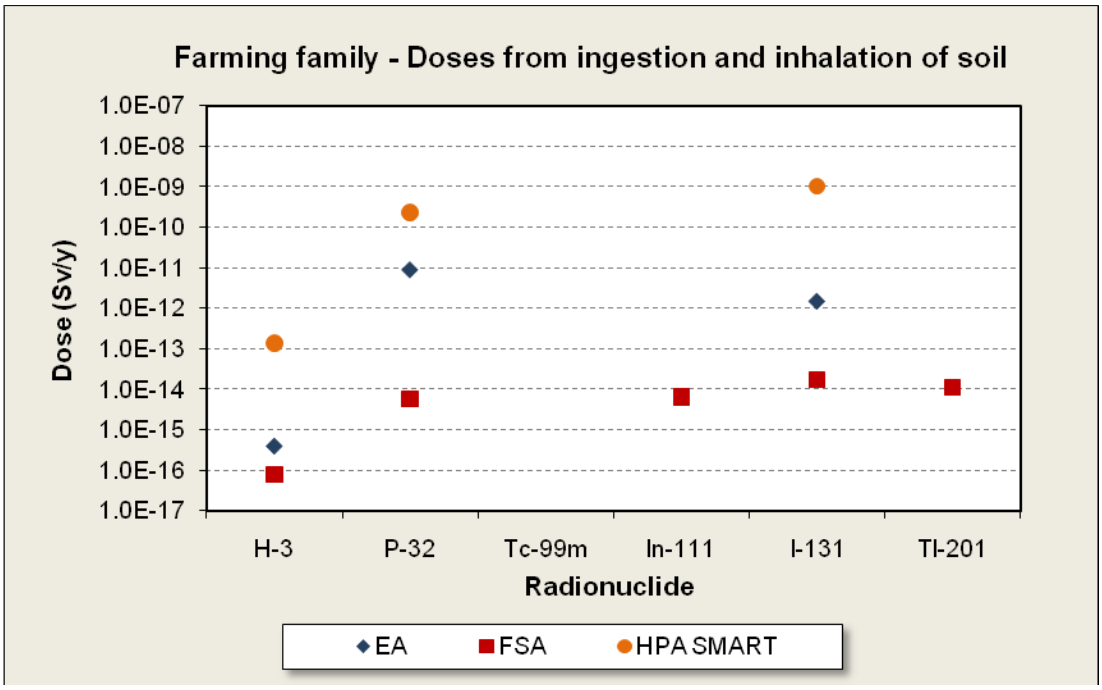

Figure 8. Doses to the farming family from inhalation and ingestion of soil

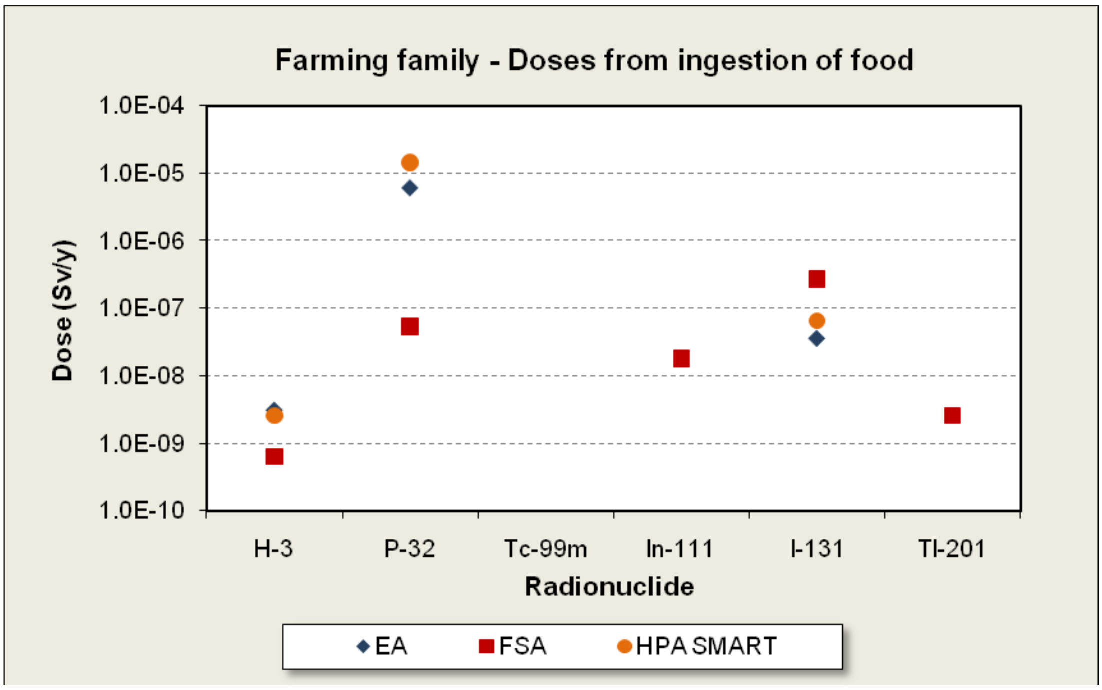

Figure 9. Doses to farming family from ingestion of foods grown on farmland treated with sewage sludge

