

Modelling the Dispersion of Radionuclides following Short Duration Releases to Rivers

R&D Technical Report P3-074/TR

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ISBN 1 844320 723

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This report develops a model suitable for assessing the dispersion of radionuclides following short duration releases to stretches of the River Thames. Concentrations and time integrated concentrations of radionuclides in the river environment downstream of the release point are presented.

Keywords

Radioactivity, river, model.

Research Contractor

This document was produced under R&D Project P3-074 by :

CEH Dorset, Winfrith Technology Centre, Winfrith Newburgh, DORCHESTER Dorset
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The Environment Agency's Project Manager for R&D Project P3-074 was:
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ACKNOWLEDGEMENTS

We would like to thank Dr Dave Timms of Portsmouth University for assistance in the field experiments.

EXECUTIVE SUMMARY

This project develops a model for assessing short duration liquid discharges of radionuclides to rivers. The assessment of doses arising from discharges to rivers is normally carried out by considering annual average discharge rates. Actual authorised discharges, however, may occur unevenly during the year or relatively high short-term discharges could occur in the unlikely event of an incident. Short term radionuclide releases could potentially result in temporary increases in radionuclide activity concentrations in water and fish which are greater than those resulting from a continuous discharge. The purpose of this project is to develop a model to assess short term releases from these sites, and where possible develop generic methods of assessing short term releases.

An advection-dispersion model was developed to predict the concentrations of radionuclides in the river environment, ie in river water, river bed sediment and in predatory fish. Uptake of radionuclides to fish was modelled by estimating rates of uptake of radionuclides via the aquatic food chain or across the gill, as appropriate. The model was used to predict the concentrations of the radionuclides in the river Thames and its tributaries as a result of short duration discharges into stretches of the Thames and River Colne. Model output is given as a series of graphs of activity concentration and time integrated activity concentration resulting from a 1 MBq discharge for the following release durations: 5 minutes, 1 h, 3 h, 12 h and 24 h. The five locations for which predictions are given were 100 m, 300 m, 1000 m, 3000 m and 10000 m downstream.

The **river volumetric flow rate** was shown to be the most important environmental variable determining activity concentrations in water, fish and sediments following a release. In general, the maximum and integrated activity concentrations in water and fish will be in inverse proportion to the river volumetric flow rate, for a given amount and duration of release.

The **duration of release** significantly influences peak radioactivity concentrations in water, $C_w(\text{max})$, particularly near to the point of discharge, where $C_w(\text{max})$ is in inverse proportion to the release duration. Duration of release does not influence the maximum activity concentrations in fish and sediments, nor does it influence the estimates of time integrated activity concentrations in water, fish and sediments.

The **fraction of radioactivity sorbed to suspended particulates**, f_p , can significantly affect radioactivity concentrations in water (dissolved phase), fish, and bed sediments. For the radionuclides studied, however, best estimates of f_p imply that the vast majority of radioactivity is in the dissolved phase ($f_p < 0.1$) for short duration releases (Table 8).

The **water temperature** significantly influences the maximum activity concentration in fish, $C_f(\text{max})$ because fish feeding rates are much lower at lower water temperatures. At a water temperature of 17°C, $C_f(\text{max})$ is predicted to be approximately five times higher than at a water temperature of 7°C for all radionuclides except radiostrontium and tritium. The water temperature has less effect on time integrated activity concentrations in fish, there being less than a factor of three decline in ΣC_f as temperature changes from 17°C to 7°C.

Generic methods of predicting pollutant concentrations in rivers were developed and recommendations made for their parameterisation and implementation.

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1. INTRODUCTION

This project develops a model for assessing short duration liquid discharges from three licensed nuclear sites: AWE Aldermaston, UKAEA Harwell, and Amersham plc. All sites currently have authorisations from the Environment Agency (EA) to discharge radioactivity to the River Thames or its tributaries. The assessment of doses arising from these discharges is normally carried out by considering annual average discharge rates. Actual authorised discharges, however, may occur unevenly during the year or relatively high short-term discharges might occur in the unlikely event of an incident. Short term radionuclide releases could potentially result in temporary increases in radionuclide activity concentrations in water and fish which are greater than those resulting from a continuous discharge. The purpose of this project is to develop a model to assess short term releases from these sites, and where possible develop generic methods of assessing short term releases.

Current dose assessment models are chiefly designed for relatively long term continuous releases of radionuclides. They therefore assume that radionuclides are released at a continuous average rate over a long period of time (one year, for example). However, for short-term releases, the rate at which a radionuclide is released to a river can have a significant influence on concentrations of the radionuclide downstream of the discharge site. The present study uses recently developed models together with extensive empirical data on pollutant dispersal in rivers to assess this influence. Most dose assessment models also assume equilibrium values of some of the key parameters such as the water-sediment distribution coefficient and the fish-water bioaccumulation factor. The assumption of equilibrium may not be valid for short duration releases because it is unlikely that there would be sufficient time for equilibrium to become established. In the present study dynamic models (largely developed following the Chernobyl accident) for transfers of radionuclides to bed sediments and the dynamics of radionuclide accumulation in fish will be used to assess short term releases.

The overall objectives of the project are therefore:

- To develop a model for assessing the impact of short duration releases from three licensed nuclear sites discharging to the Thames;
- To develop, where possible, generic methods of assessing short-term releases of radioactivity to rivers, for use at other sites and other rivers.

2. RELEASE SCENARIOS

Of the three sites studied, two (AWE Aldermaston, UKAEA Harwell) discharge via pipelines into the Thames. AWE Aldermaston discharges via a 20 km pipeline into the Thames below Pangbourne. UKAEA Harwell discharges via a pipeline into the Mill Stream branch of the Thames at Sutton Courtney (the Mill Stream rejoins the main river channel approximately 100m below the discharge point). Amersham/Nicomed discharges via Maple Lodge Sewage Treatment Works (STW) into the Grand Union Canal. All three sites currently have authorisations from the Environment Agency to discharge into the Thames. Table 1 summarises the radionuclides which are authorised to be released from the different sites, and which will be considered in this project.

The releases from each of the three sites will be briefly summarised here. Further details of the sites and their discharges can be found in Appendix A.

Table 1 List of radionuclides considered and their half-lives

Radionuclide	Half-life	Site potentially discharging
H-3	12.3 yrs	Amersham, Harwell, Aldermaston
C-14	5730 yrs	Amersham
P-32	14.3 days	Amersham
Co-60	5.27 yrs	Amersham, Harwell
Zn-65	244.3 days	Amersham, Harwell
Sr-89	53 days	Amersham, Harwell
Sr-90	28.8 yrs	Amersham, Harwell
I-125	59.4 days	Amersham, Harwell
I-131	8.05 days	Amersham, Harwell
Cs-134	2.065 yrs	Amersham, Harwell
Cs-137	30.2 yrs	Amersham, Harwell
Pu-238	87.7 yrs	Amersham, Harwell, Aldermaston
Pu-239	2.4×10^4 yrs	Amersham, Harwell, Aldermaston
Pu-240	6.5×10^3 yrs	Amersham, Harwell, Aldermaston
Am-241	432.2 yrs	Amersham, Harwell, Aldermaston
U-234	2.45×10^5 yrs	Amersham, Harwell, Aldermaston
U-235	7.08×10^8 yrs	Amersham, Harwell, Aldermaston
U-238	4.47×10^9 yrs	Amersham, Harwell, Aldermaston

2.1 Releases from AWE Aldermaston

Radioactive liquid effluents generated on site are collected at the Radioactive Effluent Treatment Plant. Following treatment they are discharged to the Thames via the Aldermaston pipeline. The pipeline is 20 km long and discharges underwater directly to the Thames. The river forms a single channel at this point. Typically, two discharges per week are made, each of volume 150 m^3 . Because of the length of the pipeline (20 km), each discharge takes about 18 hours to pass through it giving an average volumetric flow rate of the discharge of approximately $2.2 \times 10^{-3} \text{ m}^3 \text{ s}^{-1}$, approximately 0.025% of the minimum volumetric flow rate of the river. Because of the length of the pipeline,

AWE Aldermaston consider it to be impossible that any release could be of duration less than 18 hours.

2.2 Releases from UKAEA Harwell

There are currently two liquid effluent discharge routes from Harwell, to the Thames at Sutton Courtney via a pipeline, and to Lydebank Brook (surface water runoff). Authorised discharges via surface water runoff (primarily tritiated water) to Lydebank Brook are most appropriately modelled as a continuous rather than a short-term release and therefore will not be considered in this project. Therefore, only the discharge to the Thames at Sutton Courtney will be considered in this project. Liquid effluents arising from the active area of the site are routed to the Liquid Effluent Treatment Plant (LETP). Liquid effluents from all other areas of the site (known as trade wastes) are routed to holding tanks for monitoring prior to discharge. If necessary the trade waste effluents in the holding tanks can be diverted to the LETP for treatment. Effluents from the LETP and trade wastes from the holding tanks are discharged via a pipeline to the River Thames at Sutton Courtenay. Two types of tank are emptied into the pipeline, of volume 500 m³ for low level effluent and of volume 1800 m³ for trade waste. The rate of tank emptying is approximately 0.083 – 0.125 m³ s⁻¹, and the travel time of effluent through the pipeline is approximately 35-40 minutes.

2.3 Releases from Amersham

There is currently one aqueous disposal route from Amersham, to the Grand Union Canal (GUC) via Maple Lodge sewage treatment works (STW). Active discharges are routed through final discharge tanks: 6 of volume 54.6 m³ and 4 of volume 27.3 m³. The tanks take approximately four hours to discharge.

Effluent from Amersham goes to the West Hyde pumping station (taking approximately 3 hours) and is then rapidly pumped to Maple Lodge STW. Mixing in the STW reduces peak concentrations (see Appendix A) and causes a delay in transfer of radioactivity to the river of approximately 8 hours, though this will be dependent on volumetric flow rate through the STW. Assuming a residence time of effluent in the STW of 8 hours, a delay of 3 hours in the transfer from Amersham to West Hyde, and a 5 hour delay in the STW, durations of releases from Amersham have been estimated (Table 2).

Table 2 Hypothetical release durations from Amersham and resulting release duration to the river

Duration of hypothetical release from Amersham	Estimated duration of discharge from Maple Lodge STW to GUC	Approximate time lag between release from Amersham and inflow to GUC
5 minute	8 hours	
30 minute	8 hours	
3 hours	8 hours	> 8 hours
12 hours	12 hours	
24 hours	24 hours	

2.4 Release Durations Considered in this Study

The release durations considered, and illustrative release scenarios are given in Table 3. To cover the range of scenarios listed in Table 3, release durations considered in this work were 5 minutes, 30 minutes, 3 hours, 12 hours and 24 hours.

Table 3 Release durations considered in this study with illustrative release scenarios

Tank volume m ³	Discharge rate m ³ /s	Release duration			Example
		Sec	Min	Hour	
<i>Known scenarios</i>					
500	0.125	4000	67	1.1	Harwell low level effluent tank emptying quickly
500	0.083	6000	100	1.7	Harwell low level effluent tank emptying slowly
1800	0.125	14400	240	4	Harwell trade effluent tank emptying quickly
1800	0.083	21700	361	6	Harwell trade effluent tank emptying quickly
55	N/A	28800	480	8	Amersham via sewage works#
27	N/A	28800	480	8	Amersham via sewage works#
150	0.0022	68000	1100	18	Aldermaston tanks via pipeline
<i>Indicative scenarios</i>					
50	0.1	500	8	-	Indicative
500	0.1	5000	80	1.3	Indicative
5000	0.1	50000	800	13	Indicative

Minimum duration of release after mixing as it passes through the sewage works

3. MODELLING THE DISPERSION

3.1 Model Target Variables

The objective of this modelling exercise is to make predictions, for the short term release scenarios, of the following target variables:

- Maximum activity concentration in water (total and dissolved phase);
- Time integrated (one year time integral) activity concentration in water (total and dissolved phase);
- Maximum activity concentration in predatory fish;
- Integrated (one year time integral) activity concentration in predatory fish;
- Maximum activity concentration in bed sediment;
- Integrated (one year time integral) activity concentration in bed sediment.

These end points were selected because water, predatory fish and sediments are the media most likely to result in radiation exposure of the public. We have not directly considered non-predatory fish because they are less likely to be eaten by the public. In addition, radionuclide concentrations in predatory species tend to be greater than in non predatory so the calculations made for predatory fish may be used as a maximum estimate for non-predatory species if required.

Calculations are made for all of the radionuclides considered (Table 1) for the different release times from each site 5 min, 30 min, 3 h, 12 h, 24 h(see Table 2). Calculations were made for sites at the following distances downstream of the discharge points: 100 m, 300 m, 1,000 m, 3,000 m, 10,000 m.

3.2 Environmental Behaviour of Releases

The dispersion of a short duration release of effluent into a river can be divided into two phases. The first phase is characterised by relatively quick dispersion down river of the part of the discharge that remains in solution. This part of the discharge (early phase) would be expected to clear the first 10 km stretch downstream of the discharge within 2 days (usually less than one day). The second phase of the dispersion is associated with the discharge that became attached to river bed sediment. Activity associated with bed sediment may remain in the 10 km stretch downstream of the discharge point for a considerable time (the late phase).

