Cefas

Cefas contract report RB101

Testing and Validation of the PRAME modelling suite

February 2007

Science commissioned by the
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Testing and Validation of the PRAME modelling suite

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Summary

Cefas' probabilistic assessment tool for modelling dose from liquid discharges to the marine environment is tested and validated. Input parameter distributions are subjected to a variety of modifications in order to determine the effects of improving the accuracy of input, whilst maintaining a realistic level of variation and uncertainty. A clear distinction in the behaviour of so-called conservative and non-conservative radionuclides is observed, and the reason for this is investigated by testing the sensitivity of the concentration of radionuclide in water to changes in the sediment distribution coefficient (K_d). Validation exercises are performed at two nuclear power stations, using site-specific input information. The results show a very good agreement with water concentrations suggested by environmental monitoring data, and confirm the validity of using the PRAME modelling suite to predict water concentrations (and hence effective dose) due to routine liquid discharge to the marine environment from nuclear installations.

Contents

Introduction

Under the Radioactive Substances Act 1993, as amended by the Environment Act 1995, the Food Standards Agency (FSA) is a statutory consultee on applications for Authorisations for discharge of radioactive waste to the environment. These discharges can be of solid, gaseous or liquid waste. We concern ourselves here with the liquid discharges, which are typically delivered to the marine environment via dedicated offshore pipelines but can, in the case of some low-level waste, be via public sewer systems.

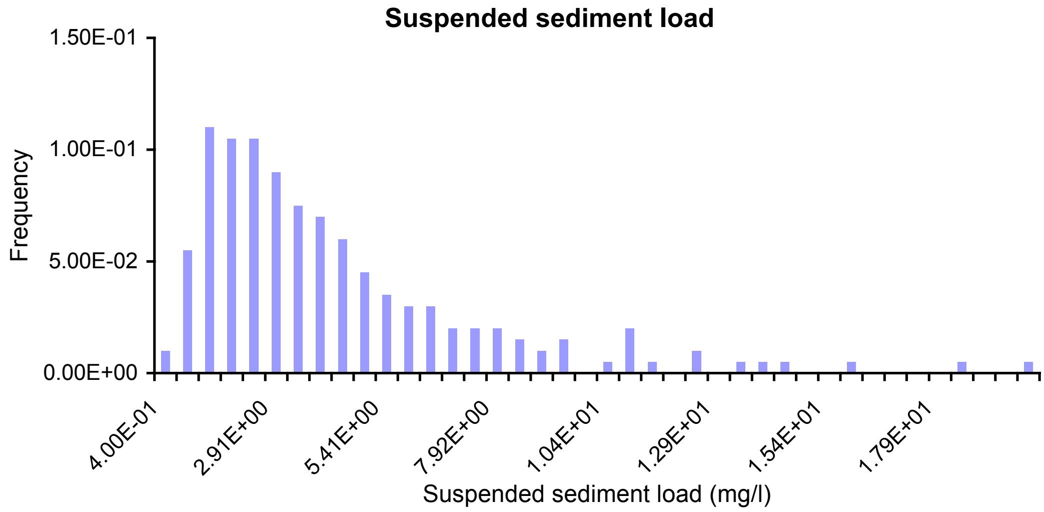
Consequently, the FSA requires the capability for modelling the transport of radionuclides in the marine food chain in order to assess the impact to human health of routine releases of liquid waste to the marine environment. This report describes the testing and validation of Cefas' probabilistic assessment tool, the Probabilistic Radiological Assessments of the Marine Environment (PRAME) modelling suite. Details of the development and use of PRAME can be found in Grzechnik *et al. (2002)* and Grzechnik *et al. (2006)*, whilst the mathematical basis of the underlying deterministic model is described by Round *(1998a,b)*.

The PRAME model aims to incorporate the uncertainty and variability of the most sensitive parameters involved in contaminant transport modelling. By describing each parameter as a probabilistic distribution, rather than a fixed value, PRAME reproduces the natural variation in time and space, as well as any uncertainty in the measurement and definition, of the input parameters. For example, the sediment suspended load of the water column can be considered both uncertain and variable – the impracticality of measuring its value over the whole model domain leads us to estimate a value from a handful of measurements. This uncertainty is then compounded by the variation of suspended sediment due to tidal and meteorological conditions and the composition of the seabed.

The output of the PRAME model consists of two distinct elements, both of which benefit from the propagation through the model of the probabilistic input method. Firstly, the concentration in water (in Bq/l) of each selected nuclide is calculated. This is presented as a probabilistic distribution, reflecting the variability and uncertainty of the model parameters. This distribution is in turn used as an input parameter in the dosimetric module of PRAME to provide a distribution of the dose equivalent (in μSv/y) due to each nuclide and pathway of exposure.

On completion of the model run, PRAME generates a series of charts displaying the shape and range of the distribution of each input parameter and the resultant outputs (Figure 1). These charts clearly illustrate the advantages of the probabilistic approach. The likelihood of each dose equivalent or water concentration can easily be observed, and links between output and input can be explored. Additionally, percentile values, uncertainty ratios, maxima and minima, and other statistical data relating to each distribution are calculated and displayed. The overall effect is to give the user a better idea of what output is *likely*, as opposed to a deterministic model, which, due to the parameter uncertainties mentioned above, simply provides one *possible* output.

(a)



(b)

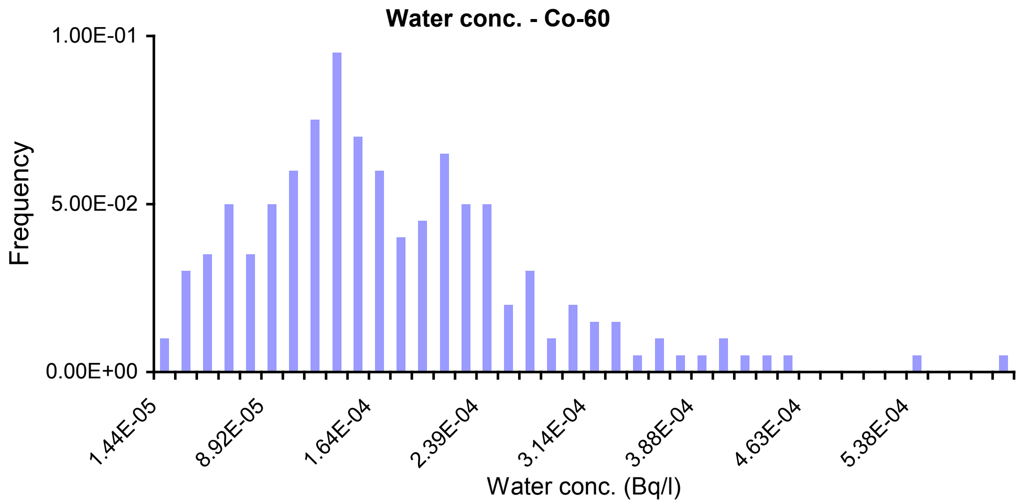


Figure 1 *Examples of the distribution charts generated by PRAME. (a) - input suspended sediment load distribution, (b) - output water concentration distribution. Horizontal axes show the input/output values, vertical axes show the frequency (or probability) with which each input/output interval was attained*

Testing and validation of the PRAME model is undertaken to ensure that it is suitable for its intended purpose of assessing the risk to the public of liquid radioactive discharges. We must make sure that the probabilistic approach is able to provide an output that is not only reflective of the uncertainty of input variables, but also constricted enough to allow proper evaluation of whether certain radiological thresholds are in danger of being exceeded. This report includes an appraisal of PRAME's ability to reflect changes in input distributions and a brief study of the sensitivity of output to uncertainty in the sediment distribution coefficient (the K_d value). We then provide a validation of the model by using site-specific input distributions to compare the model output to environmental monitoring data.

It is not the purpose of this report to test every feature of PRAME and each conceivable input for bugs or anomalies. Development of PRAME is ongoing, and it is intended that any such issues that arise during use of the model shall be addressed as necessary.

Of PRAME's two outputs, we shall use only one – the water concentration distribution – in the testing and validation procedure. This output can be verified against a number of marine samples using concentration factors (*IAEA 2004*), which relate the concentration of an element in marine biota with that of the surrounding water body. There is no comparable method of verifying the effective dose distribution; PRAME relies largely on its own water concentration module and published dose coefficients (*ICRP, 1994, 1995a,b,c*) to output a distribution of the effective dose.

Testing the PRAME model

A number of test runs of the PRAME model are employed to investigate the sensitivity of output to changes in different groups of input parameters. We shall initially use a very general set of input parameters to find the resultant water concentration from four radionuclides with an annual discharge of 1 TBq each. These are chosen to represent both conservative (^{3}H, ^{137}Cs) and non-conservative (^{241}Pu, ^{241}Am) nuclides, and as such provide a wide range of sediment distribution coefficients (hereafter referred to as the K_d value).

Subsequent model runs will examine the effect of improved parameter information. In so doing, it is shown that some investment into reducing the uncertainty associated with specific parameters can be beneficial in terms of providing a more focussed output distribution. An overview of this part of the test schedule is given in Table 1.

The second part of the test schedule concentrates on the relationship between the suspended sediment load and the K_d value. We shall observe a well-defined transition region between conservative and non-conservative nuclides, confirming the importance of the suspended sediment load in the modelling of radionuclide transport. Throughout the test schedule, an annual discharge of 1 TBq of each nuclide is assumed.

Test number	Feature(s) tested	Variation from control run	Justification / reason for parameter changes
2	Improved critical group and hydrographical parameter information	Specified distance to critical group and tidal excursion, narrowed range of mean depth and residual velocity	Use of Admiralty charts, computer software and habits surveys enables the uncertainty of these parameters to be reduced
3	Improved sediment parameter information	Specified suspended sediment load and sediment ratio; K_d values fixed to IAEA Tech. Rep. 422 recommended value	Analysis of environmental data and published research should enable such parameters to be better defined
4	Combination of tests 2 and 3	Combination of tests 2 and 3	As above
5	Narrowed range of diffusion coefficient distribution	As test 2 with narrower range of diffusion coefficient distribution	Diffusion coefficient often used as a 'tuning' parameter

Table 1 *Overview of the PRAME test schedule. Test 1 is the control run.*

Test 1 – Control run

We first perform a control run of PRAME using the parameter distributions shown in Table 2. Each parameter is sampled 200 times and the model is run in advection-diffusion mode. The purpose of this control run is to provide a reference output with which to compare that of later test runs. A description of each parameter and the dimension in which it should be entered is given in Grzechnik *et al.* (*2006*). As we are, for now, concerned only with testing the modelling of the water concentration, the input parameters on pages 5 and 6 of the PRAME GUI are redundant and can be set to any valid quantity without affecting the testing procedure.