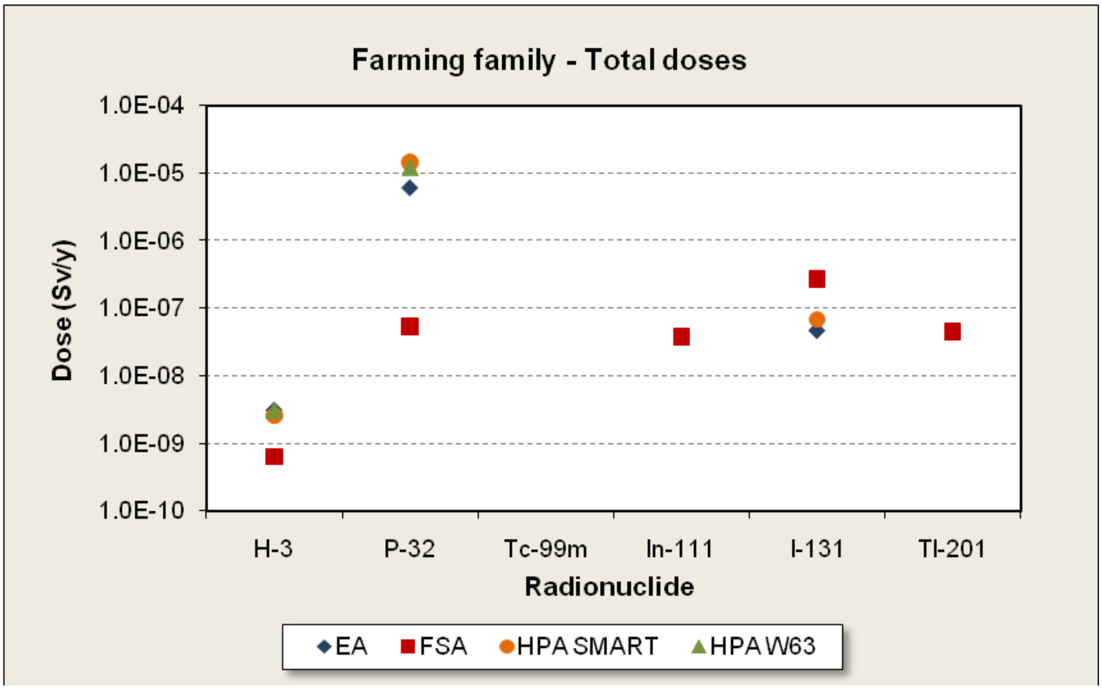

Figure 10. Total doses to farming family

4.3.1 Analysis of the methodologies used to estimate doses to the farming family

Doses to the farming family from external exposure to radioactivity in the soil are estimated using the methodology outlined in Section 4.2.1, with activity concentrations in soil rather than in effluent or sewage sludge. In addition the equation used by the FSA model does not include the geometric factor of 0.5 as there are no sewages tanks involved.

Similarly, the methodologies to calculate doses from inhalation and inadvertent ingestion of soil are those outlined in Section 4.1.1 for the calculation of doses from inhalation of sewage material, although activity concentrations in soil rather than in effluent or sludge are used. The FSA model does not consider ingestion of soil. Inadvertent ingestion of soil tends to lead higher doses than inhalation of soil, and this is therefore one of the reasons why the combined doses from inhalation and ingestion of soil estimated by the FSA model are lower than those calculated by other models.

The methodologies used to estimate doses from ingestion of food are similar to those given for inadvertent ingestion of sewage material in Section 4.1.1. The only differences are that activity concentrations in different types of foods rather than in effluent or sludge are used and that ingestion rates do not take account of the exposure time and are given in kg y^{-1} rather than kg h^{-1}.

Table 6 shows the activity concentrations in soil estimated by the EA, FSA and HPA SLUDLAND models. No activity concentrations in soil are given in the W63 model. The HPA SMART model does not calculate activity concentrations in soil from the application of sewage sludge to farmland; these are calculated by a different model called SLUDLAND, which also calculates the doses to a farming family. Differences in the activity concentrations in soil lead to differences in estimated doses to the farming family and help explain why the FSA model generally gives the lowest doses to the farming family. However, the higher concentrations in sewage sludge and soil implied by the EA model are not reflected in the dose estimates seen in Table 5. The estimated activity concentrations in soil depend on the activity concentration in sewage sludge applied to farmland, given in Table 7, the application rates of sewage sludge to land, given in Table B7, and the method of application (for example, whether the sewage is applied to the surface or ploughed in), which also varies between models. The HPA model SLUDLAND calculates activity concentrations in the top 1 cm layer of soil over a 50 year period, taking account of build-up in the soil and radioactive decay. However, for this exercise, the activity concentration in soil is unlikely to increase much from the value in the first year, due to the radioactive half-life and other properties of the radionuclides considered. The FSA model calculates soil concentration by dividing sewage application to land (Bq m^{-2} y^{-1}) by the density of the soil (kg m^{-3}) and the ploughing depth (m), and does not consider radioactive decay and build-up of the activity concentration over time.

Table 6: Activity concentrations in soil calculated by the models* (Bq kg^{-1})

Radionuclide	EA model†	FSA model	HPA SMART model#
^{3}H	1.5	$1.59\ 10^{-3}$	$6.37\ 10^{-2}$
^{32}P	$7.1\ 10^{1}$	$1.08\ 10^{-2}$	$1.09\ 10^{0}$
^{131}I	$7.1\ 10^{0}$	$1.66\ 10^{-2}$	$1.27\ 10^{-1}$

Notes:

*: No concentrations are given for the W63 model, as it does not contain enough detail to extract this information.

#: Activity concentrations in soli from the application of sewage sludge to farmland are calculated by the model SLUDLAND (Mobbs et al, 2005) rather than the SMART model. For convenience the model in this report is still referred to as HPA SMART.

†: Activity concentrations in soil for the EA model were calculated by HPA using the sludge concentrations in Table 7 multiplied by the application rate and the unit concentration in soil from the EA Dose Per Unit Release spreadsheet (Allot et al, 2006).

Table 7: Activity concentrations in sewage sludge leaving the STW* (Bq m^{-3})

Radionuclide	EA model#	FSA model	HPA SMART model
^{3}H	$6.8\ 10^{5}$	$1.32\ 10^{3}$	$2.95\ 10^{4}$
^{32}P	$2.0\ 10^{6}$ ($9.7\ 10^{5}$)	$8.95\ 10^{3}$	$3.05\ 10^{4}$
^{131}I	$3.5\ 10^{5}$ ($8.6\ 10^{4}$)	$1.38\ 10^{4}$	$6.31\ 10^{3}$

Notes:

*: Although the EA and HPA SMART models use similar radionuclide dependant partitioning coefficients, they use different solids contents for sludge

#: The EA model includes a 21 day delay before the pasture is grazed. The activity concentrations given in brackets are the activity concentrations after the 21 day decay is applied.

Other differences between the methodologies used to calculate doses to the farming family are:

- The FSA model includes calculation of doses from ingestion of foodstuffs grown on land irrigated with river water containing effluent discharged from a STW. This pathway is not considered by the other models and therefore, for the purposes of this intercomparison exercise, the doses from irrigation were not included in the doses from ingestion of food presented in Table 5. If doses from this pathway had been included, only the dose from ingestion of food for ^{3}H would have risen significantly, with no significant difference for the other radionuclides considered.

- The FSA model considers ingestion of some foodstuffs (eggs, game, pork, and poultry) which are not considered in the other models. The inclusion of these foods, however, does not have a significant effect, as consumption of beef and milk are the main contributing pathways to the doses from ingestion of food.