3.2.1 Dispersion in the early phase

Effluent released to a river disperses relatively rapidly in the vertical direction, but takes some time, and hence distance downstream, to disperse across the river width (transverse dispersion) (see Appendix B, section B3). As the effluent plume travels downstream, it also disperses along the length of the river (longitudinal dispersion) as a result of differing flow velocities in the channel. For the 10 km reaches studied here, the travel time of the effluent plume within the reach is of order one day or less. As the plume travels downstream, radionuclides may be deposited on the bed sediment as a result of sorption to suspended particles and their subsequent sedimentation. Following

passage of the plume, the activity concentration in bed sediments is expected to be greatest nearest the discharge point, the (average cross-section) activity concentration declining approximately exponentially downstream. Sedimentation of suspended particulates is expected to decrease as river flow velocity increases.

3.2.2 Dispersion in the late phase

Following the passage of the plume (the “early phase”), concentrations of radioactivity in water decline rapidly, but (much lower) activity concentrations in water may be maintained for long periods of time (the “late phase”) as a result of resuspension of radioactivity from bed sediments. Resuspension of radioactivity is most likely to occur during flood events when bed sediments can be transported long distances downstream, or out of the river into the estuary. On the other hand, Environment Agency river channel charts indicate that the studied river reaches are dredged, indicating that net accumulation of bed sediments occurs at least in parts of the studied reaches. Since it is not possible to be certain of bed sediment movement, bounding assumptions will be made which give conservative estimates of the target variables as follows:

- *Activity concentrations in water and fish.* When these are calculated, it will be assumed that there is no transfer of radionuclides to bed sediments. Under this assumption, the model would over estimate maximum and integrated activity concentrations for water and fish if in reality there was significant storage of radioactivity in bed sediments;
- *Activity concentrations in bed sediments.* It will be assumed that there is transfer of radioactivity to bed sediments, but that there is no subsequent movement of bed sediments. Under this assumption, the model would over-estimate maximum and integrated activity concentrations in bed sediments if in reality there was significant resuspension and movement of bed sediments.

3.3 Model Selection

3.3.1 Dispersion in the early phase

The choice of most suitable transport model is discussed in Appendix B. The plume transport (“early”) phase was modelled using a longitudinal advection-dispersion model, including transfers of radionuclides to bed sediments by settling of suspended particles. The model used a numerical solution to the advection-dispersion equation. Estimates were made of factors to correct for transverse mixing of the plume at short distances downstream of the release (see below and Appendix B, section B.3). Plume dispersion parameters for the advection-dispersion model were estimated using dye tracer experiments at the sites of interest (Appendix D).

3.3.2 Dispersion in the late phase

A separate model was developed for the period following the movement of the discharge plume out of the 10 km stretch (the late phase). For this period, movement and resuspension of bed sediment was modelled using a compartmental model which assumed that sediment moves from one river reach to the next via the overlying river

water. Thus the resuspension of bed sediment increases the activity concentration in the river water.

3.3.3 Uptake by fish

Transfers of radioactivity to fish were estimated using simple compartmental models. It is known that the bioaccumulation of radioactivity in fish is determined by numerous ecological and environmental factors such as the trophic level of the fish species, the fish behaviour, the water temperature and the water chemistry. Uptake may be via ingestion of contaminated food or direct transfers from the water via the gills. For most radionuclides the food chain is the primary uptake pathway, so a food uptake model was used to estimate uptake rates (Figure 1). Estimates of food ingestion rates as a function of water temperature were obtained from an empirical model for trout (Elliot 1975a,b) (Appendix C). Calculations were made for a trout of wet weight 500 g at 7, 12 and 17°C, representing typical water temperatures for winter, spring/autumn and summer respectively. Uptake and excretion rates were calculated for each of the radionuclides at each of the three water temperatures (Appendix D).

For strontium isotopes, which are primarily absorbed through the gills, a model for direct uptake via the water (Chowdhury and Blust 2001) was used to estimate the intake rate (Figure 2). Where the primary uptake pathway is uncertain, as in the case of Co for example, a food uptake model was used, but a conservative approach was taken which over-estimates the uptake rate to account for possible influence of the direct uptake (gill) pathway.

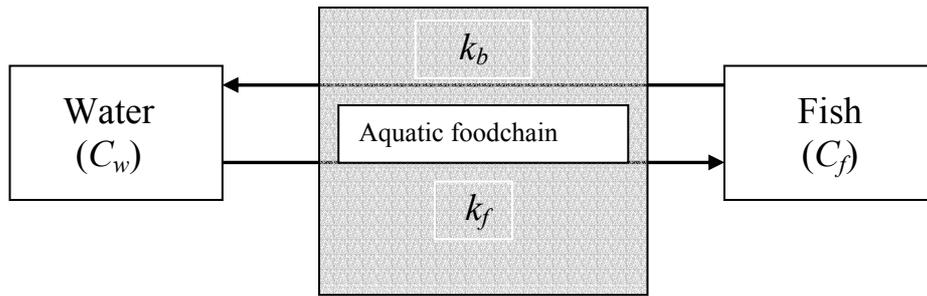


Figure 1 Illustration of model for uptake in fish via the food chain showing uptake, k_f , and excretion, k_b , rate constants

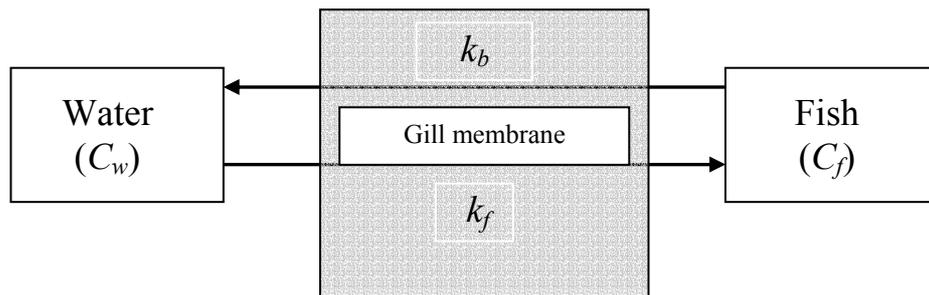


Figure 2 Illustration of model for uptake in fish via the gills. Mathematically the model is identical to the food chain model, but the uptake parameter is calculated using a Michaelis-Menten type model (Chowdhury and Blust 2001) for transfer of Sr across the gill membrane

3.3.4 Simplified model

To simplify and generalise the model output, the numerical advection-dispersion model was simplified using an analytic solution to the advection-dispersion equation and simplification of the sediment and fish uptake models (Appendix G).

3.4 River Characteristics

Three sections of river were modelled near to the discharge points for the three nuclear sites. The first stretch is the Pangbourne-Reading reach of the Thames. The channel and flow characteristics of the reach are given in Table 4. The river here is navigable and has been straightened, with a flat bed and steep sides and is believed to be dredged. The river channel is relatively uniform throughout the 10 km reach below the pipeline, and flow characteristics vary little. There is a weir and lock at Mapledurham.

The second stretch to be modelled is the Thames downstream of Sutton Courtney. The channel and flow characteristics of the reach are given in Table 5. At the start of the reach studied, the main Thames river is divided into three channels, a canal (the Culham Cut), Sutton Pools, and the Mill Stream to the south of Sutton Pools. The flow rate of the canal is negligible in comparison to that of the main river, and can be ignored. A

typical low flow rate of the mill stream is $0.44 \text{ m}^3 \text{ s}^{-1}$ (Stonell, 1999). This is approximately 7% of the total river flow under low flow conditions (Table 5). This figure, and estimates made from field studies of the Mill Stream flow rate carried out in this project, is significantly lower than average flow conditions under which “approximately one third of the average mean daily discharge [of the whole river] flows through the Mill Stream” (Stonell, 1999). After the confluence of the Mill Stream with the main river, the river is a navigable and relatively uniform channel. The river has been straightened, with a flat bed and steep sides and is believed to be dredged.

The third stretch modelled was the Grand Union Canal and the River Colne downstream of the Maple Lodge STW. At this point, the GUC and river Colne run parallel, with approximately 80% of the river flow going down the GUC. Flow and average channel characteristics of the GUC/Colne are summarised in Table 6. The discharge from Maple Lodge is into the GUC, with a storm overflow into the Colne. The GUC joins the river Colne approximately 1.3 km downstream of the Maple Lodge discharge, at which point the main flow is diverted to the Colne. The channel of the GUC is straight with the steep sides and flat bed of a canal. It is relatively deep and slow flowing and is very likely to be dredged. The Colne is wider, shallower and faster flowing with a gravel bed. The Colne enters the Thames north west of Staines, which is approximately 25 km downstream of Maple Lodge. The Thames is tidal up to Teddington Lock, which is approximately 25 km downstream of where the Colne enters the Thames.

The flow characteristics for each river reach are presented as mean and percentile flows for the year 2000. Figure 3 shows a comparison of flows at the three reaches for 7-9 years preceding this year. As shown in Figure 3, annual flows vary significantly, with the year 2000 being significantly above average.

From observations of the field dye-tracer experiments, and knowledge of the effluent discharge mechanism, an indication of transverse mixing of a plume is given in Table 7 for the three reaches studied here. In Table 7, the activity concentration in water (C_w , Bq l^{-1}) is given as a function of the total activity released in Becquerels (C_i) and the duration of release (T_i , seconds).

Table 4 Summary of river characteristics in the Pangbourne reach

Parameter	Value	Notes
Average cross sectional area of river	124.2 m ²	Mean of 5 sites within 10km downstream of each discharge. From EA depth charts.
Average width of river	59.7 m	
Mean annual volumetric flow rate	39.2 m ³ s ⁻¹	Logarithmic mean of EA measurements at Reading for year 2000.
High volumetric flow rate		
High volumetric flow rate	134 m ³ s ⁻¹	90 percentile flow from EA measurements at Reading for year 2000.
Estimated coefficient of dispersion at high flow	23 m ² s ⁻¹ (above weir)*	<i>Assume that D above weir is 10 times lower than that below.</i>
	230 m ² s ⁻¹ (below weir)	From Upper Thames data, $D = 0.0148Q^2 + 0.33Q$. Scale to Pangbourne-Reading using observed $D = 13.2 \text{ m}^2 \text{ s}^{-1}$, at $Q = 25.5 \text{ m}^3 \text{ s}^{-1}$ gives $D = 0.011Q^2 + 0.24Q$.
Estimated velocity at high flow	0.83 m s ⁻¹	Calculated from $v = Q/d.W$
Estimated depth at high flow	2.7 m	Equal to low flow value plus 0.6 m range in depth between high and low flows (P. Davidson, EA, pers. Comm.)
Medium volumetric flow rate		
Medium volumetric flow rate	39.2 m ³ s ⁻¹	Logarithmic mean of EA measurements at Reading for year 2000
Estimated coefficient of dispersion at medium flow	2.6 m ² s ⁻¹ (above weir)*	<i>Assume that D above weir is 10 times lower than that below.</i>
	26.3 m ² s ⁻¹ (below weir)	From Upper Thames data, $D = 0.0148Q^2 + 0.33Q$. Scale to Pangbourne-Reading using observed $D = 13.2 \text{ m}^2 \text{ s}^{-1}$, at $Q = 25.5 \text{ m}^3 \text{ s}^{-1}$ gives $D = 0.011Q^2 + 0.24Q$.
Estimated velocity at medium flow	0.29 m s ⁻¹	Calculated from $v = Q/d.W$
Estimated depth at medium flow	2.3 m	Estimated median between high and low flow values.
Low volumetric flow rate		
Low volumetric flow rate	9.9 m ³ s ⁻¹	10 percentile low flow from EA measurements at Reading for year 2000.
Estimated coefficient of dispersion at low flow	1.0 m ² s ⁻¹ (above weir)*	<i>From low flow dye tracer experiment.</i>
	2.4 m ² s ⁻¹ (below weir)	From Upper Thames data, $D = 0.0148Q^2 + 0.33Q$. Scale to Pangbourne-Reading using observed $D = 13.2 \text{ m}^2 \text{ s}^{-1}$, at $Q = 25.5 \text{ m}^3 \text{ s}^{-1}$ gives $D = 0.011Q^2 + 0.24Q$.
Estimated velocity at low flow	0.08 m s ⁻¹	Calculated from $v = Q/d.W$
Estimated depth at low flow	2.1 m	Calculated from EA depth charts and field measurements.