Test 2

There are a number of ways of refining the information used to select the input parameter distributions. The use of Admiralty charts and marine mapping software can give a better idea of the bathymetry and residual currents around the point of discharge, and Cefas' habits surveys provide detailed information on the location and activities of critical groups. To simulate the employment of the above, PRAME was run with narrower mean depth and residual velocity input distributions and fixed parameter values for the tidal excursion and distance to critical group. A summary of these modifications is given in Table 3.

Test 3

We now consider the effect of refining the input distributions representing the various sediment parameters. There is a great deal of literature, summarised by the IAEA (*2004*), concerning the K_d value of various elements. This may be utilised to narrow down the distribution of the K_d value according to regional and temporal factors. The IAEA recommended values are used in this test. Researchers in a number of fields routinely measure the suspended sediment load – the challenge lies in obtaining and assimilating the data – and this can

Parameter		*Lower bound*	*Upper bound*	*Percentile*	*Distribution*
Suspended sediment load		2	200	99	Log-normal
Sediment ratio		4×10^4	6×10^4	99	Uniform
Mean depth		3	50	99	Normal
Residual velocity		0.02	0.07	99	Log-normal
Diffusion coefficient		0.1	10	99	Normal
Half tidal excursion*		4×10^3	8×10^3	99	Normal
Discharge start time		0	0.5	100	Uniform
Discharge end time		0.5	1	100	Uniform
Initial spreading radius		25	75	95	Normal
Distance to critical group		0	1×10^4	95	Normal
K_d value	^{3}H	0.1	100	99	Log-normal
	^{137}Cs	400	4×10^5	"	"
	^{241}Pu	1×10^4	1×10^6	"	"
	^{241}Am	2×10^5	2×10^7	"	"

Table 2 *Input parameter distributions for the PRAME control run (Test 1). *Half tidal excursion refers to the half tidal excursion at both the critical group location and the outfall pipe.*

then be used to estimate the sedimentation rate. Information regarding the composition of the seabed in the area of discharge may also be useful in defining the sediment parameters input distributions. We shall assume that there is a substantial body of data available and fix both the suspended sediment load and sediment ratio; details are shown in Table 3.

Test 4

Test 4 combines the modifications of Tests 2 and 3 to demonstrate how refining the input distributions of both the hydrographic and sediment parameters can further reduce the uncertainty of output.

Test 5

The diffusion coefficient is often used to 'tune' contaminant dispersion models due to the large degree of uncertainty associated with it. With this in mind, Test 5 seeks to illustrate the effect of increasing the lower bound, and consequently narrowing the range, of this parameter. This modification is used in conjunction with the hydrographic parameter improvements introduced in Test 2. Table 3 confirms the input parameter distributions used in Test 5.

Test Results

The results of the control run are shown in Figure 2, and some statistical data about each distribution is listed in Table 4. The horizontal axes show the water concentration of the nuclide whilst the vertical axes show the probability of each concentration interval being attained. The probability distribution charts pertaining to the input parameters are shown in Appendix A.

<table>
<tr><th>Test number</th><th>Parameter changes from control run</th><th>Lower bound</th><th>Upper bound</th><th>Percentile</th><th>Distribution</th></tr>
<tr><td rowspan="4">2</td><td>Mean depth</td><td>10</td><td>25</td><td>99</td><td>Normal</td></tr>
<tr><td>Residual velocity</td><td>0.03</td><td>0.05</td><td>99</td><td>Log-normal</td></tr>
<tr><td>Half tidal excursion</td><td>6×10^3</td><td>6×10^3</td><td>100</td><td>Fixed</td></tr>
<tr><td>Distance to critical group</td><td>7×10^3</td><td>7×10^3</td><td>100</td><td>Fixed</td></tr>
<tr><td rowspan="6">3</td><td>Suspended sediment load</td><td>20</td><td>20</td><td>100</td><td>Fixed</td></tr>
<tr><td>Sediment ratio</td><td>5×10^4</td><td>5×10^4</td><td>100</td><td>“</td></tr>
<tr><td>K_d value ^{3}H</td><td>1</td><td>1</td><td>100</td><td>“</td></tr>
<tr><td>K_d value ^{137}Cs</td><td>4×10^3</td><td>4×10^3</td><td>“</td><td>“</td></tr>
<tr><td>K_d value ^{241}Pu</td><td>1×10^5</td><td>1×10^5</td><td>“</td><td>“</td></tr>
<tr><td>K_d value ^{241}Am</td><td>2×10^6</td><td>2×10^6</td><td>“</td><td>“</td></tr>
<tr><td>4</td><td>Combination of changes in Tests 2 and 3</td><td>See above</td><td>See above</td><td>See above</td><td>See above</td></tr>
<tr><td>5</td><td>As Test 2, except:

Diffusion coefficient</td><td>5</td><td>10</td><td>99</td><td>Normal</td></tr>
</table>

Table 3 *Summary of changes to input parameter distributions in Tests 2 – 5*

In each case, the range of calculated water concentrations varies over several orders of magnitude. This variation is a direct result of the independent random sampling of input parameters over their defined range. The difference in nuclide concentration distributions is due to their differing K_d values, the range of which explains the disparity of each factor of uncertainty (Table 4, defined as the ratio of the 95th and 5th percentiles).

	^{3}H	^{137}Cs	^{241}Pu	^{241}Am
Minimum	4.56×10^{-3}	8.36×10^{-4}	1.20×10^{-4}	9.57×10^{-6}
Maximum	0.183	0.180	0.107	2.73×10^{-2}
5^{th} %ile	1.10×10^{-2}	4.34×10^{-3}	9.39×10^{-4}	4.97×10^{-5}
95^{th} %ile	8.09×10^{-2}	6.97×10^{-2}	3.35×10^{-2}	8.30×10^{-3}
Factor of uncertainty	7.36	16.1	35.7	167

Table 4 *Selected water concentration values (Bq/l) and statistical data from the control run*

It can be seen from Figure 2 that the higher water concentrations are mainly isolated examples, or outliers, and that on balance of probability the actual water concentration is likely to be towards the lower end of the possible range. These outliers are effectively neglected by the factor of uncertainty, which is thus a reliable indicator of the likely range of water concentration. It is this factor that we are attempting to reduce by modifying the input parameter distributions, whilst maintaining a realistic level of uncertainty and variability.

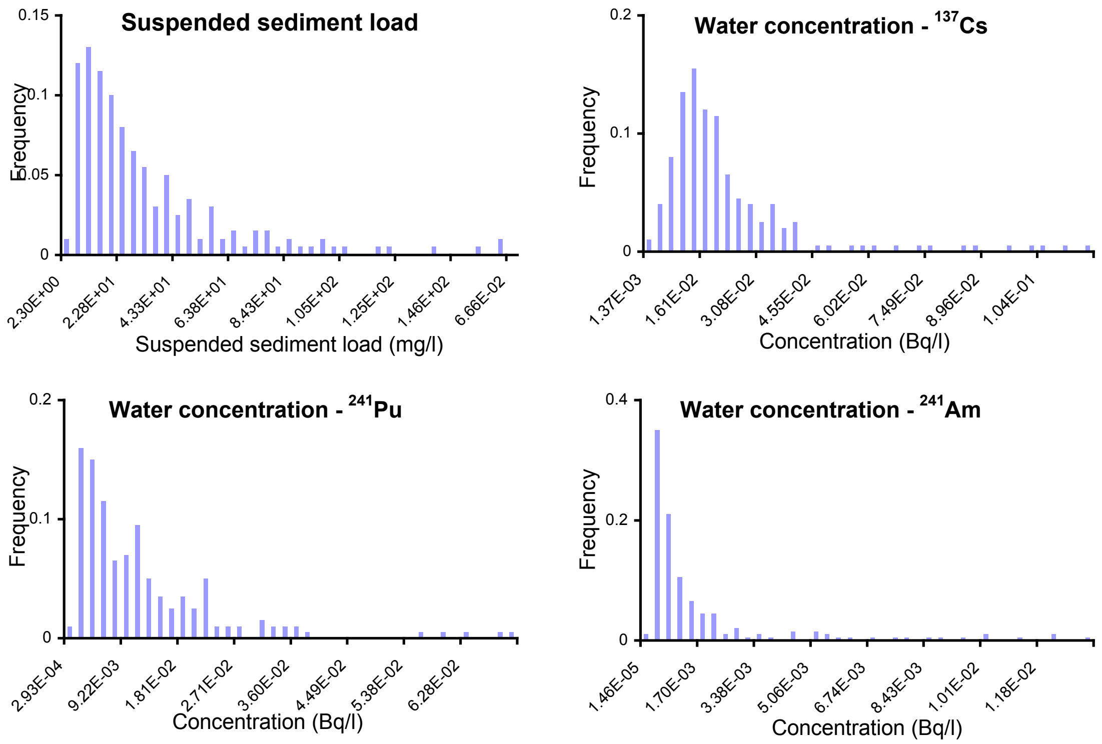


Figure 2 *Control run output showing the water concentration distributions of (a) ^{3}H, (b) ^{137}Cs, (c) ^{241}Pu and (d) ^{241}Am. Each distribution displays a long 'tail' consisting of isolated larger values*

For ease of assessment, the output concentration distributions of Tests 2 – 5 are grouped by nuclide. In this way the effect of each set of input parameter modifications can be compared, and in particular the different effects on conservative and non-conservative nuclides can be seen.

^{3}H water concentration distribution

Figure 3 illustrates the evolution of the concentration of ^{3}H through the test schedule. This shows how the output distribution changes with each of the PRAME test runs, in particular the insensitivity of output to changes in sediment parameters. Table 5 contains some statistical data relevant to these output distributions, and from this it can be seen that the factor of uncertainty is halved when the hydrographic modifications of Tests 2 and 4 are introduced.