- When estimating doses from ingestion of animal products (eg, cow meat, sheep meat, offal, milk, eggs) the HPA and EA models assume that livestock consume only fodder grown on land treated with sludge. The FSA model also assumes that livestock ingest drinking water obtained from a river that receives effluent from a STW For each radionuclide, the FSA model uses the higher activity intakes of the livestock from ingestion of fodder grown on land treated with sludge, or from ingestion of drinking water. For most of the radionuclides considered in this exercise, the animals received higher intakes from ingestion of fodder. However, for ^{131}I the FSA model calculated the food dose based on the animals' ingestion of drinking water rather than ingestion of fodder.

4.3.2 Influence of habit data on estimates of doses to the farming family

Table B7 in Appendix B contains the habit data for the farming family used by the models. There is a large difference in exposure times between the FSA model, which uses 1000 hours, based on 125 working days per year, and the HPA and EA models, which assume exposure over a full calendar year resulting in higher exposure times than that of the FSA model. This difference contributes to the lower dose estimates produced by the FSA model for the external and inadvertent ingestion and inhalation of soil pathways.

Although the EA and HPA SLUDLAND models use the same exposure times, the EA model gives lower doses than the HPA model because it assumes that people receive doses from inadvertent ingestion and inhalation of soil only when they are outside. The HPA SLUDLAND model accounts for time spent indoors when calculating external exposure, but for inadvertent inhalation and ingestion it makes no allowance for time spent indoors and assumes that doses for these exposure pathways are received throughout the year.

There are also differences in the rates of inhalation and inadvertent ingestion of soil between models. Table B7 shows that the FSA model uses the highest inhalation rate for the farming family, but does not include ingestion of soil. Doses from inadvertent ingestion are generally higher than those from inhalation of soil, up to two orders of magnitude, even though there are cases, depending on the model, where doses from inhalation of soil can be slightly higher. Therefore doses estimated by the FSA model for combined inhalation and ingestion of soil can be lower than those from other models.

The EA model considers ingestion of similar foods to the HPA model, except that it does not include ingestion of dairy products other than milk. This is one of the reasons why the doses from ingestion of food estimated by the HPA model are higher than those calculated by the EA model for ^{32}P and ^{131}I, as these radionuclides concentrate in dairy products and doses from consumption of dairy products contribute almost as much as doses from consumption of milk to the total dose in the HPA SLUDLAND model. There are also differences in the assumptions made by the different models on how consumption rates are chosen. The HPA SLUDLAND model uses higher than average consumption rates for only beef and milk, and average ingestion rates for all other foodstuffs. Both the FSA model and the HPA W63 model use higher than average values for all foods, while consumption rates used in the EA model are all average values. Consumption rates for adults used in the 2010 intercomparison exercise are given in Table B7 in Appendix B. The increase in doses calculated by the FSA model caused by the use of higher than average consumption rates for all food groups is much smaller than the reduction in doses due to the much lower application rate of sludge to farmland adopted in that model (20 times lower than the rate used by the EA and HPA). In addition the FSA model only calculates doses to adults[*], while both the EA and HPA SLUDLAND models consider doses to all age groups, including the fetus. The ingestion dose coefficients for the fetus are generally higher than those for adults, though the difference varies with radionuclides[†]. As a result, the FSA model estimates lower doses from ingestion of food than the other models for all radionuclides apart from ^{131}I. This is because the pathway which contributes most significantly to the dose for this radionuclide is ingestion of milk and the FSA model uses a higher value than the other models for the transfer factor from soil to milk. This is possibly due to differences in assumed fodder intake rates and fodder to cow transfer rates. The FSA recognises that its model is likely to overestimate doses for ingestion of ^{131}I in milk, but given that its primary goal is to protect the public from excessive exposure resulting from ingestion of food, considers that it provides the FSA with a robust upper-bound estimate of the dose.

4.3.2.1 Other parameters

As already discussed in Section 4.2.2.1, the use of different values for solids contents of sludge means that the activity concentrations in sludge calculated by the EA and HPA SMART models differ. However, the difference in solid contents of sludge at the last stage of the treatment processes in a sewage treatment plant is such that the EA model would be expected to calculate slightly higher dose estimates to the farming family than the HPA model, yet doses from inhalation and ingestion of soil calculated by the HPA model are between one and three orders of magnitude higher. Therefore other more significant factors play a role in reducing the activity concentration in soil, such as the inclusion of a 21 day delay time in the EA model between the application of sludge to pasture land and animals grazing that land to reflect the requirement of the safe sludge matrix (ADAS, 2001).

5 ANALYSIS OF RADIONUCLIDE DEPENDENT DATA USED IN THE MODELS

The parameters considered in sections 4.1.2, 4.2.2 and 4.3.2 are radionuclide independent; they remain the same whichever radionuclide is being modelled. Parameters that are dependent on the radionuclide are discussed in this section.

[*] The FSA model does not estimate doses to children in the farming family as there is less available data on food consumption rates for children than for adults. The EA and HPA models use consumption rates for children that are approximately half of the rates for adult.

[†] For ^{32}P, the dose coefficient for the fetus is 32 times greater than the dose coefficient for adult; for tritium the ratio is only a factor of two.

5.1 Basic radionuclide data

Dose coefficients for ingestion and inhalation used by each model for the radionuclides considered in the 2010 exercise are given in Table B1 in Appendix B. HPA models use a lower dose coefficient for inhalation for ^{32}P. This is because the HPA models use a dose coefficient for members of the public based on absorption type F, as recommended by ICRP Publication 72 (ICRP 1996) while other models use values for absorption type M. Also the dose coefficients for ^{3}H used in the FSA model are those for organically bound tritium, while other models use values for tritiated water. Differences in dose coefficients are not large enough to have a significant effect on the doses calculated.

As shown in Section 4.2.1, different models use similar equations to calculate doses from external exposure, but with different dose rates for external irradiation. More information on these dose rates is given in Appendix B (Section B4). The external dose factors used in the models are shown in Table B4.

5.2 Partitioning data

In order to consider doses from effluent and sludge separately, the models include partitioning of activity between the two components. The EA, HPA SMART and HPA W63 models use partitioning factors* also called removal coefficients. For example, a partitioning factor of 0.85 (see Table B5) means that 85% of the activity is partitioned into the effluent and 15% into the sludge. In the HPA SMART model the partitioning factor is used in a pair of equations, as given below:

$$C_{eff} = C^0_{eff}\, f_{part}\, f_{decay}$$

$$C_{sludge} = C^0_{eff}\left(1 - f_{part}\right) f_{decay} \frac{F_{tot}}{M_{sludge}}$$

Where C_{eff} and C_{sludge} are the activity concentration in the effluent and in the sludge after partitioning has occurred (Bq m^{-3}), C^0_{eff} is the activity concentration in the effluent (Bq m^{-3}) before partitioning occurs, f_{part} is the partitioning factor, f_{decay} is the factor for radioactive decay, F_{tot} is the total flow (m^3 d^{-1}) and M_{sludge} is the amount of sludge removed by the process (t d^{-1} or m^3 d^{-1}, assuming a density of 1 kg l^{-1}).