* *From the dye tracer experiments it was found that the coefficient of dispersion measured at sites above the Mapledurham weir (ca. 2.5 km from the top of the studied reach) was much lower (approximately one order of magnitude) than that below. This was probably due to (a) the uniformity and straightness of the river above the weir; (b) mixing of the tracer in the weir pool and (c) incomplete cross-sectional mixing in the first 500 m downstream of the discharge. Therefore different dispersion coefficients were used for sites above and below the weir.*

Table 5 Summary of river characteristics in the Sutton Courtney reach

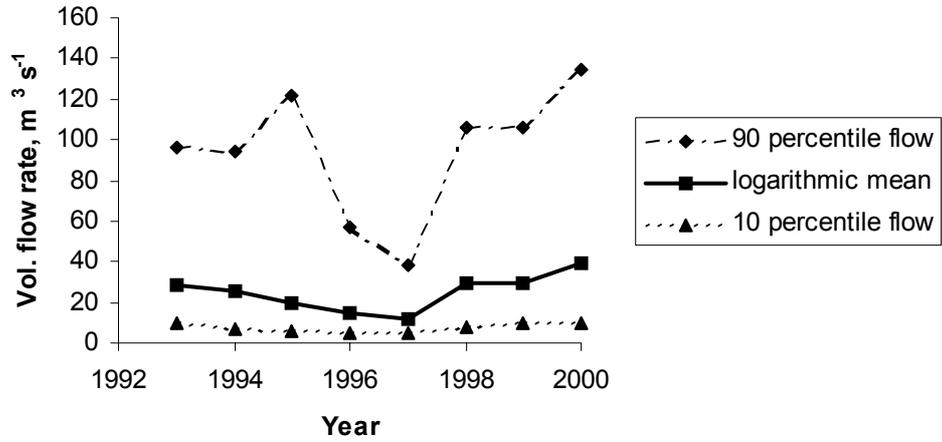
Parameter	Value	Notes
Average cross sectional area of river	59.8 m ²	Mean of 3 sites within 10km downstream of the discharge. From EA depth charts.
Average width of river	46.0 m	
Mean annual volumetric flow rate	27.9 m ³ s ⁻¹	Logarithmic mean from EA measurements at Sutton Courtney for year 2000
High volumetric flow rate		
High volumetric flow rate	101 m ³ s ⁻¹	90 percentile flow from EA measurements at Sutton Courtney for year 2000.
Estimated coefficient of dispersion at high flow	184 m ² s ⁻¹	From EA Upper Thames dye tracer data, $D = 0.0148Q^2 + 0.33Q$.
Estimated velocity at high flow	0.71 m s ⁻¹	From Upper Thames dye tracer data, $v = 0.028Q^{0.7}$
Estimated depth at high flow	3.1 m	Calculated from $d = Q/v.W$
Medium volumetric flow rate		
Medium volumetric flow rate	27.9 m ³ s ⁻¹	Logarithmic mean from EA measurements at Sutton Courtney for year 2000
Estimated coefficient of dispersion at medium flow	20.7 m ² s ⁻¹	From Upper Thames dye tracer data, $D = 0.0148Q^2 + 0.33Q$.
Estimated velocity at medium flow	0.29 m s ⁻¹	From Upper Thames dye tracer data, $v = 0.028Q^{0.7}$
Estimated depth at medium flow	2.1 m	Calculated from $d = Q/v.W$
Low volumetric flow rate		
Low volumetric flow rate	6.2 m ³ s ⁻¹	10 percentile low flow from EA measurements at Sutton Courtney for year 2000.
Estimated coefficient of dispersion at low flow	2.6 m ² s ⁻¹	From Upper Thames dye tracer data, $D = 0.0148Q^2 + 0.33Q$.
Estimated velocity at low flow	0.1 m s ⁻¹	From Upper Thames dye tracer data, $v = 0.028Q^{0.7}$
Estimated depth at low flow	1.35 m	Calculated from $d = Q/v.W$

* *Data are given for the whole river: corrections for the Mill Stream branch are given in Table 7.*

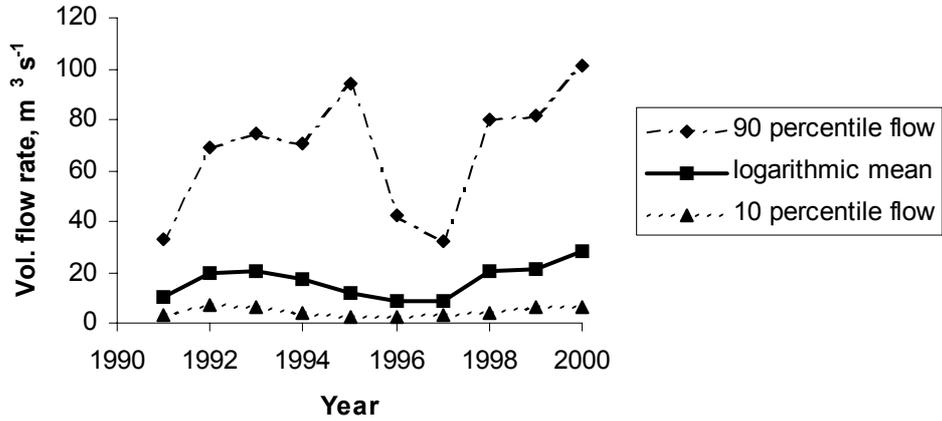
Table 6 Summary of river characteristics in the GUC/Colne reach

Parameter	Value	Notes
Average cross sectional area of river	14.8 m ²	Mean of 3 sites within 10km downstream of the discharge. From EA depth charts.
Average width of river	19.9 m	
Mean annual volumetric flow rate	5.0 m ³ s ⁻¹	Logarithmic mean from EA measurements at Denham for year 2000.
High volumetric flow rate		
High volumetric flow rate	10.5 m ³ s ⁻¹	90 percentile flow from EA measurements at Denham for year 2000.
Estimated coefficient of dispersion at high flow	19.2 m ² s ⁻¹	From Upper Thames data, $D = 0.0148Q^2 + 0.33Q$. Scale to Colne using observed $D = 12.4 \text{ m}^2 \text{ s}^{-1}$, at $Q = 8 \text{ m}^3 \text{ s}^{-1}$ and $D = 4.7 \text{ m}^2 \text{ s}^{-1}$, at $Q = 3.1 \text{ m}^3 \text{ s}^{-1}$ gives $D = 0.056Q^2 + 1.24Q$.
Estimated velocity at high flow	0.38 m s ⁻¹	From Upper Thames and 2 sets of Colne dye tracer data, $v = 0.074Q^{0.7}$
Estimated depth at high flow	1.4 m	Calculated from EA depth charts and field measurements, $d = Q/v.W$
Medium volumetric flow rate		
Medium volumetric flow rate	5.0 m ³ s ⁻¹	Logarithmic mean of EA measurements at Denham for year 2000
Estimated coefficient of dispersion at medium flow	7.6 m ² s ⁻¹	From Upper Thames data, $D = 0.0148Q^2 + 0.33Q$. Scale to Colne using observed $D = 12.4 \text{ m}^2 \text{ s}^{-1}$, at $Q = 8 \text{ m}^3 \text{ s}^{-1}$ and $D = 4.7 \text{ m}^2 \text{ s}^{-1}$, at $Q = 3.1 \text{ m}^3 \text{ s}^{-1}$ gives $D = 0.056Q^2 + 1.24Q$.
Estimated velocity at medium flow	0.23 m s ⁻¹	From Upper Thames and 2 sets of Colne dye tracer data, $v = 0.074Q^{0.7}$
Estimated depth at medium flow	1.1 m	Calculated from EA depth charts and field measurements, $d = Q/v.W$
Low volumetric flow rate		
Low volumetric flow rate	3.1 m ³ s ⁻¹	10 percentile low flow from EA measurements at Denham for year 2000.
Estimated coefficient of dispersion at low flow	4.4 m ² s ⁻¹	From Upper Thames data, $D = 0.0148Q^2 + 0.33Q$. Scale to Colne using observed $D = 12.4 \text{ m}^2 \text{ s}^{-1}$, at $Q = 8 \text{ m}^3 \text{ s}^{-1}$ and $D = 4.7 \text{ m}^2 \text{ s}^{-1}$, at $Q = 3.1 \text{ m}^3 \text{ s}^{-1}$ gives $D = 0.056Q^2 + 1.24Q$.
Estimated velocity at low flow	0.16 m s ⁻¹	From Upper Thames and 2 sets of Colne dye tracer data, $v = 0.074Q^{0.7}$
Estimated depth at low flow	1.0 m	Calculated from EA depth charts and field measurements, $d = Q/v.W$

(a) Pangbourne reach



(b) Sutton Courtney reach



(c) GUC/Colne reach

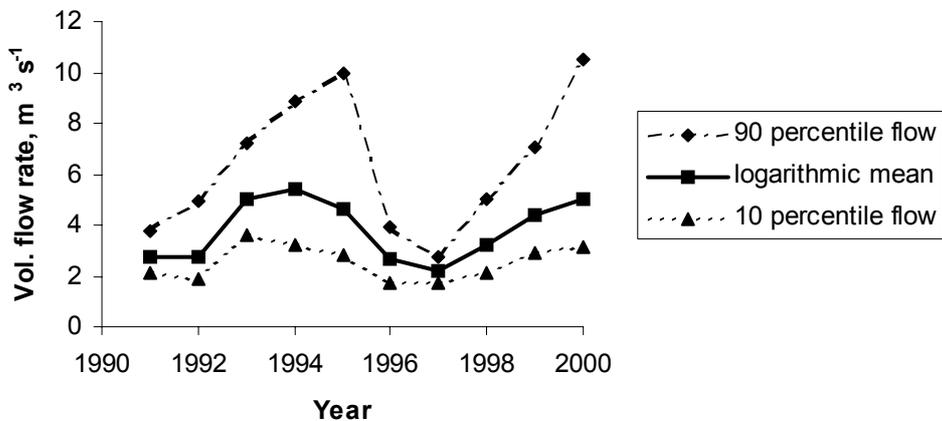


Figure 3 Annual volumetric flow rate characteristics of the different reaches for the years 1991-2000

Table 7 Mixing of the effluent in each reach

Reach	Distance downstream of discharge point (m)	Transverse and vertical mixing assumptions of the plume
Pangbourne	0	Pipeline stretches half way across river bed, discharge is from 8 nozzles in pipeline. Volumetric flow in the pipeline is approximately $2.2 \times 10^{-3} \text{ m}^3 \text{ s}^{-1}$. Maximum activity concentration (Bq l^{-1}) at the discharge point is therefore: $C_w(\text{max}) \approx \frac{C_i \times 10^{-3}}{2.2 \times 10^{-3} T_i}$ though this will be rapidly diluted as the discharge mixes with the river water.
	100	Vertical mixing is expected to have been achieved, but transverse mixing will just have begun. Assume that the plume stretches half way across the river, so on the pipeline side of the river $C_w \approx 2 \times C_w(\text{average})$ and on the other side $C_w \approx 0$.
	300	Transverse mixing will have occurred. Assume that for 2/3 of the river width $C_w \approx 1.5 \times C_w(\text{average})$ and on the 1/3 of the river on the side opposite the pipeline, $C_w \approx 0$.
	1,000	Transverse mixing will almost be complete. Assume complete mixing so average cross section activity concentrations are representative of the whole cross section.
	3,000	Complete transverse mixing (site is below a weir)
	10,000	Complete transverse mixing (site is below a weir)
Sutton Courtney	0	Pipeline stretches all the way across a weir on the Mill Stream branch of the river ensuring complete vertical mixing of the discharge. Volumetric flow in the Mill Stream is a minimum of $0.44 \text{ m}^3 \text{ s}^{-1}$, up to 20 times lower than the total river volumetric flow rate. Maximum activity concentration (Bq l^{-1}) at the discharge point is given by $C_w(\text{max}) \approx \frac{C_i \times 10^{-3}}{0.44 T_i}$
	100	Confluence of the Mill Stream with the main branch of the river occurs approximately at this point. Assume activity concentration in the Mill Stream is: $C_w(\text{max}) \approx \frac{C_i \times 10^{-3}}{0.44 T_i}$ which also gives the maximum activity concentration shortly after mixing of the two flows.
	300	Mixing of the two river flows will have begun. Assume that for 1/3 of the river width $C_w \approx 3.0 \times C_w(\text{average})$ and on the 2/3 of the river on the side opposite the Mill Stream inflow, $C_w \approx 0$.
	1,000	Mixing of the two river flows will be nearly complete. Assume that for 2/3 of the river width $C_w \approx 1.5 \times C_w(\text{average})$ and on the 1/3 of the river on the side opposite the Mill Stream inflow, $C_w \approx 0.5 \times C_w(\text{average})$.
	3,000	Assume complete transverse mixing
	10,000	Assume complete transverse mixing
Maple Lodge	0	Average volumetric flow rate of Maple Lodge STW outflow is $1.4 \text{ m}^3 \text{ s}^{-1}$. Maximum activity concentration (Bq l^{-1}) at the discharge point is therefore: $C_w(\text{max}) \approx \frac{C_i \times 10^{-3}}{1.4 \times T_i}$
	100	Assume discharge is mixed half way across river. On the discharge side of the river $C_w \approx 2 \times C_w(\text{average})$ and on the other side $C_w \approx 0$.
	300	Assume that for 2/3 of the river width, $C_w \approx 1.5 \times C_w(\text{average})$ and on the 1/3 of the river on the side opposite the discharge $C_w \approx 0$.
	1,000	Assume complete transverse mixing (site is below weir)
	3,000	Assume complete transverse mixing
	10,000	Assume complete transverse mixing

3.5 Sorption to Sediment

Radionuclides discharged into any river system may become sorbed to suspended sediments and may then settle to the river bed. This process is normally modelled by a distribution coefficient (K_d , $l\text{ kg}^{-1}$) approach, where the K_d is defined as the radioactivity concentration in the solid phase (Bq kg^{-1}) divided by the concentration in dissolved phase (Bq l^{-1}).

K_d values reported are mostly for equilibrium situations where radionuclides in solution phase are in contact with suspended sediments for sufficient time for equilibrium or near equilibrium between water and suspended solids to become established. They are therefore suitable for continuous discharges. For short duration discharges, contact times between water and sediment are likely to be shorter. The K_d approach has been used here in modelling short duration releases, but the K_{dS} used have been selected to reflect the shorter duration of contact (Table 8). The modelling carried out used estimates the fraction of activity sorbed to suspended sediments (f_p) for a given suspended sediment concentration. In order to give conservative estimates which account for potential uncertainty in the solid-solution partitioning, uptake of radionuclides onto sediments was ignored when water concentrations and fish uptake were modelled. When modelling activity concentrations in bed sediments, upper bounding values of f_p were used (Table 8).