	Test 2	*Test 3*	*Test 4*	*Test 5*
Minimum	1.86×10^{-2}	4.57×10^{-3}	1.86×10^{-2}	1.57×10^{-2}
Maximum	0.220	0.183	0.220	5.77×10^{-2}
5th %ile	2.22×10^{-2}	1.10×10^{-2}	2.22×10^{-2}	1.97×10^{-2}
95th %ile	7.25×10^{-2}	8.09×10^{-2}	7.25×10^{-2}	4.11×10^{-2}
Factor of uncertainty	3.27	7.36	3.27	2.09

Table 5 *Selected values of the output distribution of water concentration for ^{3}H, and the associated factor of uncertainty*

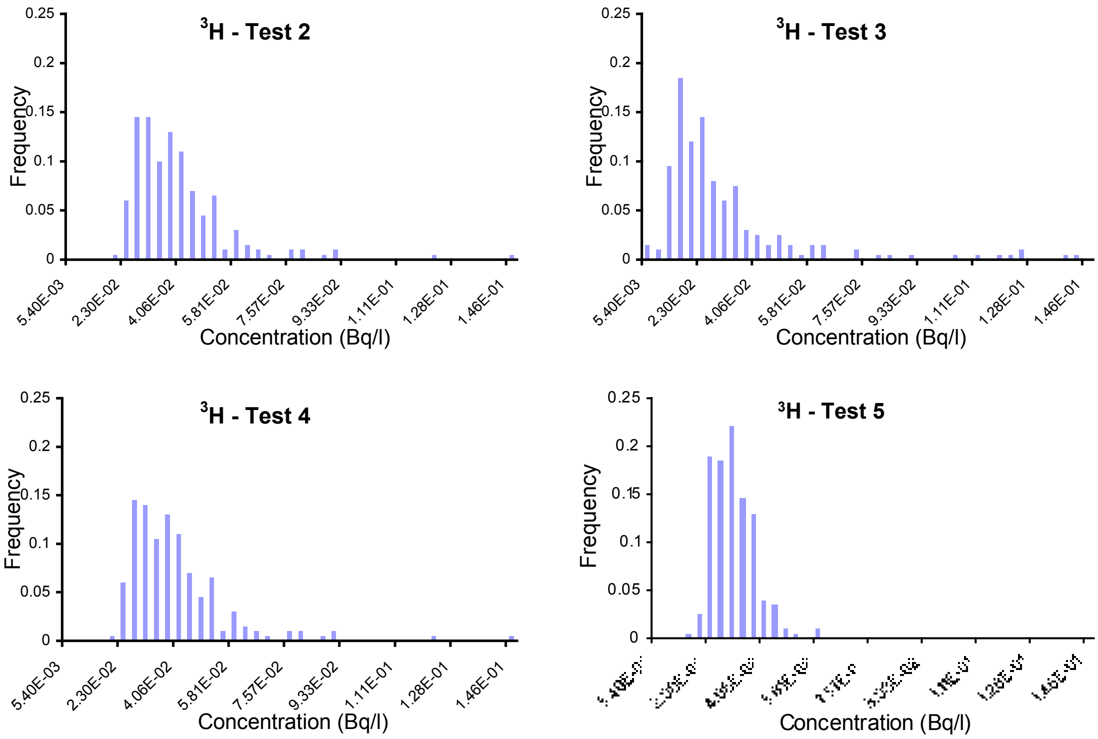


Figure 3 *Concentration distributions of ^{3}H (Bq/l). Note that the horizontal and vertical axes have been adjusted to maintain the same scale, aiding comparison of the different distributions.*

^{137}Cs water concentration distribution

Output concentrations of ^{137}Cs are shown in Figure 4, with summary statistics contained in Table 6. As for tritium, improving the hydrographic input parameters reduces the uncertainty of concentration, but in this case it is further reduced by the sediment parameter modifications of Test 3. This is due to the (recommended) K_d value of ^{137}Cs being higher than that of tritium, although both are considered to be conservative nuclides.

	Test 2	*Test 3*	*Test 4*	*Test 5*
Minimum	1.23×10^{-3}	4.23×10^{-3}	1.72×10^{-2}	1.02×10^{-3}
Maximum	0.217	0.169	0.204	4.95×10^{-2}
5^{th} %ile	7.60×10^{-3}	1.02×10^{-2}	2.05×10^{-2}	6.39×10^{-3}
95^{th} %ile	6.05×10^{-2}	7.48×10^{-2}	6.71×10^{-2}	3.59×10^{-2}
Factor of uncertainty	7.96	7.34	3.28	5.62

Table 6 *Selected values of the output distribution of water concentration for ^{137}Cs, and the associated factor of uncertainty*

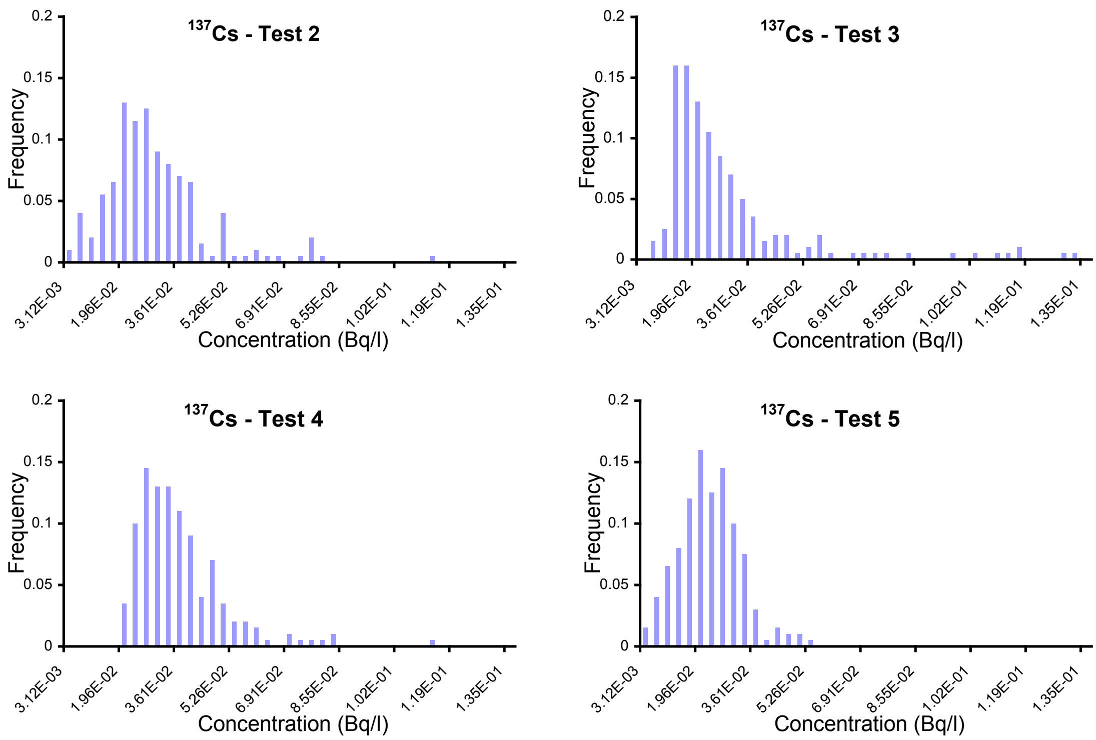


Figure 4 *Concentration distributions of ^{137}Cs (Bq/l). Note that the horizontal and vertical axes have been adjusted to maintain the same scale, aiding comparison of the different distributions.*

^{241}Pu water concentration distribution

Figure 5 charts the development of the ^{241}Pu output distribution. As a non-conservative nuclide, it is strongly affected by the sediment parameter modifications of Tests 3 and 4, resulting in a reduction in the factor of uncertainty associated with this nuclide (Table 7) from 35.7 after the control

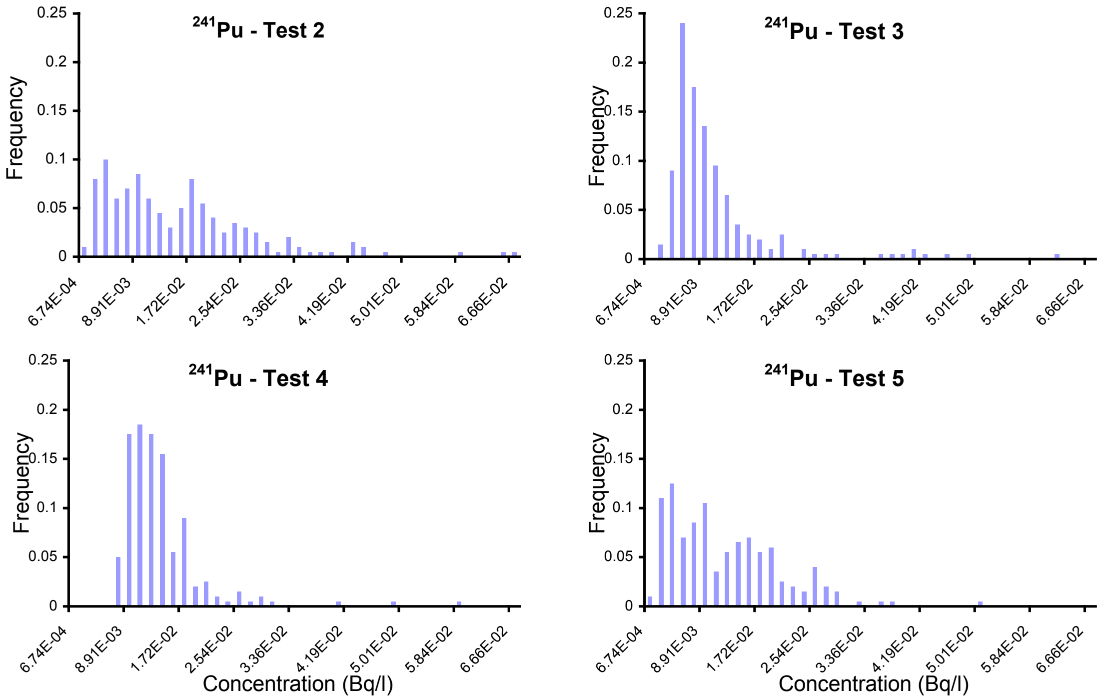


Figure 5 *Concentration distributions of ^{241}Pu (Bq/l). Note that the horizontal and vertical axes have been adjusted to maintain the same scale throughout, aiding comparison of the different distributions.*

run to just 3.24 after Test 4. Whilst the improved hydrographic parameters of test 2 play a part in this reduction, it is clear that the tighter sediment parameter distributions are primarily responsible.