The FSA model uses soil distribution coefficients (K_d) to determine how much activity goes into the sludge. When considering sewage material in the STW, the model uses the following equation to calculate the concentration in sludge, and uses that value to estimate doses from external radiation and inadvertent ingestion of sludge at the STW:

* The EA and HPA W63 models use just one factor per radionuclide. The HPA SMART model uses a factor for each stage of the treatment. However most of the partitioning occurs at the secondary treatment stage in the HPA SMART model and the factor given here is the factor for that treatment stage.

$$C_{sludge} = C_{eff}\left(1 - f_{solid}\right) + C_{eff}\,K_d\,f_{solid}$$

Where C_{sludge} (Bq t^{-1}) is the activity concentration in sludge, C_{eff} (Bq m^{-3}) is the activity concentration in the effluent, f_{solid} is the fraction of solids in sludge (0.05) and K_d is the soil distribution coefficient (l kg^{-1}).

However the following equation is used for calculating the concentration in sediment when estimating the dose from inhalation of sediment at the STW:

$$C_{sed} = C_{eff} K_d\ 0.001$$

Where C_{sed} (Bq kg^{-1}) is the activity concentration in sediments (sludge), and 0.001 $m^3\ l^{-1}$ is the conversion factor from litre to cubic meter.

Table B5, in Appendix B, gives the partitioning factors and K_d values used for the radionuclides considered in the 2010 intercomparison exercise. The partitioning factors used by the EA model are generally similar to those used in both HPA models. Where differences do occur, it is difficult to establish how they affect the dose estimates as other parameters are involved in the calculations. In particular, the EA model and the HPA SMART model divide the time spent by STW workers in different ways; this different approach may balance the differences in activity concentrations caused by differences in partitioning factors (see Section 4.2.2 for further details).

Although it is difficult to make a quantitative comparison of partitioning factors and K_d values, the K_d values used in the FSA model were compared with the partitioning factors of the HPA and EA models for a wide range of radionuclides, including the radionuclides considered in this exercise. In most cases, for the radionuclides with higher K_d values, the partitioning factors assumed for the HPA and EA models also result in a higher proportion of activity going into the sludge, which is consistent with the FSA approach. Activity concentrations in sludge estimated by the HPA SMART, EA and FSA models for the radionuclides considered in the 2010 exercise are given in Table 8. Such activity concentrations cannot be extracted from the HPA W63 model and therefore are not included in this table.

Table 8: Activity concentrations in sludge calculated by the HPA SMART, EA and FSA models

	Activity concentration in sludge (Bq kg^{-1})			
Radionuclide	HPA SMART model*	EA model	FSA model (external and ingestion doses)	FSA model (inhalation dose)
^{3}H	$1.8\ 10^{1}$	$6.8\ 10^{2}$	$1.3\ 10^{0}$	$4.2\ 10^{-2}$
^{32}P	$5.9\ 10^{1}$	$2.0\ 10^{3}$	$9.0\ 10^{0}$	$1.5\ 10^{2}$
^{99m}Tc	$2.0\ 10^{0}$	$6.0\ 10^{0}$	$1.4\ 10^{0}$	$2.1\ 10^{0}$
^{111}In	$8.3\ 10^{1}$	$6.1\ 10^{2}$	$1.7\ 10^{2}$	$3.3\ 10^{3}$
^{131}I	$2.1\ 10^{1}$	$3.5\ 10^{2}$	$1.4\ 10^{1}$	$2.5\ 10^{2}$
^{201}Tl	$8.4\ 10^{1}$	$3.6\ 10^{2}$	$1.5\ 10^{3}$	$3.1\ 10^{4}$

Note:

*: At secondary stage of sewage treatment

The doses calculated by the FSA model to STW workers from external exposures and to both sewer pipe maintenance workers and STW workers from inhalation and ingestion of sewage sludge for ^{201}Tl and ^{111}In are significantly higher than those calculated by other models (see Table 3 and Table 4). This is due to the higher activity concentrations in sludge in the FSA model for these radionuclides, as seen in Table 8. The activity concentrations in sludge calculated by the FSA model to calculate doses for both external exposure and inadvertent ingestion of sludge and inhalation of sludge sediments are higher than those calculated by the other models for ^{201}Tl, while for ^{111}In the activity concentration in sludge used to calculate doses for external exposure and inadvertent ingestion of sludge by the FSA is lower than the concentration used in the FSA model. This explains why there is a larger difference between the doses estimated by the FSA model and those from the EA and HPA models for these radionuclides.

The higher activity concentrations in sludge estimated in the FSA model are related to the high K_d values used for these radionuclides (see Table B5). Of the radionuclides considered in this exercise, ^{201}Tl has the highest K_d value in the FSA model, implying a high transfer to sludge. Both the EA and HPA SMART models give a lower activity concentration in sludge for ^{201}Tl than the FSA model. However, despite the fact that 50% of the ^{201}Tl activity is assumed to be transferred to sludge in the EA model, while the HPA SMART model partitions 80% of activity into the sludge, the EA model gives a higher activity concentration in sludge than the HPA SMART model. This result suggests that the differences in activity concentrations in sludge depend on more than just the partitioning. Table B5 also shows a moderately high value of K_d for ^{111}In. Both the EA and the HPA SMART models assume that a high percentage of activity (90% and 80% respectively) is transferred to the sludge for this radionuclide, which explains why the difference between the doses estimated by the FSA model and those from the EA and HPA models for ^{111}In is smaller than for ^{201}Tl.

The FSA model uses very low K_d values for ^{3}H and ^{99m}Tc and therefore calculates lower activity concentrations in sludge, and hence lower doses than the other models. In particular this explains why the FSA model estimates lower doses from ingestion and inhalation of sewage material to sewer pipe maintenance workers for these radionuclides, as well as tritium doses for ingestion and inhalation of soil and ingestion of food received by the farming family.

Differences in partitioning affect the activity concentration in sewage sludge and lead to differences in doses to the farming family following the application of the sludge to land. This effect is not seen in the results of the intercomparison exercise as not all radionuclides are included in all models when calculating doses to the farming family.