Table 8 Estimated (short term) K_d ($l\text{ kg}^{-1}$) values and fraction of radionuclide sorbed to solid phase assuming a suspended solids concentration, s , of 13.0 mg l^{-1}

Element	Best estimate K_d [$l\text{ kg}^{-1}$]	Fraction in solid phase []	Recommended value for model (water, fish)	Recommended value for model (bed sediment)
^3H	1	0.00001	0	0
^{14}C	1×10^4	0.12	0	0.95
^{32}P	5.7×10^3	0.07	0	0.95
Cs	5×10^3	0.04	0	0.95
Sr	10^2	0.0013	0	0.05
Zn	5×10^2	0.0065	0	0.05
I	50	0.00065	0	0.05
Co	10^3	0.013	0	0.05
U	50	0.00065	0	0.05
Pu	10^3	0.013	0	0.05
Am	5×10^3	0.04	0	0.95

4. MODEL OUTPUT FOR SITES IN THE THAMES CATCHMENT

A model has been developed which can be used to predict the concentrations of radionuclides in the river environment, ie in river water, river bed sediment and in predatory fish. The model was then used to predict the concentrations of the radionuclides in two reaches of the river Thames and in the GUC/River Colne. Model output is given as a series of graphs of activity concentration and time integrated activity concentration resulting from a hypothetical 1 MBq discharge for 5 release durations and five locations downstream. The release durations modelled were 5 minutes, 1 h, 3 h, 12 h and 24 h. The five locations for which predictions were required were 100 m, 300 m, 1000 m, 3000 m and 10000 m downstream. The discharges considered were hypothetical and were designed to enable a model to be developed and tested over a range of river reaches with different characteristics and for a range of short duration discharges.

4.1 Estimating Maximum Water Concentrations Close to the Discharge Point

Of all the target variables, the maximum water concentration is the most dependent on river characteristics and time period of discharge. For maximum concentrations close to the discharge point, however, it is possible to relate maximum water concentration simply to the river volumetric flow rate (Figure 4). Figure 4 is only valid for distances down river, x' , where $x' < v \times T_i$. For the three sites studied here, Figure 4 applies for distances up to 24, 144, 864, 3456 and 6912 metres for times of discharge 5 min, 30 min, 3 h, 12 h, 24 h respectively. Thus, for a 5 minute discharge, Figure 4 gives the maximum water concentration at any distance up to 24 metres downstream of the discharge. For a 24 hour discharge, Figure 4 gives the maximum water concentration at any distance up to 6912 metres downstream of the discharge. These minimum distances are for the lowest water velocity and will be greater for medium or high flow conditions. For distances greater than $x' < v \times T_i$, Figure 4 will over-estimate maximum activity concentrations in water and the graphs presented in section 4.2 should be used.

Figure 4 gives estimates for maximum water concentrations in the dissolved phase where there is no sorption to suspended sediments ($f_p = 0$). For the three sites studied here, it was assumed, when calculating activity concentrations in water, that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$, Figure 4 gives the total activity concentration, $C_T(x)$, in both phases which can be converted to a dissolved phase activity concentration using: $C_w(x) = C_T(x) \times (1 - f_p)$.

4.1.1 Required input data

- Volumetric flow rate of the river
- Fraction of radioactivity in particulate phase (f_p)

4.1.2 Notes and assumptions

1. It is assumed that the radionuclide is discharged in the dissolved phase at a uniform rate during the discharge period.

2. The estimate in Figure 4 is for the average concentration across the river cross section. Correction factors for transverse mixing (Table 7 and Appendix B.3) may be required if the source is not well mixed across the river.
3. It is assumed that there is no dispersion of the plume downstream. Figure 4 should therefore only be used for distances down river, x' , where $x' < v \times T_i$. For the three sites studied here, Figure 4 applies for distances of 24, 144, 864, 3456 and 6912 metres for times of discharge 5 min, 30 min, 3 h, 12 h, 24 h respectively. These minimum distances are for the lowest water velocity and will be greater for medium or high flow conditions. Downstream of distances x' , Figure 4 will overestimate (by up to approximately one order of magnitude or more) the maximum water concentration.
4. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the maximum activity concentration of 12%.
5. In the present study, it was assumed, when calculating activity concentrations in water that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$ it is conservatively assumed that there is no transfer of radioactivity to bed sediments. Under this assumption, sorption of radioactivity to suspended sediments, therefore, reduces the amount of radioactivity in the dissolved phase of the river water ($C_w(x) = C_T(x) \times (1 - f_p)$) but has no effect on the total activity concentration passing point x . For the reaches studied here, this would lead to a maximum over-estimate of the total and dissolved phase maximum activity concentrations of approximately a factor of 2. This maximum over-estimate would occur for the furthest point ($x = 10$ km) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity.

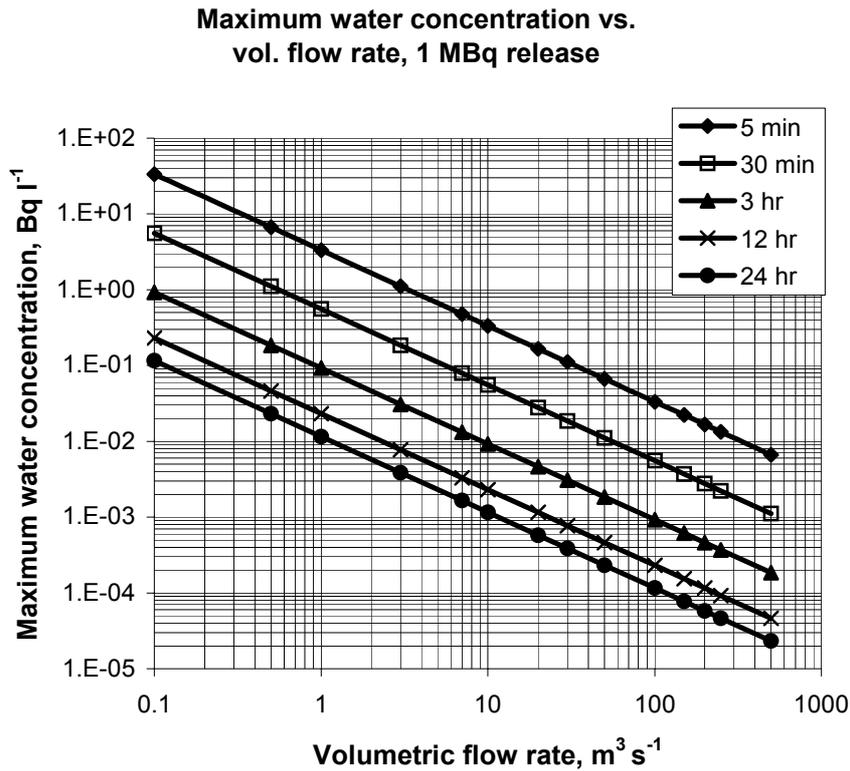


Figure 4 Maximum water concentrations (cross section average) for different discharge times for sites close to the discharge point. Applies to a 1 MBq release of any of the radionuclides

4.2 Estimating Maximum Water Concentrations Downstream of the Discharge Point

Downstream of the discharge point, maximum water concentrations may be estimated using the simplified model in Appendix G. Since this varies with time of discharge, volumetric flow rate, water velocity and coefficient of dispersion, it is not possible at this stage to give simple graphical output which is applicable to all rivers. Output specific to the three sites studied will therefore be given here as Figures 5-7. Generalised estimates of downstream activity concentrations for other rivers are discussed in Section 5 below.

Figures 5-7 give estimates for maximum water concentrations in the dissolved phase where there is no sorption to suspended sediments ($f_p = 0$). For the three sites studied here, it was assumed, when calculating activity concentrations in water, that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$, it gives the total activity concentration in both phases which can be converted to a dissolved phase activity concentration using: $C_w(x) = C_T(x) \times (1 - f_p)$.

Figure 8 gives estimates of maximum water concentrations for a given volumetric flow rate ($Q = 10 \text{ m}^3 \text{ s}^{-1}$) for releases of duration 5 minutes, 3 hours and 24 hours.

4.2.1 Required input data

- Volumetric flow rate (low, medium, high) of river
- Fraction of radioactivity in particulate phase (f_p)
- Distance downstream of discharge.

4.2.2 Notes and assumptions

1. It is assumed that the radionuclide is discharged in the dissolved phase at a uniform rate during the discharge period.
2. The estimates in Figures 5-7 are for the average concentration across the river cross section. Correction factors for transverse mixing (Table 7 and Appendix B3) may be required if the source is not well mixed across the river.
3. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the maximum activity concentration of 12%.
4. In the present study, it was assumed, when calculating activity concentrations in water that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$ it is conservatively assumed that there is no transfer of radioactivity to bed sediments. Sorption of radioactivity to suspended sediments, therefore, in the model, reduces the amount of radioactivity in the dissolved phase ($C_w(x) = C_T(x) \times (1 - f_p)$) but has no effect on the maximum total water activity concentration. For the reaches studied here, this would lead to a maximum over-estimate of the total and dissolved phase activity concentrations of approximately a factor of 2. This maximum over-estimate would occur for the furthest point ($x = 10$ km) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity.

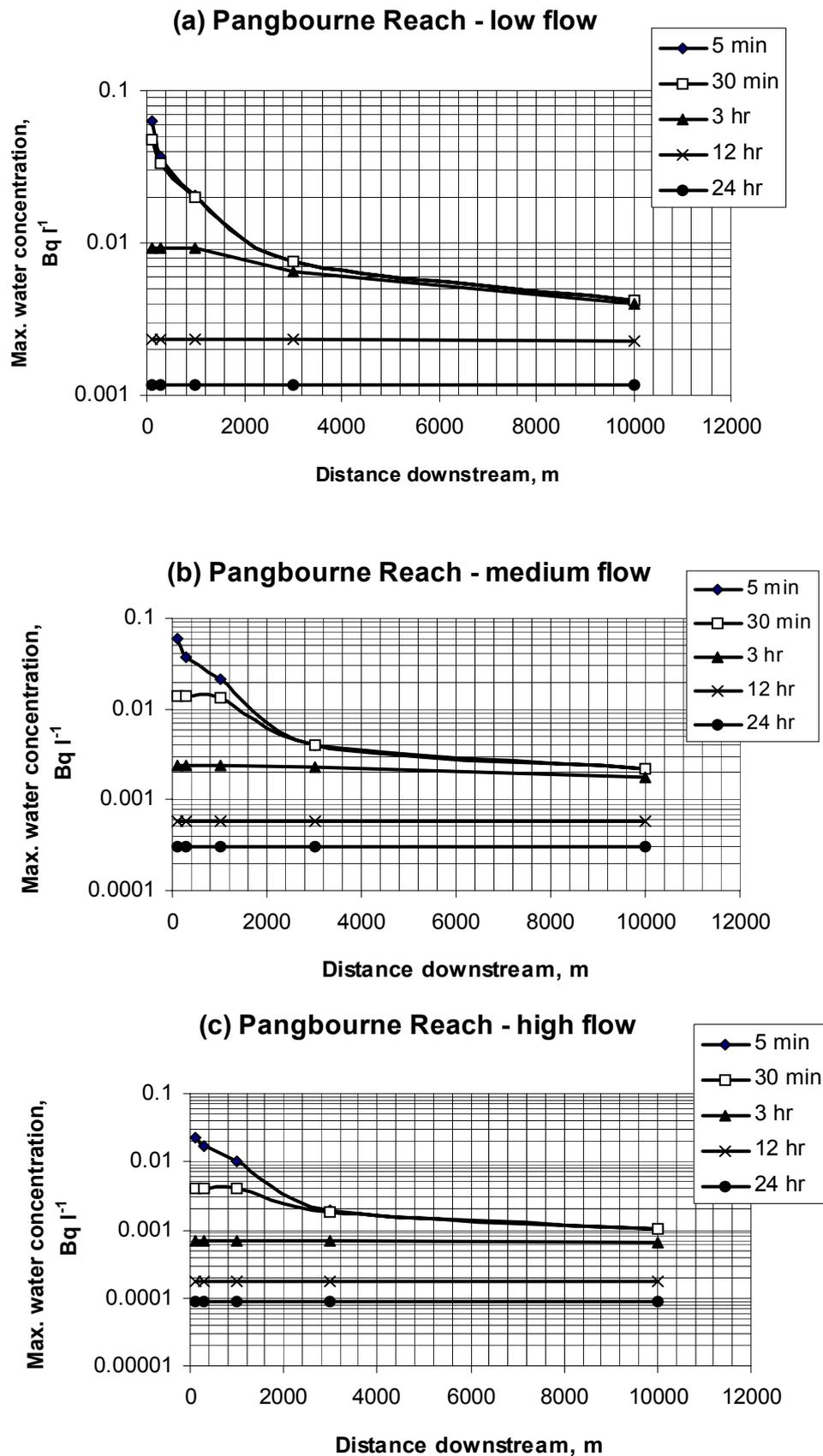


Figure 5 Maximum water concentration at different distances downstream of the assumed discharge point, for different discharge times (1 MBq input) of any of the radionuclides