	Test 2	*Test 3*	*Test 4*	*Test 5*
Minimum	2.25×10^{-4}	1.52×10^{-3}	6.14×10^{-3}	2.11×10^{-4}
Maximum	0.108	6.05×10^{-2}	7.29×10^{-2}	5.01×10^{-2}
5th %ile	1.68×10^{-3}	3.65×10^{-3}	7.35×10^{-3}	1.50×10^{-3}
95th %ile	4.05×10^{-2}	2.65×10^{-2}	2.38×10^{-2}	2.61×10^{-2}
Factor of uncertainty	24.1	7.27	3.24	17.5

Table 7 *Selected values of the output distribution of water concentration for ^{241}Pu, and the associated factor of uncertainty*

^{241}Am water concentration distribution

The reduction in the factor of uncertainty of ^{241}Am over the test schedule is even more marked. Its high particle reactivity implies that the concentration of this nuclide in water is almost exclusively driven by the amount of sediment available for adsorption. Consequently, when the suspended sediment load is fixed in test 3, the uncertainty of the output concentration falls dramatically (Table 8). Also, whereas the factor of uncertainty of ^{241}Am does not proportionately reduce as much as it does for the other nuclides when comparing the control run to Test 2, it sees a far higher proportionate decrease from the control run to Test 4. This again demonstrates the importance of improving sediment parameter information, particularly when modelling the discharge of non-conservative nuclides, to reduce the factor of uncertainty of output. Figure 6 displays the output distributions associated with this nuclide.

	Test 2	*Test 3*	*Test 4*	*Test 5*
Minimum	1.83×10^{-5}	1.11×10^{-4}	4.48×10^{-4}	8.99×10^{-6}
Maximum	2.85×10^{-2}	4.41×10^{-3}	5.31×10^{-3}	1.48×10^{-2}
5^{th} %ile	8.59×10^{-5}	2.67×10^{-4}	5.36×10^{-4}	7.03×10^{-5}
95^{th} %ile	9.66×10^{-3}	1.92×10^{-3}	1.73×10^{-3}	6.62×10^{-3}
Factor of uncertainty	112	7.19	3.23	94.2

Table 8 *Selected values of the output distribution of water concentration for ^{241}Am, and the associated factor of uncertainty*

Test 6 – sensitivity of output to changes in the K_d value

We now briefly comment on the sensitivity of the output concentration to changes in the K_d value. It was previously remarked that the sediment distribution coefficient of many of the elements has been the subject of a number of research papers (*IAEA, 2004*). Despite this, there is still considerable uncertainty associated with this parameter due to the heterogeneity of coastal sediment types and seabed composition, and also the variation of the K_d with marine conditions which cannot be accounted for on the temporal scale of the PRAME model. Hence, the input distribution of the K_d value may often vary over several degrees of magnitude (as evidenced by the range of derived K_ds published in *IAEA, 2004*).

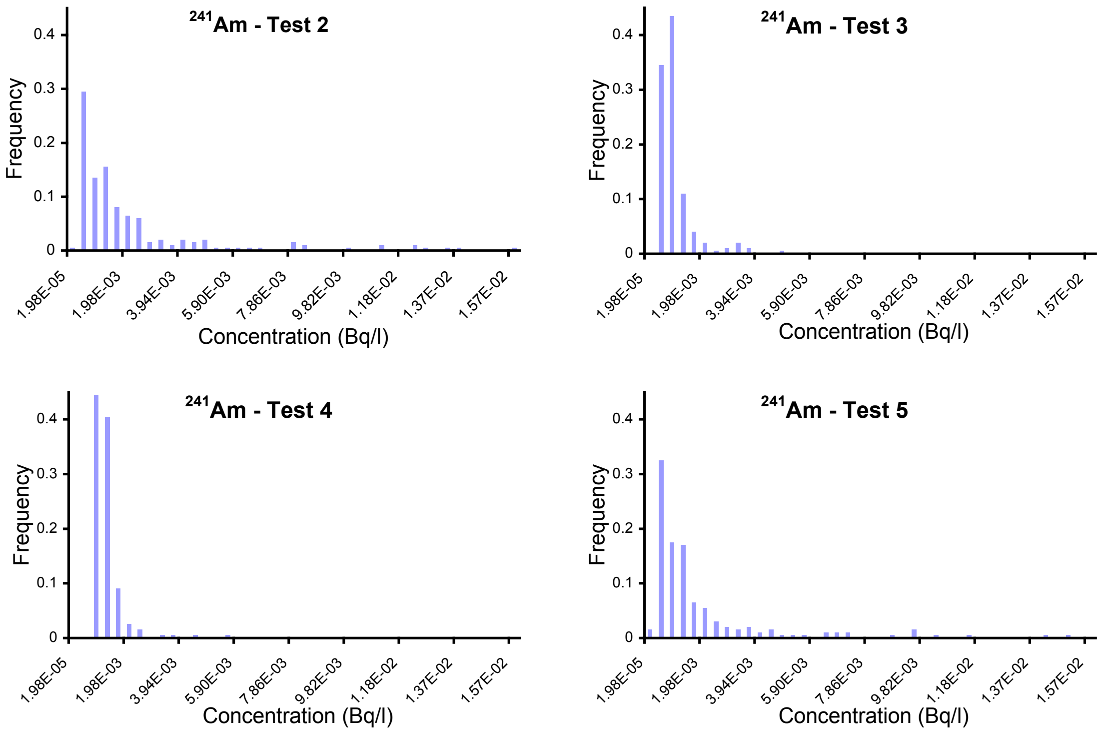


Figure 6 *Concentration distributions of ^{241}Am (Bq/l). Note that the horizontal and vertical axes have been adjusted to maintain the same scale throughout, aiding comparison of the different distributions.*

To examine the potential effect of this variation, a number of configurations of the PRAME model have been run in which all input parameters except the K_d are fixed. We then match the output water concentration of each individual PRAME run to the input K_d value to obtain a picture of the relationship between the two. This process is then repeated for different values of the suspended sediment load. The fixed parameter values are listed in Table 9, and the K_d value is assigned a log-normal distribution with 1st and 99th percentile values of 1 and 1×10^7 respectively. A discharge of 1 TBq per year is again assumed; the choice of nuclide is almost arbitrary[1].

Figure 7 shows the variation of output concentration with K_d value (note that the horizontal axis is scaled logarithmically). A steep transition can be observed between two regions in which the concentration remains relatively constant. Lower values of the K_d, leading to the higher of these water concentrations, are associated with conservative nuclides that remain in solution, whilst nuclides with a stronger affinity for the particulate phase would be found toward the right-hand side of the graph.

Parameter		*Value*
Suspended sediment load	Test 6	10
	Test 7	100
	Test 8	1000
Sediment ratio		5×10^4
Mean depth		15
Residual velocity		0.04
Diffusion coefficient		5
Half tidal excursion*		5×10^3
Discharge start time		0.54
Discharge end time		0.92
Initial spreading radius		50
Distance to critical group		0

Table 9 *The fixed parameter values used in place of input distributions for the* K_d *sensitivity test. *Half tidal excursion refers to the half tidal excursion at both the critical group location and the outfall pipe.*

The location of the transition region varies with the suspended sediment load. Essentially, the more sediment available in the water column the more likely the process of adsorption is to occur, and so a smaller K_d value is necessary to commence the shift from the dissolved to the particulate phase. A higher suspended sediment load also accelerates the transition between these states (recall that the horizontal axis is logarithmic), although in each of the examples in Figure 7 the gradient of the water concentration in the transition appears to be constant with respect to the logarithm of the K_d value.

[1] The only factor differentiating nuclides in this stage of the test is their respective half-lives. This can be shown to have no qualitative effect on the sensitivity of output to *Kd* value, and a negligible effect on quantitative output except in the case of very short-lived nuclides.

The importance of this sensitivity lies in the modelling of nuclides with K_ds that lie on or near to this transition region. In such cases, any input distribution of the K_d value may straddle the stable and transitional zones and lead to an increased factor of uncertainty in the output water concentration. Similarly, variations in the suspended sediment load could lead to an unforeseen rise or fall in water concentrations. The improvement of sediment parameter knowledge is therefore most important when the K_d distribution of any nuclides contained in the liquid discharge lies in the region of transition, and in particular when large values of the suspended sediment load may be encountered.

A comparison of the change in output concentration with K_d value of several different nuclides, for a fixed value of the suspended sediment load, is shown in Figure 8. These nuclides were selected for their wide range of half-lives and demonstrate that the choice of nuclide used in PRAME during this section of the testing procedure was indeed arbitrary.

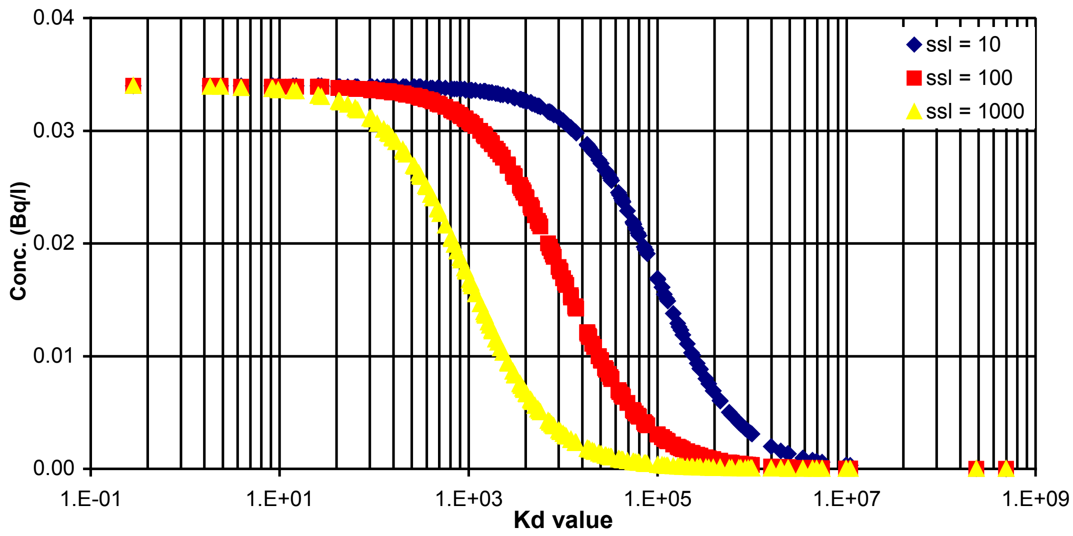


Figure 7 *Sensitivity of the output water concentration (Bq/l) to changes in the* K_d *value at different values of suspended sediment load (ssl). The horizontal axis is logarithmically scaled.*