6 CONCLUSIONS AND RECOMMENDATIONS

Although the models used in this exercise are relatively simple, there are still a number of factors to be considered, such as methodologies and habit data, with the result that it is not always easy to determine what causes the differences in dose estimates. The intercomparison exercise has found a number of differences between the models and has examined a number of possible causes for these differences, as listed in Table 9. Some of these differences are not significant, while others were considered significant enough to require further investigation.

The main differences were found in the methodologies used to calculate external doses to maintenance workers within large sewer pipes; in the way partitioning between effluent and sewage sludge is modelled; in the assumptions about exposure times and in the approach adopted to divide overall exposure times between different stages of sewage treatment and/or exposure from sludge and effluent and in the methods used to calculate doses to the farming family, where several factors including the application rate of sludge to land, exposure times and ingestion of soil were all found to be important in the calculation of doses.

It is important to consider the differences found in this comparison exercise in context and look at the intended purposes of the models in order to determine whether the models are fit for their purpose.

The models included in this intercomparison have been primarily developed as scoping models for prospective assessments and therefore include a large degree of caution. The aim of prospective assessments is to determine whether or not discharges of radioactivity will lead to doses that exceed the dose limits to help determine if such a discharge is acceptable. However, assessments of doses from discharges to sewers do not currently consider total discharges from a number of users to a single sewage treatment facility. Therefore the EA and SEPA also apply a large degree of caution by comparing estimated doses from prospective assessments with a 20 μSv y^{-1} dose criterion to decide whether a more refined assessment is required to better reflect the specific exposure situation being assessed. Therefore, as long as models are suitably cautious so as not to underestimate doses, and more refined modelling can be carried out if required, differences in dose estimates between models need not be considered serious, even if some of the those differences are substantial.

The models are known to contain cautious assumptions about habit data and other factors. There is therefore some confidence that the models are unlikely to underestimate activity concentrations and doses. In particular, when combined with the approach taken to reassess if estimated doses exceed a dose criterion, it is considered extremely unlikely that an assessed discharge would incorrectly be judged to be acceptable. This is reinforced by findings by SEPA (Dale, 2010) where comparison of retrospective monitoring with predicted estimates indicated that their model, which is similar to the models considered in this exercise, is overestimating by at least two orders of magnitude, and is therefore fit for the purpose of adequately protecting members of the public.

However, this does not mean that no further effort should be spent in trying to improve the models and some areas have been identified in Table 9 where further work may be required to improve the modelling of discharges to sewers and assessments of doses. To achieve this objective it would be useful to have some validation of the models by comparing model estimates of activity concentrations in environmental materials with real measurements, such as those available from SEPA.

It is also recognised that a problem with models such as those considered in this intercomparison exercise is lack of data. For example there may be uncertainly in nuclide specific data, or poor availability of data concerning worker practices and exposure times. While, as discussed above the models used to assess doses from discharges to sewer are considered fit for purpose, effort should be made to ensure that data used in the models is as complete and specific to the situation being modelled as possible.

Table 9: Findings and recommendations of the 2010 sewer intercomparison exercise

	Finding	Recommendation
1	The EA's decision not to assess doses to maintenance workers working in large sewer pipes, on the grounds that doses to these workers are less than the doses to STW workers, appears to be justified, at least for the radionuclides considered in this comparison exercise. With the exception of the doses calculated by the HPA W63 model for ^{3}H and ^{32}P, total doses to STW workers are higher than the HPA SMART estimates of total doses to sewer pipe maintenance workers for each radionuclide, and most are higher than the FSA model estimates.	
2	There is an error with the method used to calculate external dose to sewer pipe maintenance workers in the FSA model, which is based on the report of a CEFAS study (Brownless and Round, 2000). There is also an error in the calculation of the line strength, C_L, used by the FSA model (see Section 4.1.1 and Appendix A.	A high priority should be given to investigate the errors in the FSA model and in the report of a CEFAS study (Brownless and Round 2000).
3	The FSA model does not consider inhalation in the sewer pipe of any radionuclides apart from tritiated water. However this is unlikely to make any significant difference to the total doses.	
4	The EA model is the only model to use an external dose factor for ^{32}P	
5	There are differences between doses to STW workers estimated by the HPA SMART and W63 models. The HPA SMART model considers sewage treatment as a series of stages, while the HPA W63 model considers the treatment to be a single stage and uses single values for activity concentration and exposure time. As a result doses estimated by the HPA W63 model are about 10 times lower than those estimated by the HPA SMART model.	
6	Differences in the method used to partition activity between effluent and sludge were found to lead some significant differences in dose estimates. The FSA model uses soil distribution coefficients (K_d) to model partitioning of activity to sludge, while the other models use partitioning factors. For some radionuclides (^{111}In and ^{201}Tl) the K_d value used in the FSA model can lead to rather different activity concentrations and hence doses to workers from those estimated by the EA and HPA models. Some differences were found between partitioning factors used by the EA and HPA models, but these differences were not significant. Differences between models in activity concentrations in effluent and sludge also lead to different activity concentrations in soil following the application of sludge to land, and hence differences in estimations of doses to a farming family.	Because this affects all pathways and population groups, and because some noticeable differences have been seen in this exercise, it is recommended that the use of partitioning should be given a high priority if further work is carried out to try to improve sewer modelling that. In particular, factors for ^{111}In and ^{201}Tl should be investigated. It may also be worthwhile for the agencies to review the use of external dose factors for ^{111}In and ^{201}Tl, where significant differences have been found.
7	Differences in the suspended solids load in sludge also affect activity concentrations and estimates of dose to STW workers and the farming family; the doses to the sewer workers calculated by the EA model are around 1.5 times higher than those estimated by the HPA SMART model.	
8	The FSA model only considers exposure from sludge tanks in the STW, while the other models consider exposure from both effluent and sludge. As activity concentrations in sludge tend to be higher than concentrations in effluent, doses estimated by the FSA model tend to be higher than those from the other models by about one or two orders of magnitude.	
9	In models where exposures from effluent and sludge are considered separately, the overall exposure time is split differently. The EA model assumes that STW workers spend the majority of their time close to the sewage tanks, while the HPA SMART model assumes that workers spend the majority of their time close to the sludge tanks or the solids component of the sewage treatment stages, where activity concentrations are higher. As a result the HPA SMART model calculates higher doses to STW workers.	A review of the parameter values used in the models would be beneficial. In particular it would be useful to focus on how occupancies are split at the STW.