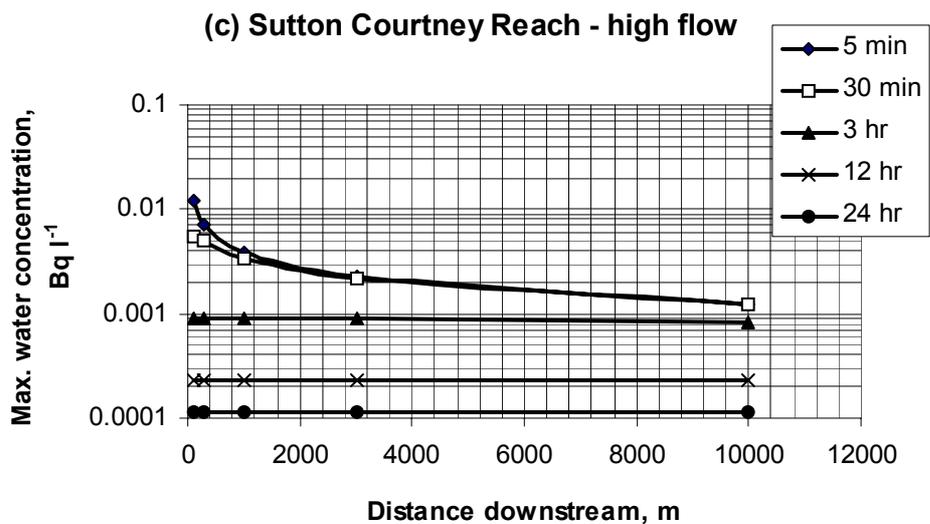
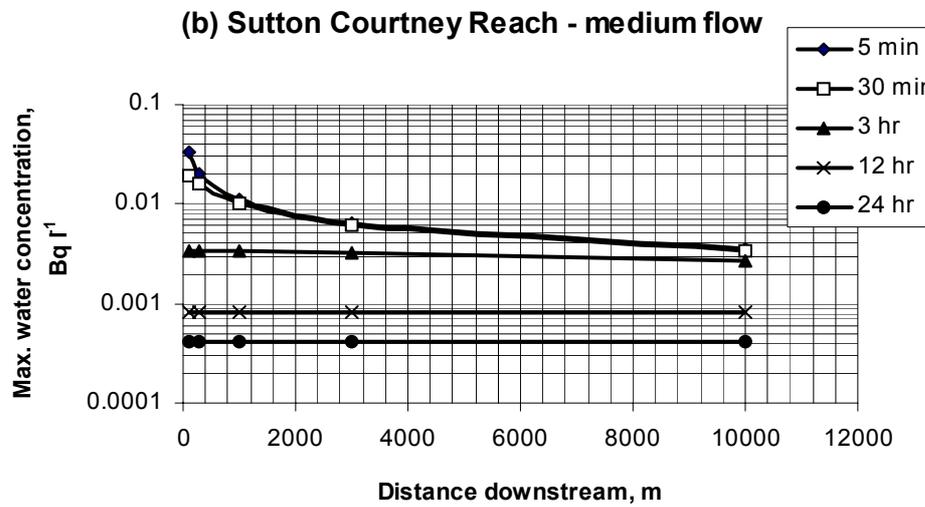
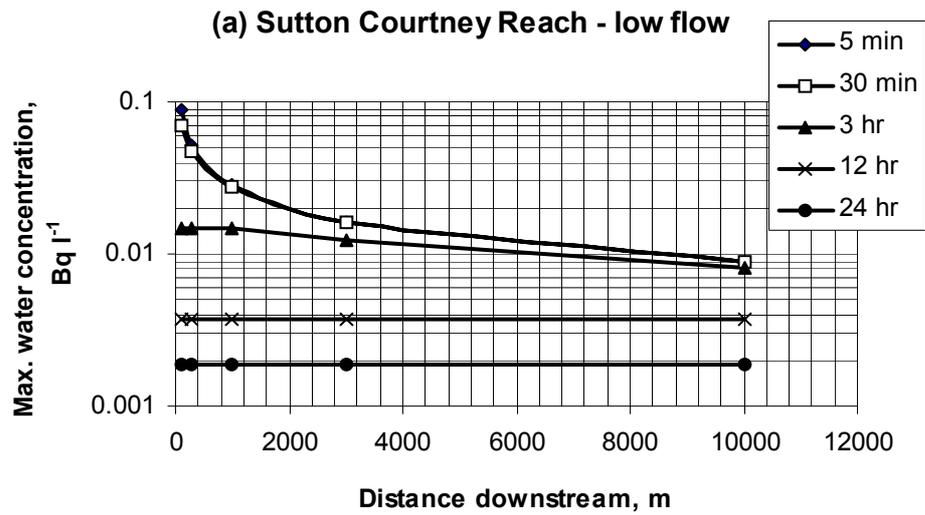


Figure 6 Maximum water concentration at different distances downstream of the assumed discharge point, for different discharge times (1 MBq input) of any of the radionuclides

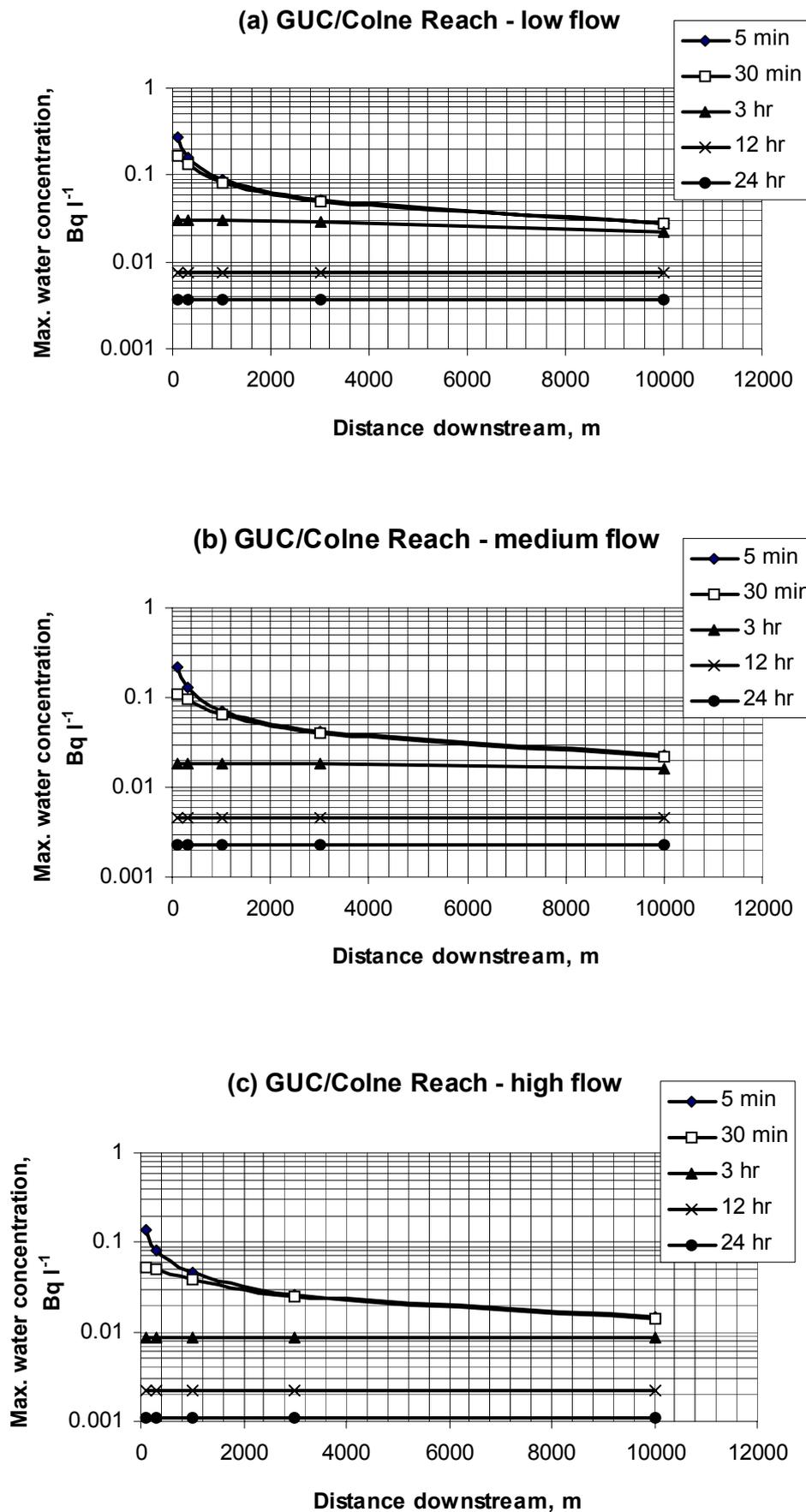
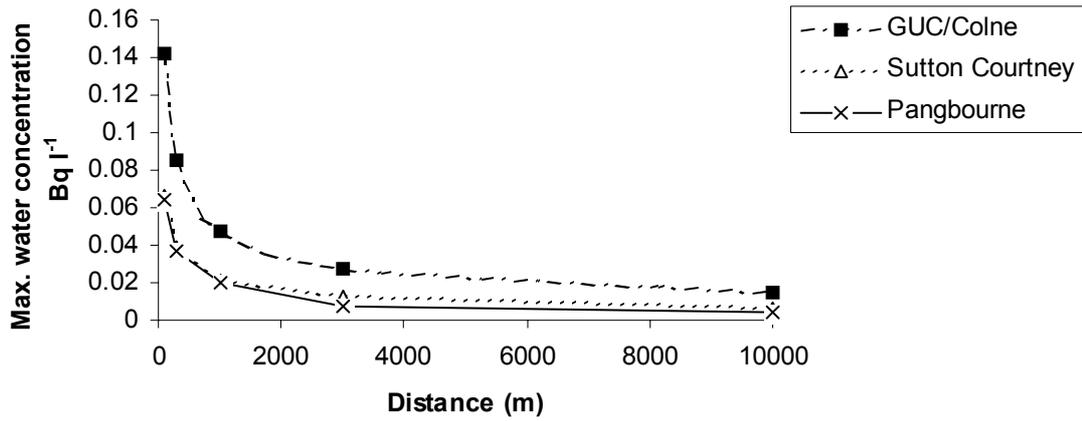
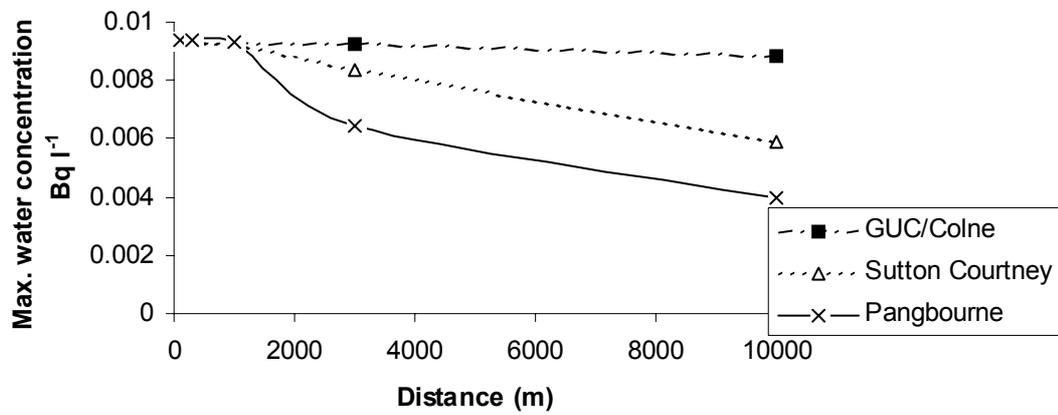


Figure 7 Maximum water concentration at different distances downstream of Maple Lodge, for different discharge times (1 MBq input) of any of the radionuclides

Release duration 5 min; $Q = 10 \text{ m}^3/\text{s}$



Release duration 3 hr; $Q = 10 \text{ m}^3/\text{s}$



Release duration 24 hr; $Q = 10 \text{ m}^3/\text{s}$

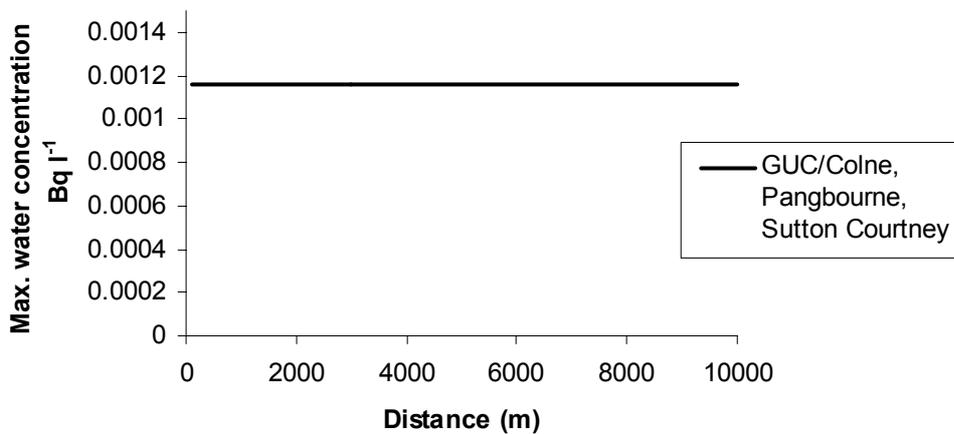


Figure 8 Maximum concentration in water for a volumetric flow rate of $10 \text{ m}^3 \text{ s}^{-1}$ following releases of different durations

4.3 Estimating Time Integrated Water Concentrations

Figure 9 gives the estimated integrated (dissolved and total) water concentration, $\Sigma C_w(x)$, for all radionuclides, for all release times ≤ 24 h, for all distances x and for all integration times greater than the time of travel of the plume out of reach x . For the three reaches studied here, this applies to all sites for all integration times > 3 days. Once the plume has passed out of the reach, under the assumptions given below, the time integrated activity concentration in water does not vary, so Figure 9 applies to all integration times greater than 3 days.