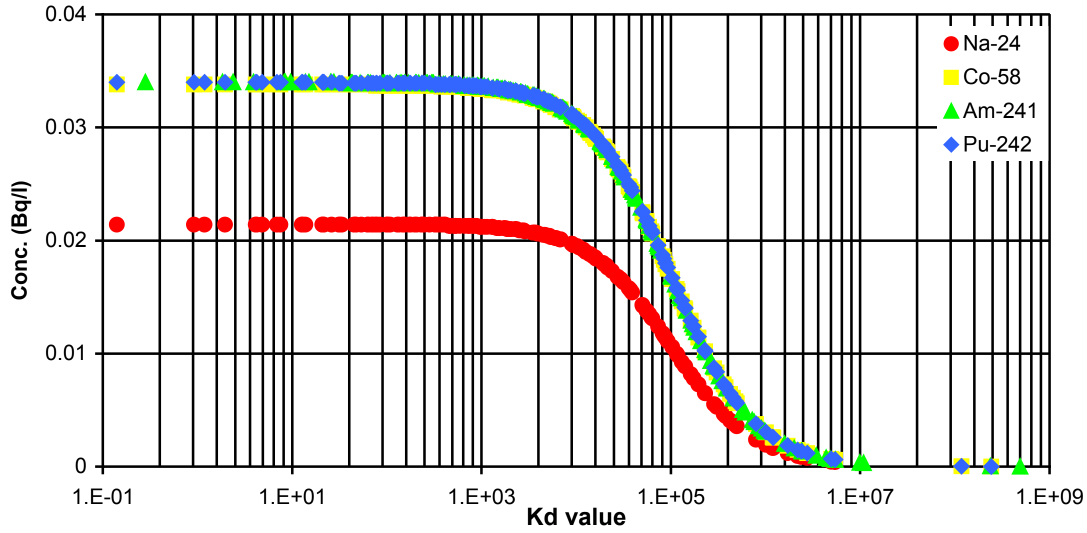


Figure 8 *Comparison of the sensitivity of different nuclides to changes in the K_d value for ssl = 10. Notice that the concentrations of ^{241}Am (half-life of 432.2 years), ^{242}Pu (3.76 x 10^5 years) and ^{58}Co (70.88 days) are indistinguishable from one another, whereas the concentration of ^{24}Na (half life of 14.96 hours) is significantly lower. The transition region between conservative and non-conservative concentrations remains unchanged. This shows that the choice of nuclide for the K_d sensitivity test was arbitrary with respect to qualitative behaviour, and indeed the concentration is only significantly reduced when the nuclide in question has a particularly short half-life. The horizontal axis is logarithmically scaled.*

Validation of the PRAME model

The previous section tested the PRAME model's response to changes in a variety of input parameter distributions. We now apply the model to two nuclear sites in the UK to demonstrate that, with well-defined input parameters, PRAME is suitable for modelling the likely water concentration distributions around these sites.

We shall compare the output water concentration distributions with values obtained, via published concentration factors (CFs), from environmental monitoring programmes operated by the Environment Agency (EA) and the Food Standards Agency (FSA). CFs for marine biota are listed in Technical Report 422 (*IAEA, 2004*) and concentrations recorded in the marine environment can be found in the RIFE reports (*EA et al., 2006*). These samples are typically taken from a number of locations in the vicinity of the nuclear installation's outfall, and PRAME models this variation in distance by sampling the 'distance to critical group' parameter uniformly over two (95th percentile) tidal excursions. Where sampled concentrations are reported as being '< *x*' we conservatively assume the concentration to be '*x*'.

Hartlepool power station

Hartlepool nuclear power station is situated on the north-east coast of England, two miles south of the town of Hartlepool. The station discharges liquid radioactive waste into the North Sea via an underwater pipeline extending 200 metres into Tees Bay. The principal radiological components of this waste are ^{3}H, ^{35}S and ^{60}Co; an authorisation also exists for 'other radionuclides' to be discharged in this manner.

Monitoring data for the marine environment in the vicinity of the power station consist of the concentrations of various nuclides in marine samples of fish, crustacea, mollusca, sediment and seaweed. There is no data available concerning concentrations of ^{35}S in the marine environment, so we neglect this nuclide from our model run. We therefore concentrate on modelling the water concentration of ^{3}H and ^{60}Co, as well as ^{137}Cs, which we assume to make up the 'other radionuclides' discharged. This assumption is due to the availability of monitoring data for ^{137}Cs and it being a consistent, and major, component of the 'other radionuclides' (*OSPAR, 2005*).

A PRAME model was run using input distributions (Table 10) based on Cefas' sediment data (*Jenkinson, 2006*), Admiralty charts, operator information and expert elicitation (*Aldridge, 2005*). The advection-diffusion mode was chosen and 200 model runs were undertaken, with the nuclide discharge rates input as the mean of the past five years actual discharges from Hartlepool nuclear power station. The modelled water concentration was then compared to an implied water concentration distribution derived from monitoring data recorded over the past five years.

Parameter		*Lower bound*	*Upper bound*	*Percentile*	*Distribution*
Suspended sediment load		0.8	11.58	95	Log-normal
Sediment ratio		2.8×10^{4}	4×10^{4}	99	Normal
Mean depth		5.9	14.1	95	Normal
Residual velocity		0.018	0.041	95	Log-normal
Diffusion coefficient		0.42	1.08	95	Normal
Half tidal excursion*		1.68×10^{3}	4.32×10^{3}	95	Normal
Discharge start time		0.54	0.54	100	Uniform
Discharge end time		0.92	0.92	100	Uniform
Initial spreading radius		50	50	100	Uniform
Distance to critical group		0	8.63×10^{3}	100	Uniform
K_d value	^{3}H	0.1	100	99	Log-normal
	^{60}Co	3×10^{4}	3×10^{6}	"	"
	^{137}Cs	400	4×10^{5}	"	"

Table 10 *PRAME input parameter distributions for initial Hartlepool assessment,* Hart_1. **Half tidal excursion refers to the half tidal excursion at both the critical group location and the outfall pipe.*

Results

To compare the two sets of data, we first trim the distributions to remove any anomalous outliers. This leaves the 5th and 95th percentiles as our lower and upper bounds for comparison. We can then see, from Table 11 and Figure 9, that the modelled distribution of both ^{3}H and ^{60}Co is somewhat higher than the implied water concentration, with the modelled mean lying above the range suggested by the monitoring data, although there is some overlap in both cases. The modelled distribution of ^{137}Cs, however, lies entirely within the implied range, and there is a good agreement between the mean values of both.

To address the disparity in the modelled and implied distributions of ^{3}H and ^{60}Co, a second PRAME model was run. The diffusion coefficient distribution was changed from the range recommended by Aldridge (*2005*) to a normal distribution centred on the value suggested by Jenkinson & Grzechnik (*2006*)[2]. The effect of this parameter change was to decrease the modelled concentration of all three nuclides (Figure 9); in the case of ^{3}H and ^{60}Co this brought the modelled and implied distributions into good agreement, with the modelled mean now lying within the implied range, and in the case of ^{137}Cs the modelled range remained wholly within the implied range, albeit the mean concentrations have diverged from one another. This seemingly low modelled ^{137}Cs distribution, combined with the fact that we have overestimated its discharge rate by considering it to constitute the entire non-specific discharge, could be construed as a poor result for the PRAME model. However, the legacy of weapons testing, the Chernobyl accident and high levels of ^{137}Cs discharge from Sellafield during the 1970s have led to elevated concentrations of this nuclide in the North Sea. This would explain the relatively high mean and upper bound of the implied water concentration, and renders a quantitative comparison difficult.

		Water concentration (Bq/l)		
		5th percentile	*Mean*	*95th percentile*
^{3}H	*Hart_1*	36.6	73.9	122
	Hart_2	14.2	28.9	46.7
	Implied	15	30	68
^{60}Co	*Hart_1*	3.5×10^{-5}	1.7×10^{-4}	3.64×10^{-4}
	Hart_2	1.4×10^{-5}	6.67×10^{-5}	1.41×10^{-4}
	Implied	1.75×10^{-6}	2.45×10^{-5}	1.03×10^{-4}
^{137}Cs	*Hart_1*	1.07×10^{-3}	2.27×10^{-3}	3.94×10^{-3}
	Hart_2	4.19×10^{-4}	8.88×10^{-4}	1.57×10^{-3}
	Implied	1.43×10^{-4}	2.39×10^{-3}	5.89×10^{-3}

Table 11 *Modelled and implied water concentrations obtained from environmental monitoring data around the Hartlepool nuclear power station. Input parameter distributions for model Hart_1 are shown in Table 10; model Hart_2 uses the same except for an increased diffusion coefficient (see footnote 2)*

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[2] The value recommended for use with the WAT model at this site is $5 m^2 s^{-1}$; here we use a normal distribution with 5th/95th percentile values of 3/7.

Although further modification of the input parameters is clearly possible, we consider the *Hart_2* model to be sufficient to validate PRAME in this instance. Table 14 shows the improvement between the two PRAME models in terms of the agreement of modelled concentration ranges with their implied analogues, with a remarkable level of agreement, given the inherent uncertainties in input parameters, being attained by the *Hart_2* model run.

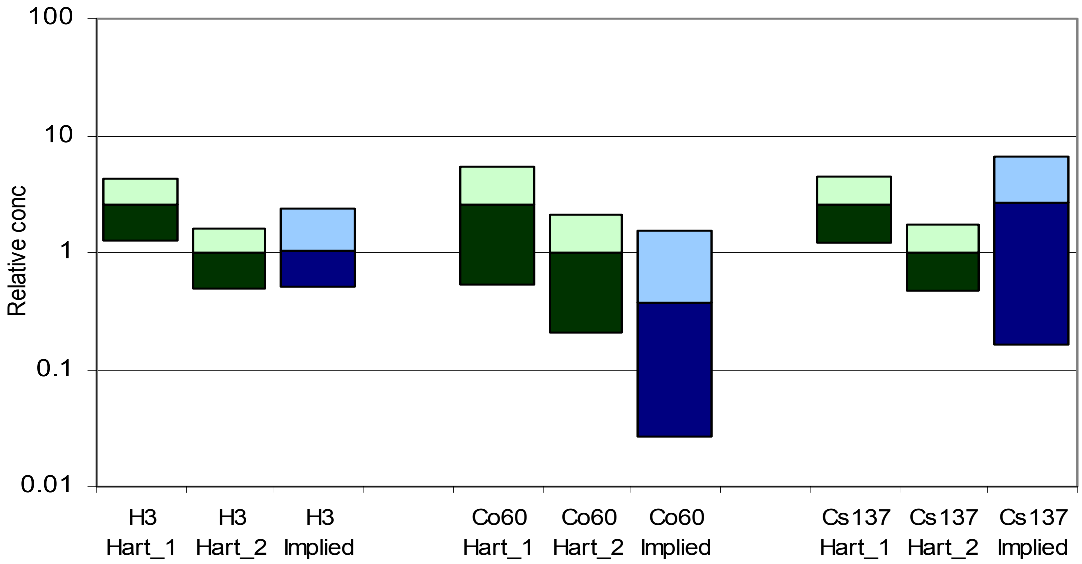

Figure 9 *Comparison of the 5th – 95th percentile ranges from two PRAME models and environmental monitoring data at Hartlepool nuclear power station. Vertical axis is scaled logarithmically, and each range has been scaled according to the mean concentration of the respective nuclide in the* Hart_2 *model run. Concentrations are thus relative to this mean value – absolute concentrations are listed in Table 11. Lower portion of each bar represents the 5th%ile – mean, upper portion represents mean – 95th%ile.* Hart_2 *model range is clearly a better fit to the implied range for ^{3}H and ^{60}Co.*

Heysham power station

Heysham nuclear power station, situated in Morecambe Bay in the north-west of England, consists of two advanced gas-cooled reactors. Both Heysham 1 and Heysham 2 are authorised to discharge liquid radioactive waste via a pipeline extending 500 metres into Morecambe Bay. Similarly to Hartlepool power station, the authorisation covers ^{3}H, ^{35}S and ^{60}Co as well as non-nuclide specific discharges.