Table 9: Findings and recommendations of the 2010 sewer intercomparison exercise

	Finding	Recommendation
10	The models use different application rates of sludge to land, with the FSA model using an application rate that is 20 times lower than that used in the other models.	It is recommended that the method used to calculate doses to the farming family is reviewed. This should include reviewing the sewage application rates and how often it is applied and whether the sewage is injected or ploughed. Some models may not consider certain radionuclides when calculating doses to the farming family, as they assume that they have decayed significantly by the time sewage sludge is applied to farmland. The agencies should review the assumptions made on this matter.
11	The FSA model does not consider ingestion of soil for the farming family. Ingestion of soil tends to give doses that can be one or two orders of magnitude higher than inhalation of soil; as a result the doses estimated by the FSA model are lower.	
12	There is a difference in exposure time for the farming family between the FSA model and the HPA and EA models. The EA and HPA models assume that exposure occurs all year round, while the FSA model assumes an exposure time of 1000 hours.	
13	Both the EA and HPA SLUDLAND models consider doses to all age groups, including the fetus, whereas the FSA model only considers adults.	
14	The EA model assumes that the farming family ingest and inhale soil only when they are outside, whereas the HPA model assumes that ingestion and inhalation of soil occur at any time. This results in the HPA model using an exposure time of twice that used in the EA model.	
15	The HPA SLUDLAND model uses higher than average consumption rates for only beef and milk, whereas the EA and FSA consider higher than average consumption rates for all food types. However, this does not lead to a large difference in the doses estimated	
16	The EA model does not include ingestion of dairy products other than milk. In the HPA SLUDLAND model the contribution of milk to the total dose from ingestion of food is almost equal to that of dairy products for ^{32}P and ^{131}I, which concentrate in milk. Therefore doses from ingestion of terrestrial food calculated by the EA model for these radionuclides are about half of those estimated by the HPA SLUDLAND if all other factors are equal.	
17	The concentration factor for milk of ^{131}I is a lot higher in the FSA model, possibly due to differences in the intake rates of fodder by the cows and in the fodder to cow transfer rates assumed in the model	
18	There are some differences between the models regarding which radionuclides are considered when doses from sludge applied to farmland are calculated.	
19	The models are largely fit for the purpose of assessing whether a discharge will lead to doses that exceed the relevant dose limits. It is recognised that as screening models for prospective assessments, the models will tend to overestimate the actual dose that are received.	Although it is recognised that validating models is difficult, it would be very useful to compare model estimates with real measurements of activity concentrations in effluent and sludge, such as those available from SEPA. It may be possible to achieve this through use of an MSc student.

7 REFERENCES

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APPENDIX A

Calculation of the Line Strength, C_L

The FSA model is based on a study by the Centre for Environment, Fisheries and Aquaculture Science (CEFAS) (Brownless and Round, 2000). It treats the pipe as an infinite line source and calculates the line strength, the activity per unit length of pipe, C_L (Bq m^{-1}). This line strength, C_L, is based on the amount of solid deposited in the sewer pipe, and calculated as the cross-section area, A (m^2), of sediment in the pipe multiplied by the activity concentration (C_{sed}, Bq m^{-3}) in the sediment:

$$C_L = A\, C_{sed}$$

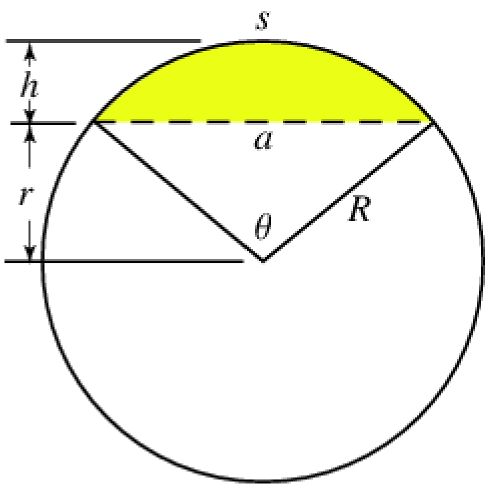

Figure A1. Cross section of a sewer pipe, showing the area of deposited sediment

Figure A1 shows a sewer pipe of radius R; A is the area in yellow, h is the depth of sediment on the pipe wall and r is the difference between R and h. A is the difference between the area of the sector θ of the circle with radius R, $\pi R^2 \frac{\theta}{2\pi}$, and the area of the triangle with sides a, R and R and angle θ, $\frac{R^2}{2}\sin\theta$:

$$A = \pi R^2 \frac{\theta}{2\pi} - \frac{R^2}{2}\sin\theta = \frac{R^2}{2}(\theta - \sin\theta)$$

From Figure A1:

$$\cos\theta = \frac{r}{R} \quad \text{or} \quad \theta = \arccos\frac{r}{R}$$

A therefore becomes:

$$A = \frac{R^2}{2}\left(2\arccos\left(\frac{r}{R}\right) - \sin\left(2\arccos\left(\frac{r}{R}\right)\right)\right) = R^2\arccos\left(\frac{r}{R}\right) - \frac{R^2}{2}\sin\left(2\arccos\left(\frac{r}{R}\right)\right)$$

As sin 2θ = 2 sin θ cos θ, A becomes:

$$\begin{aligned} A &= R^2\arccos\left(\frac{r}{R}\right) - \frac{R^2}{2}2\sin\left(\arccos\left(\frac{r}{R}\right)\right)\cos\left(\arccos\left(\frac{r}{R}\right)\right) \\ &= R^2\arccos\left(\frac{r}{R}\right) - R^2\sin\left(\arccos\left(\frac{r}{R}\right)\right)\frac{r}{R} = R^2\arccos\left(\frac{r}{R}\right) - rR\sin\left(\arccos\left(\frac{r}{R}\right)\right) \end{aligned}$$

The activity concentration, C_{sed}, in the sediment deposited on the pipe is calculated as:

$$C_{sed} = C_{eff}\, K_d\, \rho_{sed}\, R_{wd}$$

Where C_{eff} is the activity concentration in the effluent (Bq m^{-3}), K_d is the soil distribution coefficient (l kg^{-1}), ρ_{sed} is the density of sediment (kg l^{-1}) and R_{wd} is the wet to dry ratio of the sediment

Therefore, the line strength C_L (Bq m^{-1}) is calculated as:

$$C_L = A\, C_{sed} = \left(R^2\arccos\left(\frac{r}{R}\right) - rR\sin\left(\arccos\left(\frac{r}{R}\right)\right)\right) C_{eff}\, K_d\, \rho_{sed}\, R_{wd}$$

The equation implemented in the FSA model is slightly different:

$$C_L = A\, C_{sed} = \left(R^2\arccos\left(\frac{r}{R}\right) - R\sin\left(\arccos\left(\frac{r}{R}\right)\right)\right) C_{eff}\, K_d\, \rho_{sed}\, R_{wd}$$

The equation implemented in the FSA model is therefore incorrect and underestimates the line strength. In the model the radius of the pipe, R, and the depth of the sediment, h, are assumed to be 1 m and 0.2 m, respectively, meaning that r has a value of 0.8 m. The line strength calculated by the FSA model is therefore 3.8 times lower than the correct value.