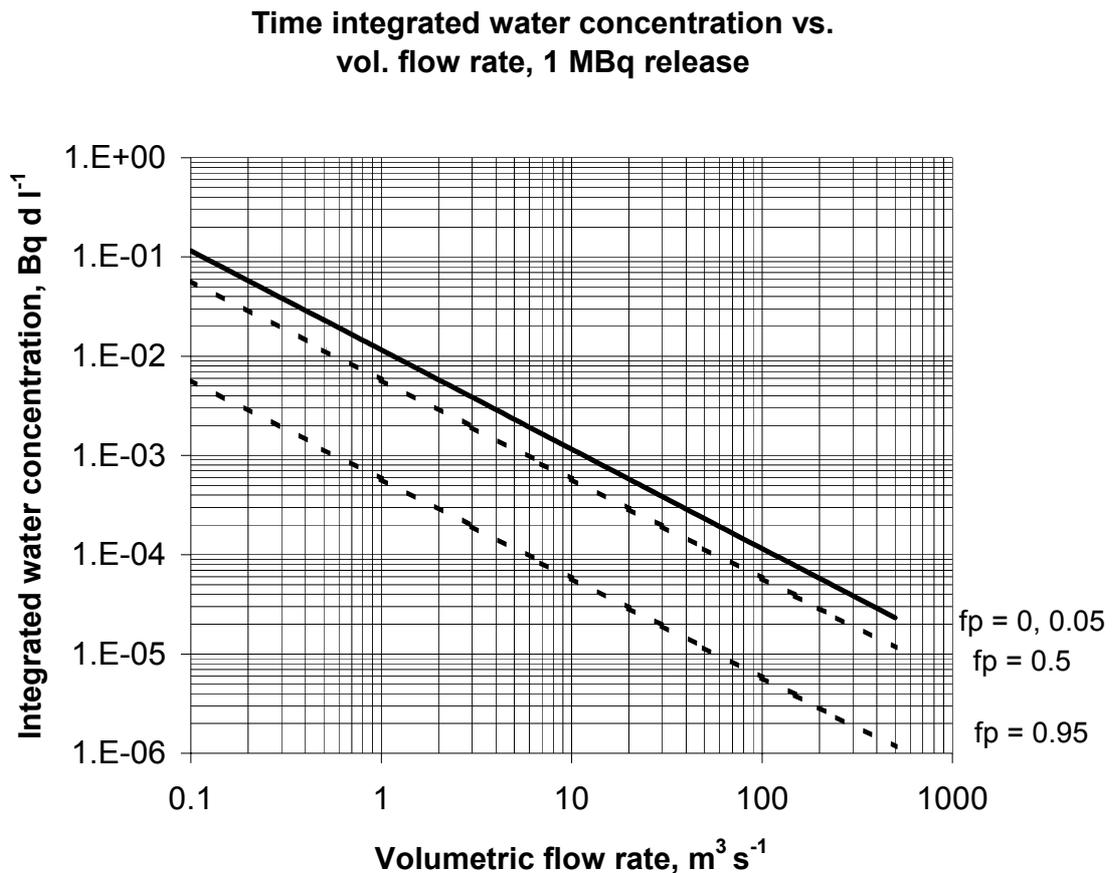


Figure 9 Time integrated water concentrations (for all times greater than 3 days) vs. volumetric flow rate (1 MBq release) for all radionuclides, all release times ≤ 24 h, and all distances downstream up to 10 km. The solid line gives estimates for total (dissolved plus particulate phase) activity concentration and for dissolved-phase activity concentration where $f_p \leq 0.05$. The dotted lines give estimates of dissolved-phase activity concentration for $f_p = 0.5$ and for $f_p = 0.95$ as indicated

4.3.1 Required input data

- Volumetric flow rate of the river
- Fraction of radioactivity in particulate phase (f_p)

4.3.2 Notes and assumptions

1. The estimate in Figure 9 is for the average concentration across the river cross section. Correction factors for transverse mixing (Table 7 and Appendix B.3) may be required if the source is not well mixed across the river.
2. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the integrated activity concentration of 12%.
3. In the present study, it was assumed, when calculating activity concentrations in water that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$, it is conservatively assumed that there is no transfer of radioactivity to bed sediments. Sorption of radioactivity to suspended sediments, therefore, reduces the amount of radioactivity in the dissolved phase ($\Sigma C_w(x) = \Sigma C_T(x) \times (1 - f_p)$) but in the model has no effect on the time integrated total water activity concentrations. For the reaches studied here, this would lead to a maximum over-estimate of the total and dissolved phase integrated activity concentrations of approximately a factor of 2. This maximum over-estimate would occur for the furthest point ($x = 10$ km) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity.

4.4 Estimating Maximum Concentrations in Fish

Figure 10 gives the estimated maximum activity concentrations in whole fish (per kg wet weight) as a function of volumetric flow rate for all the radionuclides except tritium (see below). The estimates apply for all release times ≤ 24 h, for all distances x , and for a water temperature of 12°C. For other temperatures, the maximum concentrations should be multiplied by the correction factors given in Table 9.

Figure 10 gives estimates for maximum water concentrations in fish where there is no sorption to suspended sediments ($f_p = 0$). For the three sites studied here, it was assumed, when calculating activity concentrations in fish, that $f_p = 0$ for all radionuclides. In cases where $f_p > 0$, these estimates should be multiplied by $(1 - f_p)$: $C_f(x) \rightarrow C_f(x) \times (1 - f_p)$.

Table 9 Correction factors for converting estimates of maximum fish concentration from a water temperature of 12°C to other temperatures

Radionuclide	Ratio: $\frac{C_f(x) \text{ at } 7^\circ \text{C}}{C_f(x) \text{ at } 12^\circ \text{C}}$	Ratio: $\frac{C_f(x) \text{ at } 17^\circ \text{C}}{C_f(x) \text{ at } 12^\circ \text{C}}$
All RN's except Sr	0.42	2.0
Sr-89, Sr-90	1.0	1.0

Estimates of maximum concentration of tritium (H-3) in fish may be made by multiplying the maximum concentration of tritium in water by the fish-water concentration factor. Since, for tritium, $CF = 1.0 \text{ l kg}^{-1}$ for all water temperatures, this is equal to the maximum water concentration.

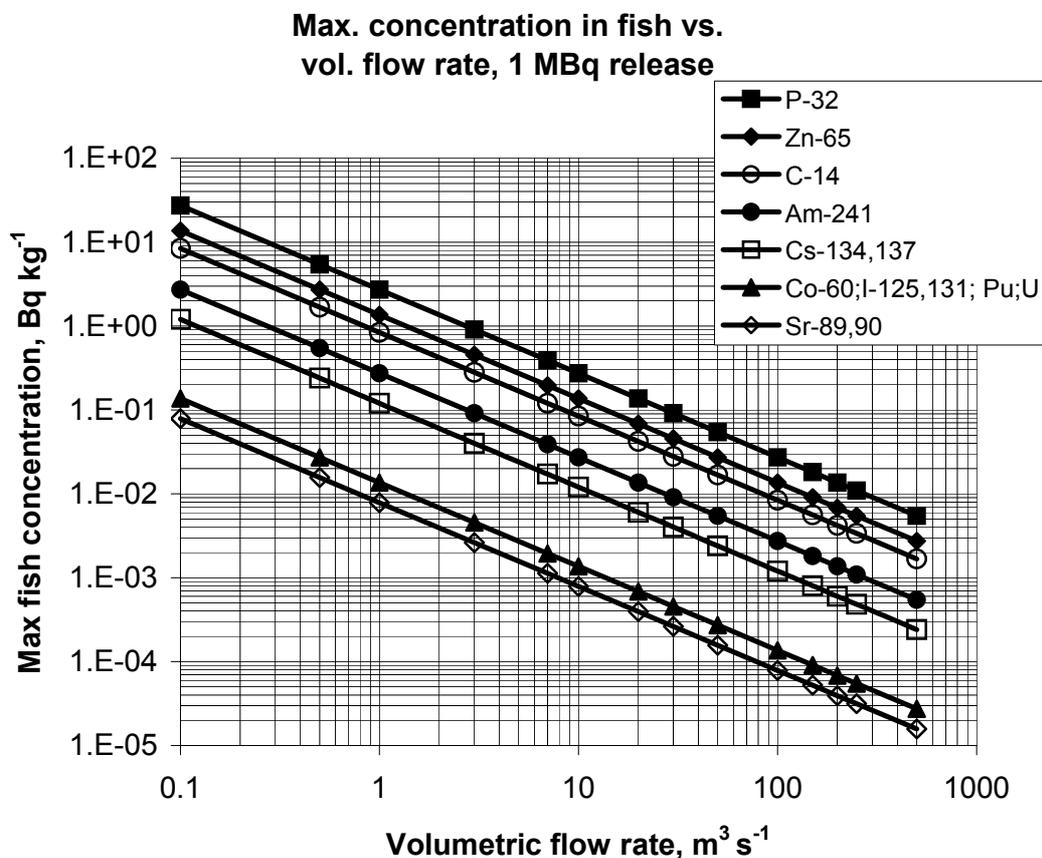


Figure 10 Estimated maximum concentrations in fish for a 1MBq release of different radionuclides at a water temperature of 12°C. For other temperatures, the maximum concentrations should be multiplied by the correction factors given in Table 9. The results apply to all distances <10 km downstream and for all release times

4.4.1 Required input data

- Volumetric flow rate of the river
- Fraction of radioactivity in particulate phase (f_p)

4.4.2 Notes and assumptions

1. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the maximum activity concentration of 12%.
2. In the present study, it was assumed, when calculating activity concentrations in fish that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$, it is conservatively assumed that there is no transfer of radioactivity to bed sediments. Sorption of radioactivity to suspended sediments, therefore, reduces the amount of radioactivity in the dissolved phase but in the model has no effect on the maximum activity concentration in fish at point x . For the reaches studied here, this leads to a maximum over-estimate of the maximum fish activity concentrations of approximately a factor of 2. This maximum over-estimate occurs for the furthest point ($x = 10$ km) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity.
3. The estimates give maximum concentrations for an average fish in a particular reach of river. Maxima for individual fish will vary according to feeding behaviour and movement of the fish within the river.

4.5 Time Integrated Concentrations in Fish

Figure 11 gives the estimated time integrated activity concentrations (for integration times of one week and one year) in whole fish (per kg wet weight for a 500 g Trout) as a function of volumetric flow rate. The graphs apply to all the radionuclides except tritium (see below). The estimates apply for all release times ≤ 24 h, for all distances x , and for a water temperature of 12°C. For other temperatures, the integrated concentrations should be multiplied by the correction factors given in Table 11. These correction factors are different to those in Table 9 (particularly for the one year integration time) because different physical decay rates and excretion rates affect the time integrated concentration but have less effect on the maximum activity concentration.

Figure 11 gives estimates for integrated water concentrations in fish where there is no sorption to suspended sediments ($f_p = 0$). For the three sites studied here, it was assumed, when calculating activity concentrations in fish, that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$, these estimates should be multiplied by $(1-f_p)$: $\sum C_f(x) \rightarrow \sum C_f(x) \times (1 - f_p)$.