We shall again concentrate on modelling ^{3}H, ^{60}Co and ^{137}Cs, as there are no monitoring results for ^{35}S, and ^{137}Cs is a major constituent of the 'other radionuclides' discharged (*OSPAR, 2005*). The PRAME input parameter distributions used in the initial Heysham assessment are shown in Table 12; 200 runs of the advection-diffusion model were undertaken with nuclide discharge rates input as an average of the past five years actual discharges. The 'other radionuclides' discharged were assumed, as before, to be ^{137}Cs. The modelled output was compared to a distribution of water concentrations formed, using standard CFs, from environmental monitoring data recorded over the past five years. We refer to this as the implied concentration.

Parameter	*Lower bound*	*Upper bound*	*Percentile*	*Distribution*
Suspended sediment load	1.33	45.69	95	Log-normal
Sediment ratio	6×10^4	7.5×10^4	99	Normal
Mean depth	5.9	14.1	95	Normal
Residual velocity	0.025	0.08	95	Log-normal
Diffusion coefficient	0.67	1.33	95	Normal
Half tidal excursion*	8.24×10^3	1.975×10^4	95	Normal
Discharge start time	0	0	100	Uniform
Discharge end time	1	1	100	Uniform
Initial spreading radius	50	50	100	Uniform
Distance to critical group	0	3.95×10^4	100	Uniform
K_d value ^{3}H	0.1	100	99	Log-normal
K_d value ^{60}Co	3×10^4	3×10^6	"	"
K_d value ^{137}Cs	400	4×10^5	"	"

***Table 12** PRAME input distribution parameters for initial Heysham assessment, Heys_1. *Half tidal excursion refers to the half tidal excursion at both the critical group location and the outfall pipe.*

Results

Both the modelled and implied water concentration distributions were trimmed at the 5th and 95th percentiles to remove outliers, and the resultant ranges compared in Table 13 and Figure 10. There is significant agreement between the modelled and implied ranges of ^{3}H and ^{60}Co, indicating that the PRAME model is well parameterised in this instance. Although the implied mean concentration of ^{60}Co is slightly higher than the 95th percentile of the modelled distribution, a number of the monitoring data are presented as 'less than' results, which indicates that the actual mean concentration will be lower than this and quite possibly within the range of modelled results.

		Water concentration (Bq/l)		
		5th percentile	*Mean*	*95th percentile*
^{3}**H**	*Heys_1*	24.1	47.7	80.1
	Implied	25	46.8	72.6
^{60}**Co**	*Heys_1*	1.99×10^{-6}	2.14×10^{-5}	6.02×10^{-5}
	Implied	2.5×10^{-6}	6.42×10^{-5}	1.72×10^{-4}
^{137}**Cs**	*Heys_1*	7.24×10^{-4}	2.61×10^{-3}	4.94×10^{-3}
	Implied	6.43×10^{-3}	6.11×10^{-2}	0.14

***Table 13** Modelled and implied water concentrations obtained from environmental monitoring data around the Heysham nuclear power station. Input parameter distributions for model Heys_1 are shown in Table 12*

The marked disparity in the range of ^{137}Cs concentration, especially when allowing for the overestimation of discharge rate in the PRAME model, can again be explained as a legacy of weapons testing, the Chernobyl accident and, particularly given its proximity, the high levels of ^{137}Cs discharged from Sellafield in the 1970s. The modelled concentration being somewhat lower than the implied concentration strongly aids this interpretation, as does a remark in a recent RIFE report (*EA et al., 2006*) that 'concentrations of ^{137}Cs in sediments were … largely due to Sellafield'.

We consider *Heys_1* output to be sufficiently well matched to the implied concentration ranges in the cases of ^{3}H and ^{60}Co (see Table 14) as to validate the PRAME model in this instance, and render a refined model unnecessary.

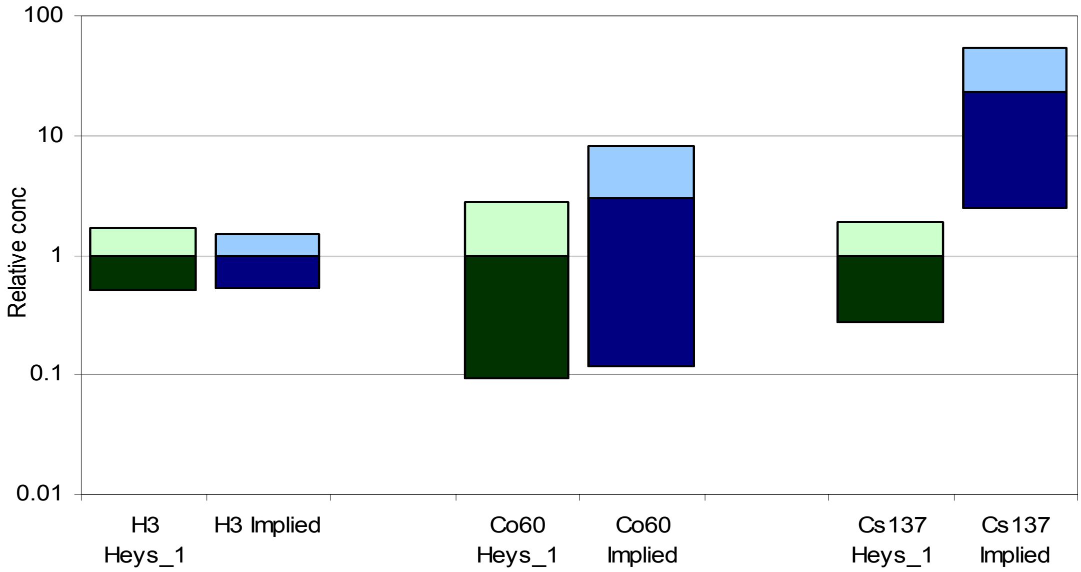

Figure 10 *Comparison of the 5^{th} – 95^{th} percentile ranges from two PRAME models and environmental monitoring data at Heysham nuclear power station. Vertical axis is scaled logarithmically, and each range has been scaled according to the mean concentration of the respective nuclide in the Heys_1 model run. Concentrations are thus relative to this mean value – absolute concentrations are listed in Table 13. Lower portion of each bar represents the 5^{th}%ile – mean, upper portion represents mean – 95^{th}%ile. Modelled ranges of ^{3}H and ^{60}Co show good agreement with concentration ranges implied from monitoring data, discrepancy in ^{137}Cs ranges explained by legacy of Sellafield discharges and other factors.*

		Agreement of modelled and implied concentration ranges (%)	***Factor of uncertainty***
Hart_1	^{3}H	49.4	3.34
	^{60}Co	23.88	10.4
	^{137}Cs	100	3.69
Hart_2	^{3}H	98.3	3.29
	^{60}Co	88.3	10.1
	^{137}Cs	100	3.75
Heys_1	^{3}H	93.3	3.32
	^{60}Co	96.6	30.25
	^{137}Cs	0	6.82
Implied (Hartlepool)	^{3}H	-	4.53
	^{60}Co	-	58.9
	^{137}Cs	-	41.2
Implied (Heysham)	^{3}H	-	2.90
	^{60}Co	-	68.8
	^{137}Cs	-	21.77

Table 14 *Level of agreement between the modelled water concentrations and their respective implied concentrations, expressed as the percentage of the modelled 5^{th} – 95^{th} percentile range lying inside the 5^{th} – 95^{th} percentile range of the implied concentration. The factor of uncertainty of each concentration range is also listed, showing in particular that the improvement in the level of agreement from* Hart_1 *to* Hart_2 *was not gained by simply increasing the uncertainty of output, but through the better choice of input parameters.*

Concluding remarks

We have presented the results of a series of test and validation exercises undertaken to demonstrate the ability of Cefas' probabilistic PRAME model to effectively determine the concentration of radionuclides in the marine environment due to regular discharges. The effect of varying input parameter distributions was investigated, and the positive consequence of better information regarding these inputs was observed in tighter, more focussed output distributions.

We compared output distributions using their factor of uncertainty, which is calculated as the ratio of 95th to 5th percentiles of the distribution. This gives a measure of the range of output, whilst filtering extreme values that could be classed as outliers. Differences in the behaviour of conservative and non-conservative nuclides were thus observed, and a further investigation of the sensitivity of output to changes in the K_d value identified a transitional region where nuclide concentrations would drop rapidly from one stable value to another as K_d increased. The precise location of this region is strongly dependent on the suspended sediment load, as we would anticipate.

The PRAME model was then employed to attempt to recreate the concentration distributions of various nuclides in the vicinity of two nuclear power stations. Model output was compared to an implied water concentration distribution derived from marine samples and CFs. The range of this implied concentration demonstrated the benefit of the probabilistic output over its deterministic equivalent, as there is often no single concentration dominant over a region but rather a dynamic range that varies in both time and space.

Hartlepool power station was the subject of our first validation exercise (*Hart_1*). Input parameter distributions from a variety of sources were used, but the output water concentrations of ^{3}H and ^{60}Co were generally in poor agreement with the implied distribution and range. A second model was run (*Hart_2*), in which the input diffusion coefficient distribution (a particularly uncertain and variable parameter, and one often used to 'tune' such models) was increased to reflect the recommendation of Jenkinson & Grzechnik (*2006*). The output distributions of ^{3}H and ^{60}Co from *Hart_2* were in far better agreement (Table 14) with their respective implied distributions. Although the modelled distribution of ^{137}Cs from *Hart_2* was below the mean value of the implied distribution, the 'background' levels of ^{137}Cs present in the North Sea mean that such underestimation is to be expected.