References

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APPENDIX B

Model Data

B1 BASIC RADIONUCLIDE DATA

Table B1 shows the dose coefficients for adults used by the three agencies (HPA SMART and W63 use the same values) for the radionuclides considered in the 2010 intercomparison exercise. Two differences can be seen: the HPA uses a lower inhalation dose coefficient for ^{32}P than the other agencies, and the FSA uses different coefficients for ^{3}H. The use of a different coefficient for ^{32}P by the HPA is because the HPA assumes a dose coefficient for the public based on lung class F, as recommended by ICRP Publication 72 (ICRP 1996) and the other models use a dose coefficient for lung class M. Also different coefficients are used for ^{3}H, because the FSA model considers organically bound tritium, while the other models consider tritiated water. Tables B2 and B3 give the dose coefficients for inhalation and ingestion for children, infants and the fetus used when estimating doses to the farming family.

Table B1: Dose coefficients for adults

	Inhalation dose coefficient (Sv Bq^{-1})			Ingestion dose coefficient (Sv Bq^{-1})		
Radionuclide	EA model	FSA model	HPA models	EA model	FSA model	HPA models
^{3}H	$1.8\ 10^{-11}$	$4.5\ 10^{-11}$	$1.8\ 10^{-11}$	$1.8\ 10^{-11}$	$4.2\ 10^{-11}$	$1.8\ 10^{-11}$
^{32}P	$3.4\ 10^{-9}$	$3.4\ 10^{-9}$	$7.7\ 10^{-10}$	$2.4\ 10^{-9}$	$2.4\ 10^{-9}$	$2.4\ 10^{-9}$
^{99m}Tc	$1.9\ 10^{-11}$	$1.9\ 10^{-11}$	$1.9\ 10^{-11}$	$2.2\ 10^{-11}$	$2.2\ 10^{-11}$	$2.2\ 10^{-11}$
^{111}In	$2.3\ 10^{-10}$	$2.3\ 10^{-10}$	$2.3\ 10^{-10}$	$2.9\ 10^{-10}$	$2.9\ 10^{-10}$	$2.9\ 10^{-10}$
^{131}I	$7.4\ 10^{-9}$	$7.4\ 10^{-9}$	$7.4\ 10^{-9}$	$2.2\ 10^{-8}$	$2.2\ 10^{-8}$	$2.2\ 10^{-8}$
^{201}Tl	$4.4\ 10^{-11}$	$4.4\ 10^{-11}$	$4.4\ 10^{-11}$	$9.5\ 10^{-11}$	$9.5\ 10^{-11}$	$9.5\ 10^{-11}$

Table B2: dose coefficients for inhalation (Sv Bq^{-1}) for children, infants and fetus

	Child		Infant		Fetus	
Radionuclide	EA model	HPA SLUDLAND	EA model	HPA SLUDLAND	EA model	HPA SLUDLAND
^{3}H	$2.3\ 10^{-11}$	$2.3\ 10^{-11}$	$4.8\ 10^{-11}$	$4.8\ 10^{-11}$	$3.1\ 10^{-11}$	$3.1\ 10^{-11}$
^{32}P	$5.3\ 10^{-9}$	$1.8\ 10^{-9}$	$1.5\ 10^{-08}$	$7.5\ 10^{-9}$	$6.5\ 10^{-9}$	$1.0\ 10^{-8}$
^{99m}Tc	$3.4\ 10^{-11}$	NC	$9.9\ 10^{-11}$	NC	NC	NC
^{111}In	$4.1\ 10^{-10}$	NC	$1.2\ 10^{-9}$	NC	NC	NC
^{131}I	$1.9\ 10^{-8}$	$1.9\ 10^{-8}$	$7.2\ 10^{-8}$	$7.2\ 10^{-8}$	NC	NC
^{201}Tl	$9.4\ 10^{-11}$	NC	$3.3\ 10^{-10}$	NC	NC	NC

Key:

NC = not considered

Table B3: Dose coefficients for ingestion (Sv Bq^{-1}) for children, infants and fetus

	Child		Infant		Fetus	
Radionuclide	EA model	HPA SLUDLAND	EA	HPA SLUDLAND	EA	HPA SLUDLAND
^{3}H	$2.3\ 10^{-11}$	$2.3\ 10^{-11}$	$4.8\ 10^{-11}$	$4.8\ 10^{-11}$	$3.1\ 10^{-11}$	$3.1\ 10^{-11}$
^{32}P	$5.3\ 10^{-9}$	$1.9\ 10^{-8}$	$1.9\ 10^{-8}$	$1.9\ 10^{-8}$	$2.4\ 10^{-8}$	$2.5\ 10^{-8}$
^{99m}Tc	$4.3\ 10^{-11}$	NC*	$1.3\ 10^{-10}$	NC	NC	NC
^{111}In	$5.9\ 10^{-10}$	NC	$1.7\ 10^{-9}$	NC	NC	NC
^{131}I	$5.2\ 10^{-8}$	$5.2\ 10^{-8}$	$1.8\ 10^{-7}$	$1.8\ 10^{-7}$	NC	NC
^{201}Tl	$1.8\ 10^{-10}$	NC	$5.5\ 10^{-10}$	NC	NC	NC

Key:

NC = not considered

Table B4 shows effective gamma dose rates used in the models. The HPA SMART and W63 models use effective gamma dose rates at 1 m distance. The EA model uses external dose rates from raw sewage tanks at the STW. As described in Section 4.2.1, the FSA model uses an effective gamma dose rate, DR_γ (Sv h^{-1} per Bq t^{-1}) given by e equation:

$$DR_\gamma = DR_f\, \bar{E}_\gamma\, f_D$$

where DR_f is a dose rate factor per unit activity concentration ($2.88\ 10^{-13}$ Gy h^{-1} per Mev Bq^{-1} per Bq t^{-1}); $\bar{E}_\gamma$ is the mean gamma energy per disintegration (MeV Bq^{-1}); and f_D is a factor used to convert absorbed dose in air to effective dose (0.85 Sv Gy^{-1}).