Table 10 Correction factors for converting estimates of time integrated fish concentration from a water temperature of 12°C to other temperatures

Radionuclide	Ratio: $\frac{\Sigma C_f(x) \text{ at } 7^\circ C}{\Sigma C_f(x) \text{ at } 12^\circ C}$	Ratio: $\frac{\Sigma C_f(x) \text{ at } 17^\circ C}{\Sigma C_f(x) \text{ at } 12^\circ C}$
One week integration time		
All RN's except Sr	0.45	2.0
Sr-89,90	1.0	1.0
One year integration time		
Cs-134, Cs-137	0.65	1.24
P-32	0.52	1.52
I-125	0.70	1.21
I-131	0.49	1.66
Sr-89, Sr-90	1.0	1.0
Co-60	0.53	1.44
C-14	0.57	1.30
Pu, U, Am	1.0	1.0
Zn-65	0.88	1.07

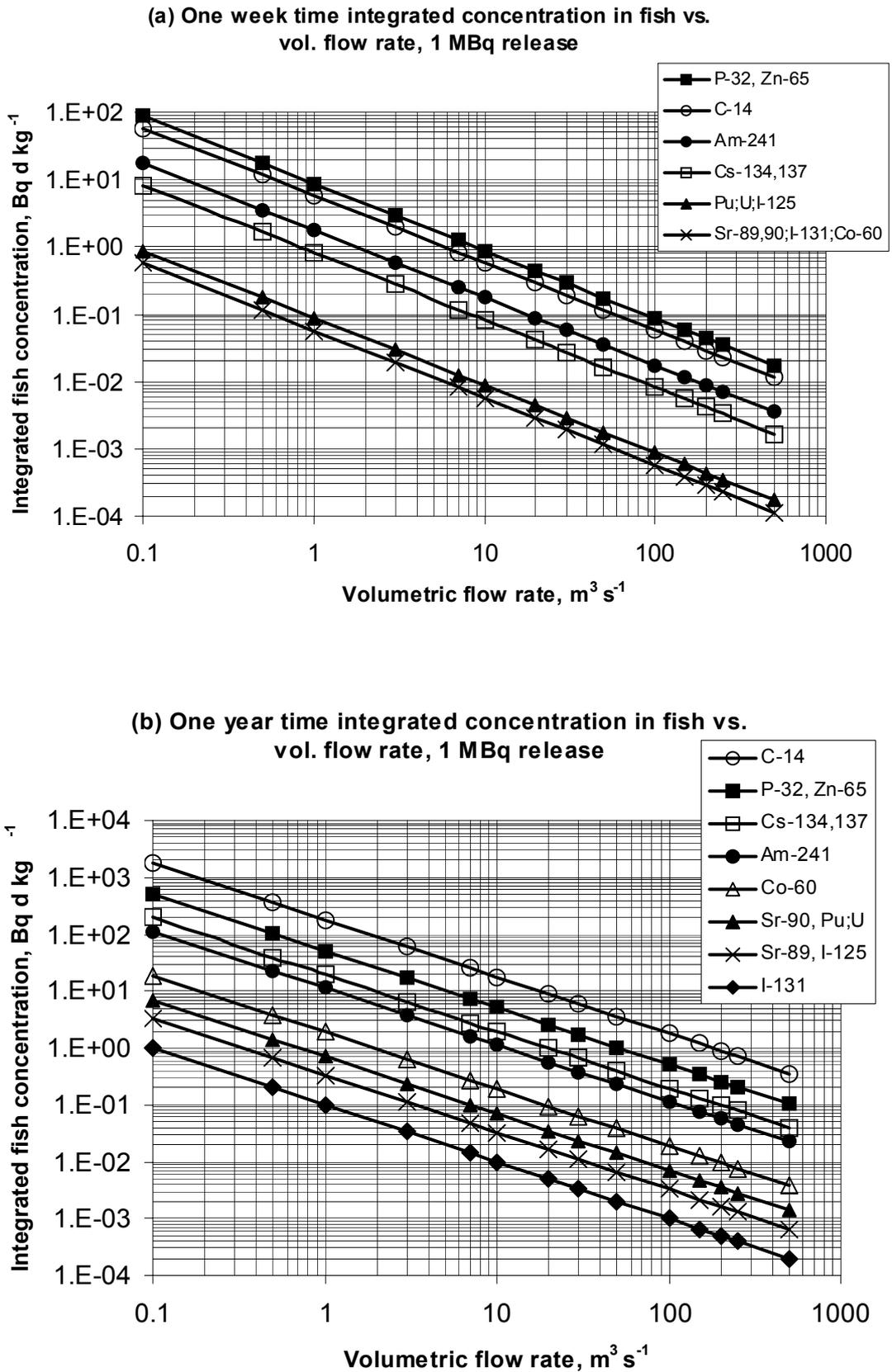


Figure 11 Estimated (a) one week; (b) one year time integrated activity concentrations in fish for a 1MBq release of different radionuclides at a water temperature of 12°C. For other temperatures, the integrated concentrations should be multiplied by correction factors given in Table 10. Results apply to all distances <10 km downstream and for all release times

Estimates of the time integrated concentration (all integration times > 3 days) of tritium (H-3) in fish may be made by multiplying the time integrated concentration of tritium in water by the fish-water concentration factor. Since, for tritium, $CF = 1.0 \text{ l kg}^{-1}$ for all water temperatures, this is equal to the integrated water concentration.

4.5.1 Required input data

- Volumetric flow rate of the river
- Fraction of radioactivity in particulate phase (f_p)

4.5.2 Notes and assumptions

1. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the integrated activity concentration of 12%.
2. In the present study, it was assumed, when calculating activity concentrations in fish that $f_p = 0$ for all radionuclides. For cases where $f_p > 0$, it is conservatively assumed that there is no transfer of radioactivity to bed sediments. Sorption of radioactivity to suspended sediments, therefore, reduces the amount of radioactivity in the dissolved phase but in the model has no effect on the integrated activity concentration in fish at point x . For the reaches studied here, this leads to a maximum over-estimate of the integrated fish activity concentrations of approximately a factor of 2. This maximum over-estimate occurs for the furthest point ($x = 10 \text{ km}$) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity.
3. The estimates give integrated concentrations for an average fish in a particular reach of river. Concentrations for individual fish will vary according to feeding behaviour and movement of the fish within the river.

4.6 Maximum Concentration in Bed Sediments

Figure 12 gives the maximum activity concentration in bed sediments as a function of volumetric flow rate. The estimates apply to all distances downstream and to all radionuclides.

**Maximum sediment concentration vs.
vol. flow rate, 1 MBq release**

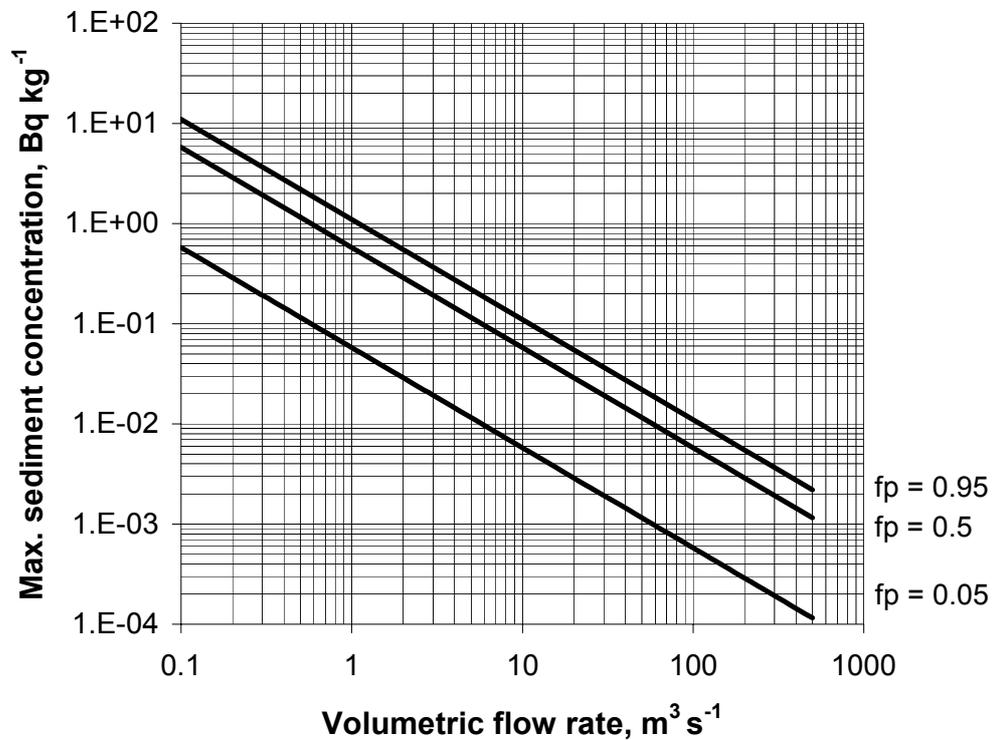


Figure 12 Maximum sediment concentrations (cross section average) for different volumetric flow rates and particulate sorbed fractions. Applies to a 1 MBq release of any of the radionuclides

4.6.1 Required input data

- Volumetric flow rate of the river
- Fraction of radioactivity in particulate phase (f_p)

4.6.2 Notes and assumptions

1. The estimate in Figure 12 is for the average concentration across the river cross section. Correction factors for transverse mixing (Table 7 and Appendix B.3) may be required if the source is not well mixed across the river.
2. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the maximum activity concentration of 12%.
3. It is conservatively assumed that there is no loss of radioactivity in the river water as a result of transfer of radioactivity to bed sediments. Sorption of radioactivity to suspended sediments, therefore, reduces the amount of

radioactivity in the dissolved phase ($C_w(x) = C_T(x) \times (1 - f_p)$) but in the model has no effect on the total activity concentration passing point x . For the reaches studied here, this leads to a maximum over-estimate of the total and dissolved phase integrated activity concentrations of approximately a factor of 2. This maximum over-estimate occurs for the furthest point ($x = 10$ km) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity.

4. The dry mass per unit wet volume of bed sediments, ρ_s (kg m^{-3}), is taken to be equal to that observed in the River Blackwater, a tributary of the Thames, $\rho_s = 500 \text{ kg m}^{-3}$ (F H Denison, unpublished results). The sediment mixing depth, d_s , is taken as 0.02 m.

4.7 Time Integrated Concentration in Bed Sediments

Figures 13 and 14 give the time integrated (one week and one year time periods) activity concentration in bed sediments as a function of volumetric flow rate. The estimates apply to all distances downstream and to all radionuclides.

4.7.1 Required input data

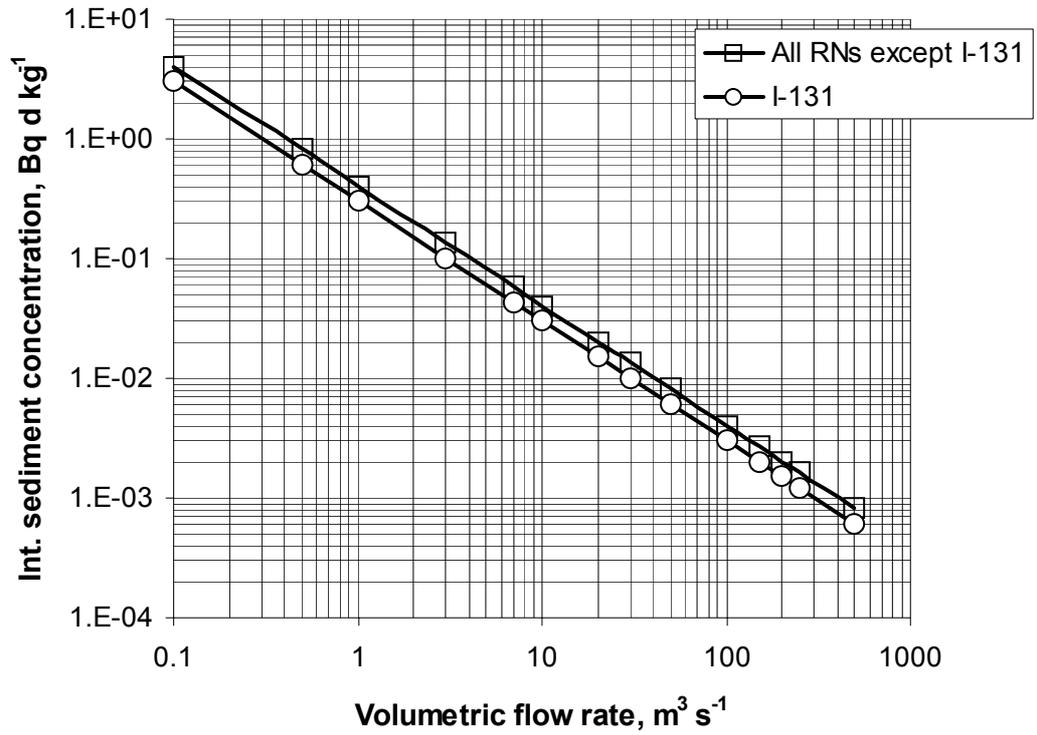
- Volumetric flow rate of the river
- Fraction of radioactivity in particulate phase (f_p)

4.7.2 Notes and assumptions

1. The estimates in Figures 13 and 14 are for the average concentration across the river cross section. Correction factors for transverse mixing (Table 7 and Appendix B.3) may be required if the source is not well mixed across the river.
2. It is conservatively assumed that there is no radioactive decay of the radionuclide as it travels down the river. For the fastest-decaying radionuclide (I-131) and for the longest travel time (10 km site downstream, at the end of the Pangbourne reach, low flow conditions) considered here, this leads to an over-estimate of the maximum activity concentration of 12%.
3. It is conservatively assumed that there is no loss of radioactivity in the river water as a result of transfer of radioactivity to bed sediments. Sorption of radioactivity to suspended sediments, therefore, reduces the amount of radioactivity in the dissolved phase ($C_w(x) = C_T(x) \times (1 - f_p)$) but in the model has no effect on the total activity concentration passing point x . For the reaches studied here, this leads to a maximum over-estimate of the total and dissolved phase integrated activity concentrations of approximately a factor of 2. This maximum over-estimate occurs for the furthest point ($x = 10$ km) for the highest particulate sorbed fraction ($f_p = 0.95$) and for the lowest combination of mean river depth and water velocity. The time integrated concentration for all radionuclides of $T_{1/2} > 2$ years is calculated assuming $T_{1/2} = \infty$. This leads to a maximum overestimate for Cs-134 ($T_{1/2} = 2.1$ years) of 15% and for Co-60 ($T_{1/2} = 5.27$ years) of 7%. For all other long-lived radionuclides the error is negligible.
4. The dry mass per unit wet volume of bed sediments, ρ_s (kg m^{-3}), is taken to be equal to that observed in the River Blackwater, a tributary of the Thames, $\rho_s =$

500 kg m⁻³ (F H Denison, unpublished results). The sediment mixing depth, d_s , is taken as 0.02 m.