Our second validation exercise looked at nuclide concentrations around Heysham power station. The PRAME model, *Heys_1*, gave output distributions of ^{3}H and ^{60}Co that coincided significantly with the implied concentration distributions of these nuclides in the region. The modelled and implied ranges of ^{137}Cs did not overlap at all, with the implied concentrations being somewhat higher than those predicted by the PRAME model. This, however, can be firmly linked to the proximity of the reprocessing plant at Sellafield, and in particular the high levels of ^{137}Cs discharged there in the

1970s. Concentrations of ^{137}Cs in seawater in the Morecambe Bay area are estimated to be between 0.1 and 0.2 Bq/l (*EA et al., 2006*), principally due to this Sellafield legacy. This is in good agreement with our implied concentration distribution, which had a 95th percentile value of 0.14 Bq/l and a maximum value of 0.16 Bq/l.

The success of these validation exercises clearly shows the PRAME model is suitable for its intended use. The range of implied water concentrations derived via CFs also highlights the benefit of the probabilistic approach in providing an output distribution. It is worth noting that the factor of uncertainty of the implied distributions was greater than that of the modelled equivalent in all but one case, despite the PRAME input distributions being highly variable.

Further testing and validation of the dosimetric module of PRAME is desirable, but as there are no data concerning the dose equivalent received by members of the public it would be difficult to extend the method of validation presented here to this part of the model. A first step towards this might be to permit some variation of the CFs used in the model. These are currently fixed at the IAEA recommended values for each phylum (fish, crustacea, mollusca etc.) (*IAEA, 2004*), as were used in forming the implied water concentration distributions. However, it is widely held that there is some intra-phylum variation. Modifying PRAME to sample a distribution of the CF for each phylum, and generating an intermediate output distribution combining water concentration and CF, would allow us to compare the concentration of nuclides in marine foodstuffs being input to the dosimetric module with monitoring data. This would appear to be a better means of comparison than the implied water concentration method, as it preserves the actual monitoring data rather than regressing to the water concentration that brought it about, and incorporates a degree of uncertainty and variability into a parameter that is currently (artificially) fixed.

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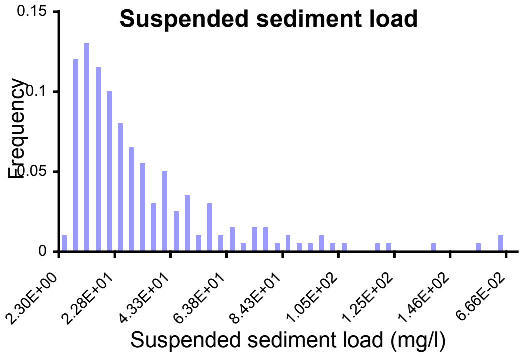
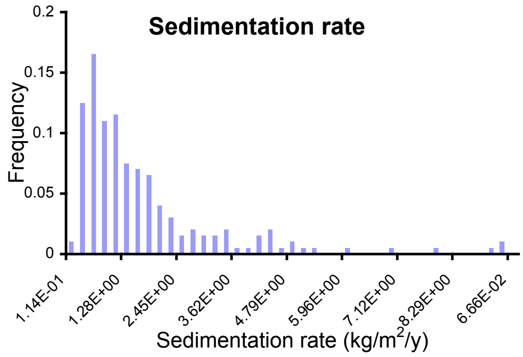
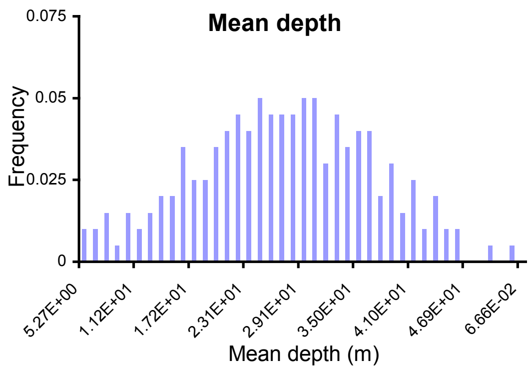
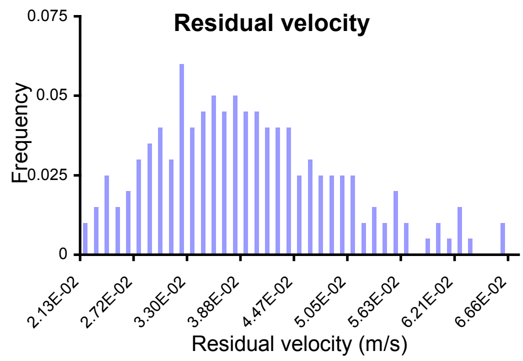
Appendix A – Control run input parameter distributions

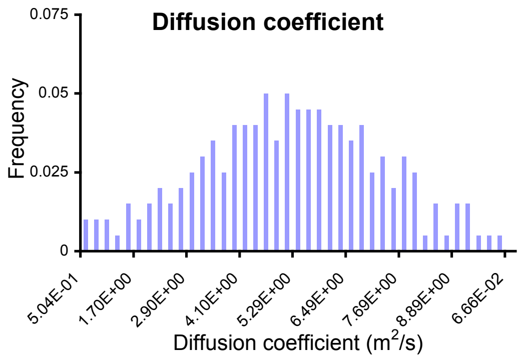
We now present the input parameter distributions used in the control run. Similar charts can be automatically generated by the PRAME application, which places them in an Excel worksheet alongside the raw sampling data and some statistics relevant to that distribution.

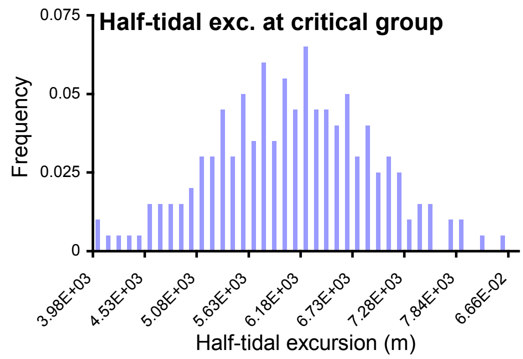


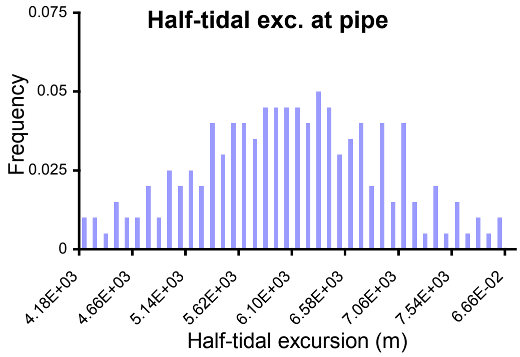


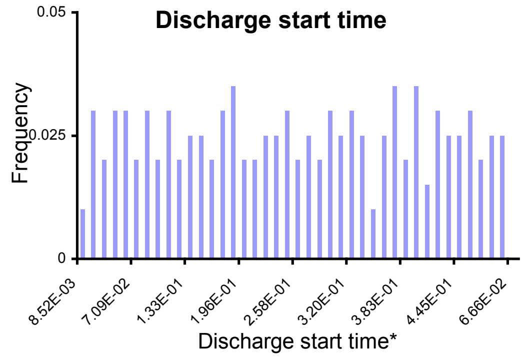


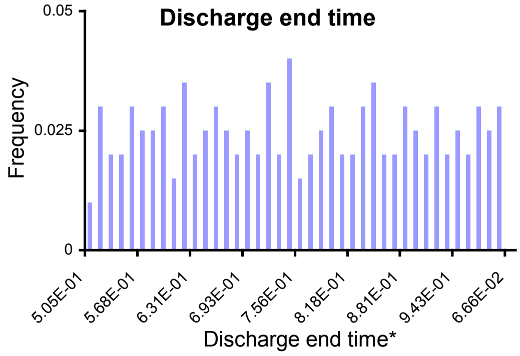


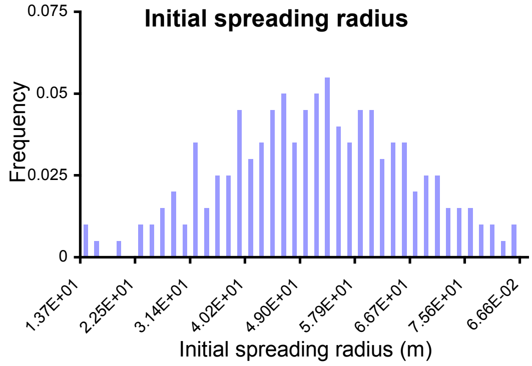


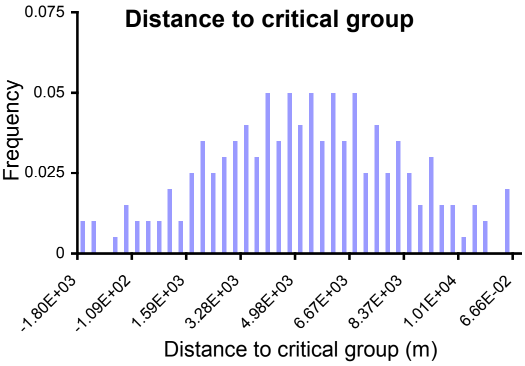


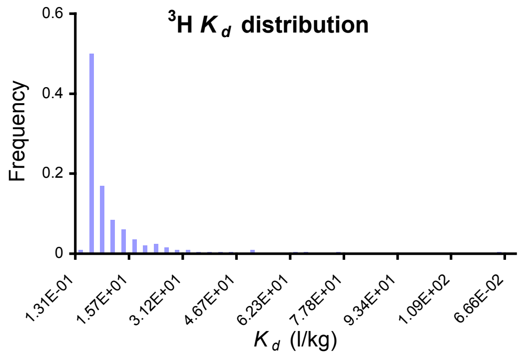


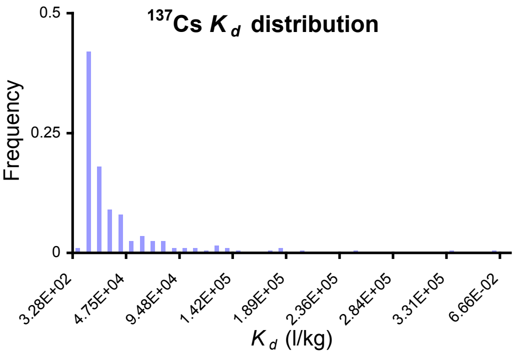


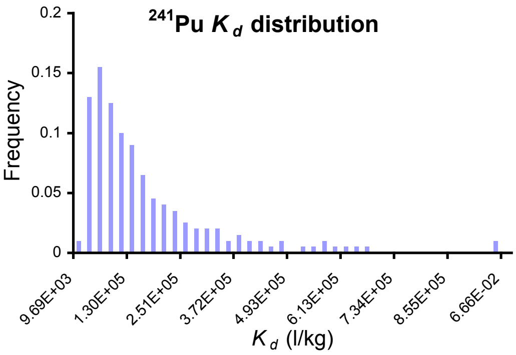


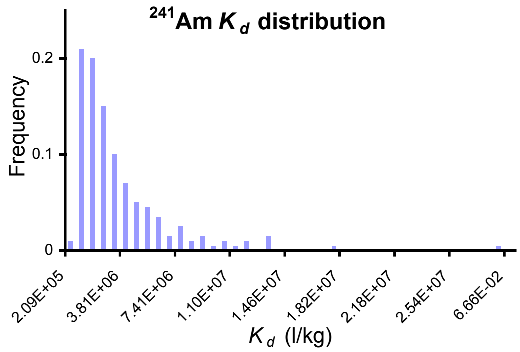












*Discharge start/end times are input as fractions of a tidal cycle, where 0/1 represent low water and 0.5 represents high water