The effective gamma dose rates adopted in the HPA W63 model are slightly higher than those used in the HPA SMART model. The effective gamma dose rates used by the HPA W63, HPA SMART and FSA models are within 25% of each other. Although these values cannot be directly compared with the EA dose rates, which include shielding from tanks, they are within a factor of 3. The EA model also includes an external dose factor for ^{32}P that is not included in the other models.

Table B4: Gamma dose rates and mean gamma energies used in the calculation of doses from external exposure

	Mean gamma energy per disintegration (MeV Bq^{-1})	Effective gamma dose rates (Sv h^{-1} per Bq t^{-1})			
Radionuclide	FSA model	FSA model	EA model	HPA W63	HPA SMART
^{3}H (HTO)*	0.0				
^{32}P*	0.0		$6.3\ 10^{-16}$		
^{99m}Tc	$1.26\ 10^{-1}$	$3.08\ 10^{-14}$	$1.5\ 10^{-14}$	$3.4\ 10^{-14}$	$3.0\ 10^{-14}$
^{111}In	$4.05\ 10^{-1}$	$9.91\ 10^{-14}$	$5.6\ 10^{-14}$	$1.1\ 10^{-13}$	$8.8\ 10^{-14}$
^{131}I	$3.81\ 10^{-1}$	$9.33\ 10^{-14}$	$6.2\ 10^{-14}$	$9.8\ 10^{-14}$	$8.1\ 10^{-14}$
^{201}Tl	$9.32\ 10^{-2}$	$2.28\ 10^{-14}$	$7.5\ 10^{-15}$	$2.4\ 10^{-14}$	$2.0\ 10^{-14}$

Note:

*: Emissions with a gamma energy less than 0.01 MeV Bq^{-1} are ignored in the calculation of gamma dose rates

B2 PARTITIONING DATA

The EA and both the HPA SMART and W63 models use partitioning factors, or removal coefficients, indicating the fraction of radionuclide that goes into the effluent and the fraction that goes into the sludge. The FSA model uses soil distribution coefficients (K_d) to determine how much activity ends up in the sludge. Table B5 gives partitioning factors and K_d values used by different models for the radionuclides considered in the 2010 intercomparison exercise.

Table B5: Partitioning factors and K_d values used in the sewer models

	Partitioning factors				K_d value in FSA model (l kg^{-1})
	EA model		HPA models (secondary stage)*		
Radionuclide	Effluent	Sludge	Effluent	Sludge	
^{3}H (HTO)	0.85	0.15	0.85	0.15	$3.0\ 10^{-2}$
^{32}P	0.2	0.8	0.5	0.5	$1.1\ 10^{2}$
^{99m}Tc	0.9	0.1	0.9	$0.1^{\#}$	$1.5\ 10^{0}$
^{111}In	0.1	0.9	0.2	0.8	$2.4\ 10^{3}$
^{131}I	0.8	0.2	0.8	0.2	$1.8\ 10^{2}$
^{201}Tl	0.5	0.5	0.2	0.8	$2.2\ 10^{4}$

Notes:

*: The HPA SMART model includes a partitioning factor for each treatment stage (primary, secondary, tertiary). For most radionuclides the factor for the primary stage is 1 (ie all the activity stays in the effluent) so the secondary stage factors are shown in this table. However, for iodine some partitioning is assumed to occur at the primary treatment stage, with 0.93 of the original activity partitioned to the effluent and 0.07 to the sludge in the primary treatment

#: The HPA W63 model uses a different removal efficiency (0.9) for ^{99m}Tc to include an allowance for radioactive decay

B3 HABIT DATA

Table B6 shows the habit data used by the models when estimating doses to sewer pipe maintenance workers and STW workers. For STWs the table shows overall occupancy times at the facility. In the EA and HPA SMART model different occupancy times are used for different locations, close to sewage and sludge tanks; the breakdown differs between models. Table B7 shows the habit data used when estimating doses to the farming family. The higher than average consumption rates shown in the table for the FSA models are taken from the FSA spreadsheet model used to calculate doses for this comparison exercise. Consumption rates used in the EA model are all average values. The HPA SLUDLAND model uses higher than average consumption rates for beef and milk and average values for all other foods; the W63 model uses higher than average values for all foods.

Table B6: Habit data for sewer pipe maintenance and STW workers

Parameter	EA model	FSA model	HPA SMART/ SLUDLAND	HPA W63
Exposure time of sewer pipe maintenance workers (h y^{-1})	N/A	200	1600	N/A
Exposure time of STW workers (h y^{-1})	2000	1000	1800	2000
Breathing rate (m^3 h^{-1})	1.2	1.38	1.2	1.2
Ingestion rate of sewage material (kg h^{-1})	5 10^{-6}	4 10^{-6}	5 10^{-6}	5 10^{-6}

Table B7: Habit data for farming family

Parameter		EA model	FSA model	HPA SMART/ SLUDLAND	HPA W63
Application rate of sludge to land (kg m^{-2} y^{-1})		8	0.4	8	8
Exposure time (h y^{-1})		8760	1000	8760	8760
Fraction of time spent indoors	Adult	0.5	0.885	–	–
Inhalation rate (adult, m^3 h^{-1})		0.92	1.38	0.93	0.92
Ingestion rate of soil (kg y^{-1})	Adult	0.0083	–	0.0083	–
	Infant	0.044	–	0.044	–
Ingestion of cow meat (adult†, kg y^{-1})		45	45	45	45
Ingestion of cow offal (adult†, kg y^{-1})		10	10*	2.75	10*
Ingestion of cow milk (adult†, kg y^{-1})		240	240	240	240
Ingestion of milk products (adult†, kg y^{-1})		–	–	20	60
Ingestion of sheep meat (adult†, kg y^{-1})		25	25	8	25
Ingestion of sheep offal (adult†, kg y^{-1})		10	10*	2.75	10*
Ingestion of green vegetables (adult†, kg y^{-1})		80	45	N/A	N/A
Ingestion of root vegetables (adult†, kg y^{-1})		130	160#	N/A	N/A

Notes:

*: Where a single rate is given for offal this has been equally split between cow offal and sheep offal.

#: Includes potatoes and root vegetables

†: The FSA model do not calculate doses to farming family children as there is less available data on food consumption rates for children than for adults. The EA and HPA models use consumption rates for children that are approximately half of the rates for adult

References

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