Appendix B – Comparison of probabilistic and deterministic model output

The output range of water concentration distribution from the control run and tests 2 – 5 was compared to the deterministic output of the WAT model (*Round, 1998a*). The WAT input parameter values (Table B1) were taken as the mean of the probabilistic input distributions, and the output results are displayed in Figures B1 – B4 alongside the 5^{th} and 95^{th} percentiles of the PRAME output distributions.

Each of the WAT output concentrations falls comfortably inside the 5^{th} – 95^{th} percentile range of the respective PRAME output distribution. This agreement would seem to indicate that the probabilistic input method used in the PRAME model is well conditioned, and does not, on the whole, tend toward extreme values of water concentration.

Parameter	*Control*	*Test 2*	*Test 3*	*Test 4*	*Test 5*
Suspended sediment load	31.3	31.3	20	20	31.3
Sedimentation rate	1.6	1.6	1	1	1.6
Mean depth	26.6	17.5	26.6	17.5	17.5
Residual velocity	0.039	0.039	0.039	0.039	0.039
Diffusion coefficient	5.05	5.05	5.05	5.05	7.5
Half tidal excursion*	6000	6000	6000	6000	6000
Discharge start time	0.25	0.25	0.25	0.25	0.25
Discharge end time	0.75	0.75	0.75	0.75	0.75
Initial spreading radius	50	50	50	50	50
Distance to critical group	4980	7000	4980	7000	7000

Table B1 *WAT input parameters used to compare deterministic output to PRAME output from control run and tests 2 – 5. These were chosen to be the mean values of the respective PRAME input parameter distribution. Note that WAT requires a sedimentation rate rather than the sediment ratio. *Half tidal excursion refers to the half tidal excursion at both the critical group location and the outfall pipe.*

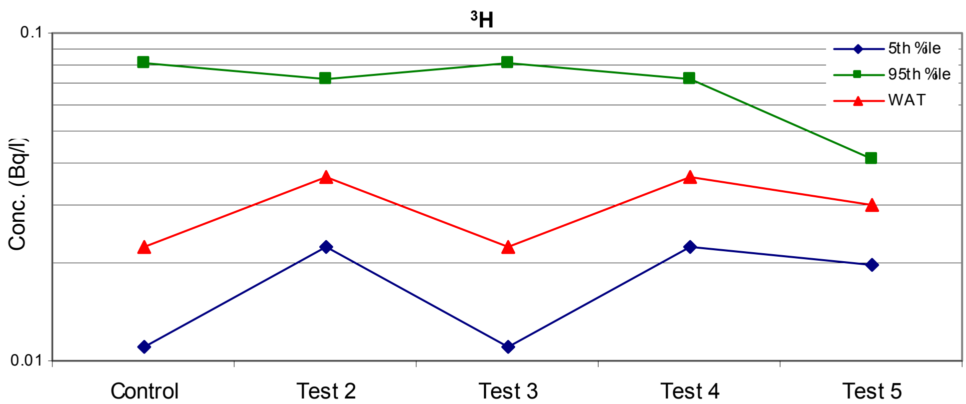


Figure B1 *Comparison of the WAT output concentration of ^{3}H with the 5^{th} and 95^{th} percentiles from the relevant PRAME output distributions. Vertical scale is logarithmic.*

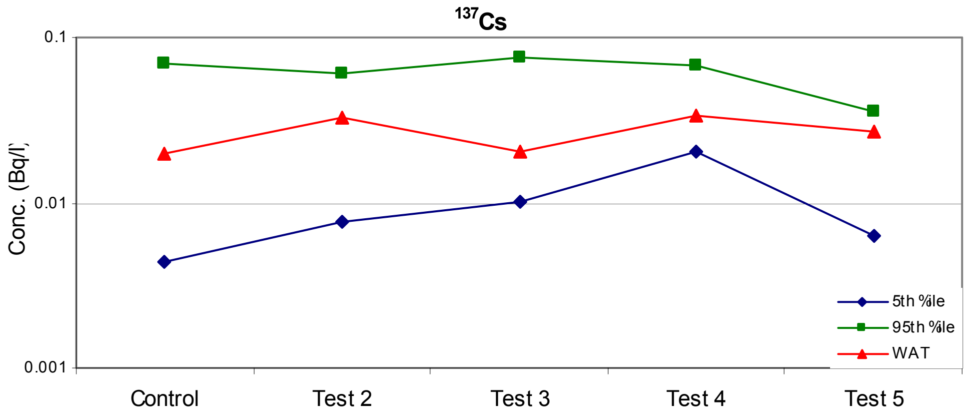


Figure B2 *Comparison of the WAT output concentration of ^{137}Cs with the 5th and 95th percentiles from the relevant PRAME output distributions. Vertical scale is logarithmic.*

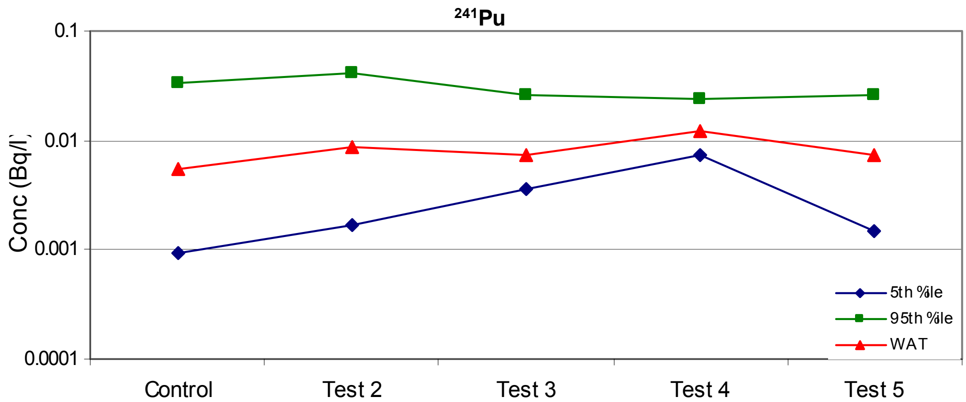


Figure B3 *Comparison of the WAT output concentration of ^{241}Pu with the 5th and 95th percentiles from the relevant PRAME output distributions. Vertical scale is logarithmic.*

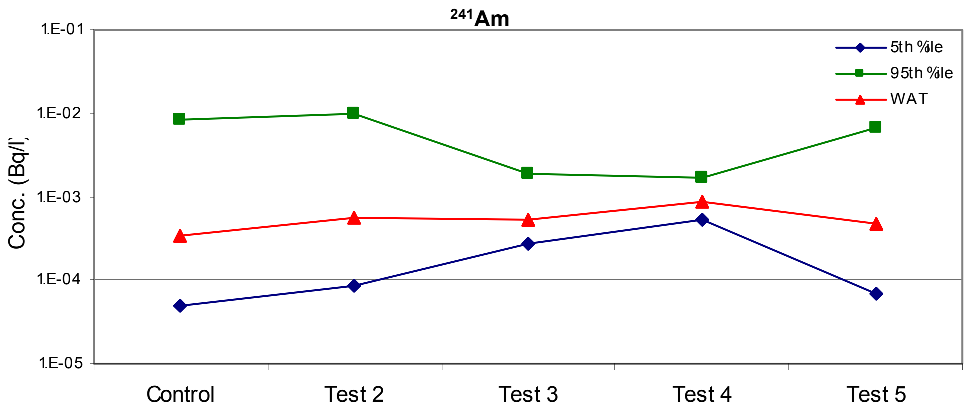


Figure B4 *Comparison of the WAT output concentration of ^{241}Am with the 5th and 95th percentiles from the relevant PRAME output distributions. Vertical scale is logarithmic.*

About us

Cefas is a multi-disciplinary scientific research and consultancy centre providing a comprehensive range of services in fisheries management, environmental monitoring and assessment, and aquaculture to a large number of clients worldwide.

We have more than 500 staff based in 3 laboratories, our own ocean-going research vessel, and over 100 years of fisheries experience.

We have a long and successful track record in delivering high-quality services to clients in a confidential and impartial manner.
(www.cefas.co.uk)

Cefas Technology Limited (CTL) is a wholly owned subsidiary of Cefas specialising in the application of Cefas technology to specific customer needs in a cost-effective and focussed manner.

CTL systems and services are developed by teams that are experienced in fisheries, environmental management and aquaculture, and in working closely with clients to ensure that their needs are fully met.
(www.cefastechnology.co.uk)

Customer focus

With our unique facilities and our breadth of expertise in environmental and fisheries management, we can rapidly put together a multi-disciplinary team of experienced specialists, fully supported by our comprehensive in-house resources.

Our existing customers are drawn from a broad spectrum with wide ranging interests. Clients include:

- international and UK government departments
- the European Commission
- the World Bank
- Food and Agriculture Organisation of the United Nations (FAO)
- oil, water, chemical, pharmaceutical, agro-chemical, aggregate and marine industries
- non-governmental and environmental organisations
- regulators and enforcement agencies
- local authorities and other public bodies

We also work successfully in partnership with other organisations, operate in international consortia and have several joint ventures commercialising our intellectual property.

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