(a) One week time integrated sediment concentration vs. vol. flow rate, 1 MBq release, $f_p = 0.05$



(b) One week time integrated sediment concentration vs. vol. flow rate, 1 MBq release, $f_p = 0.95$

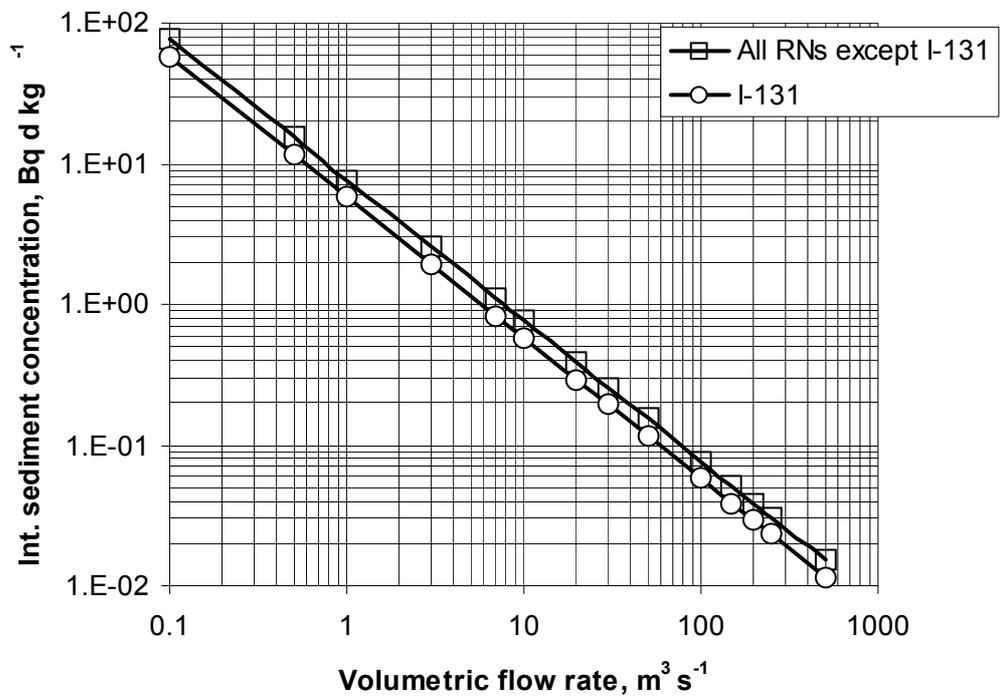


Figure 13 One-week time integrated activity concentrations in bed sediments following a 1 MBq release for (a) $f_p = 0.05$; (b) $f_p = 0.95$.

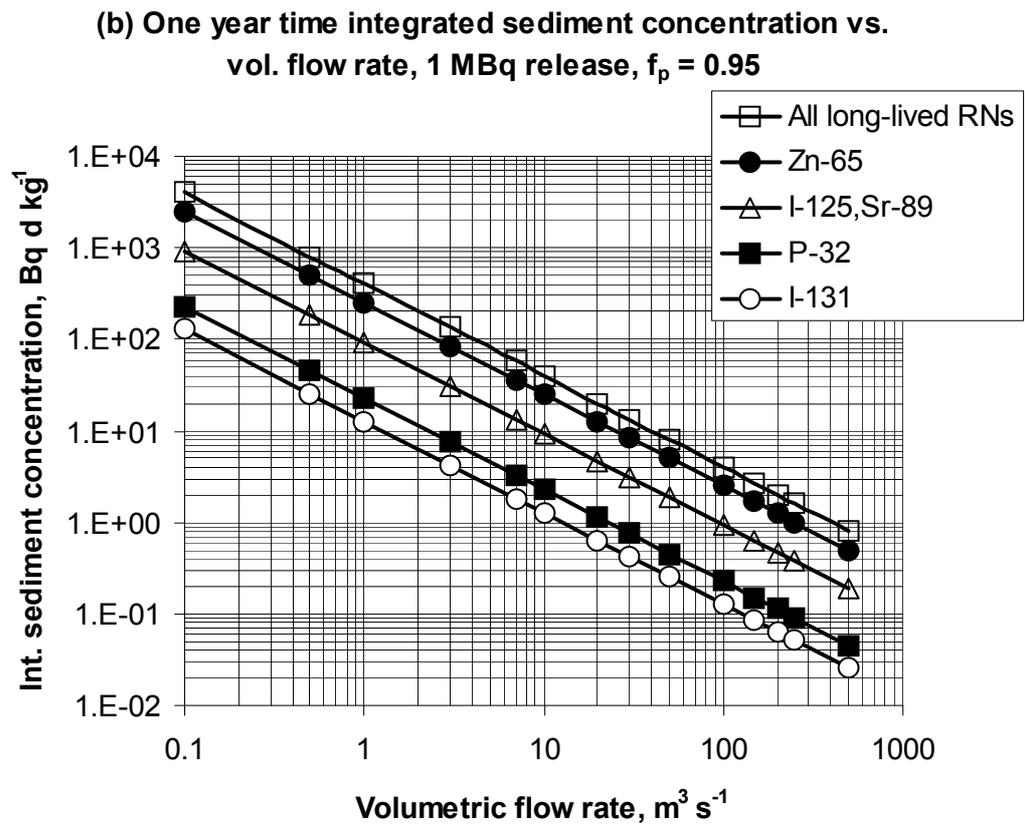
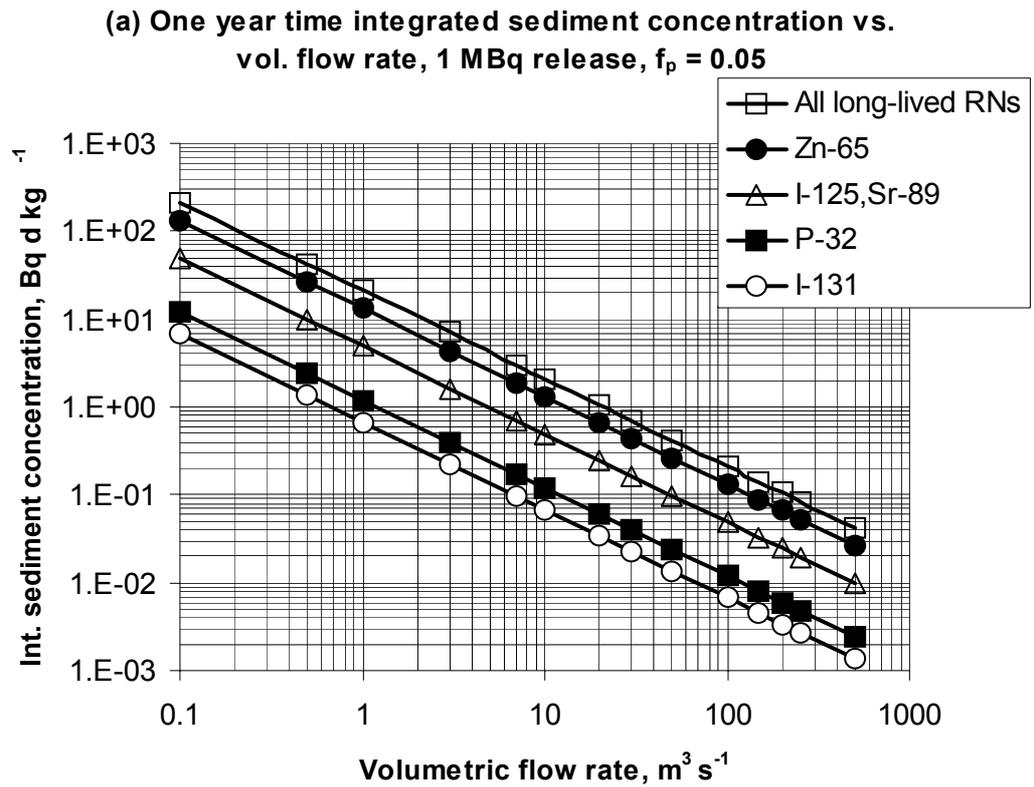


Figure 14 One-year time integrated activity concentrations in bed sediments following a 1 MBq release for (a) $f_p = 0.05$; (b) $f_p = 0.95$. “Long-lived” radionuclides are those of $T_{1/2} > 2$ y

4.8 Tabular Output

Tables are presented in Appendix H of instantaneous concentrations and time integrated concentrations in filtered river water, total water (filtered plus unfiltered), fish at 12° C and sediment, for a 1 MBq release to a river flowing at 10 m³ s⁻¹, release duration of 3 hours, 1 km downstream.

4.9 Key Variables

From the model runs carried out, 4 key variables have been identified which have the greatest influence the predicted concentrations in one or more parts of the river environment. They are river volumetric flow rate, duration of release, fraction of radionuclide sorbed to suspended sediments and water temperature.

4.9.1 River volumetric flow rate

The river volumetric flow rate is the most important environmental variable determining activity concentrations in water, fish and sediments following a radionuclide release. In general, the maximum and integrated activity concentrations in water and fish will be in inverse proportion to the river volumetric flow rate, for a given amount and duration of release.

4.9.2 Duration of release

The duration of release significantly influences peak radioactivity concentrations in water, $C_w(\text{max})$, particularly near to the point of discharge, where $C_w(\text{max})$ is in inverse proportion to the release duration. Duration of release does not influence the maximum activity concentrations in fish and sediments, nor does it influence the estimates of time integrated activity concentrations in water, fish and sediments.

4.9.3 Fraction of radioactivity sorbed to suspended sediments

The fraction of radioactivity sorbed to suspended particulates, f_p , can significantly affect radioactivity concentrations in water (dissolved phase), fish, and bed sediments. If 95% of the radioactivity is sorbed to suspended particulates, activity concentrations in water (dissolved phase) and fish are reduced by a factor of 20 or more. For the radionuclides studied, however, best estimates of f_p imply that the vast majority of radioactivity is in the dissolved phase ($f_p < 0.1$) for short duration releases (Table 8). Therefore, the model estimates of activity concentrations in dissolved water and fish always assume that all of the radioactivity is in the dissolved phase. The activity concentrations in bed sediments are clearly strongly dependent on the assumed fraction of activity in the suspended particulate phase. For calculating activity concentrations in bed sediments we have made conservative estimates of this parameter (Table 8).

4.9.4 Water temperature

The water temperature significantly influences the maximum activity concentration in fish, $C_f(\text{max})$ because fish feeding rates are much lower at lower water temperatures. At a water temperature of 17°C, $C_f(\text{max})$ is predicted to be approximately five times higher

than at a water temperature of 7°C for all radionuclides except radiostrontium. The water temperature has less effect on time integrated activity concentrations in fish, there being less than a factor of three decline in ΣC_f as temperature changes from 17°C to 7°C.

Although the model for radiostrontium uptake does not account for different water temperatures (it assumes a water temperature of 25°C) it can be expected that strontium uptake would also decline at lower temperatures as fish metabolic rates would be lower.

5. GENERALISING THE MODEL

The model developed is in principle applicable to all rivers in England and Wales. Most of the target variables may be estimated using only the volumetric flow rate of the river, and an estimate of the radionuclide particulate sorbed fraction. The estimates are therefore in principle applicable to other rivers. There are, however, some important limitations to the general application of the model:

1. The maximum concentration of a radionuclide in water downstream of the discharge is dependent upon the rate of longitudinal dispersion in the river. For the three sites studied, this dispersion rate was measured, but for other sites it is unlikely that there would be dye-tracer dispersion information available. A generalised model for peak dispersion is therefore required: this is given below in section 5.3.
2. In an accident situation, some indication of the time of travel of the plume down river would be required. This can be estimated, for the three sites studied, from the water velocity under different flow conditions. A generalised model for this is, however, required for other rivers and other sites. Such a model is presented below in sections 5.1 and 5.2, though this requires further validation before it is applied to other rivers.
3. The fraction of radionuclide absorbed to suspended particulates was estimated assuming water chemical conditions prevailing in the Thames which is not representative of all rivers. The conservative assumptions inherent in the values of this parameter in the model mean that it is unlikely that water chemistry changes would significantly affect model output. It would be valuable, however, to investigate this further for other water chemical conditions. In addition, the models assume that radionuclides are discharged in soluble form. This may not be true in some discharges.
4. The rate of uptake and concentration factors of radionuclides in fish also depend on water chemistry – the values used in the model assume water chemical conditions prevailing in the Thames – a hard water river of relatively high nutrient status. Fish-water concentration factors, and therefore uptake rates may be much higher (around one order of magnitude) for rivers of different water chemistry. In particular, radiostrontium concentration factors will be much higher in soft water rivers and radiocaesium concentration factors and uptake rates will be much higher in rivers of lower potassium concentration (Appendix D, Table D.7). There is also evidence that concentration factors of other radionuclides are strongly influenced by water chemistry, as illustrated in Table 11. Whilst this variation has to some extent been accounted for by the generally conservative estimates of CF used in the model (Table 11), some changes in the model parameter values may be required for other rivers. Such correction could certainly be required for radiostrontium and radiocaesium.

In the absence of site specific measurements of river flow velocity and plume dispersion, models developed by the US Geological Survey (USGS) (Jobson, 1997) may be used to estimate these parameters from easily available river characteristics. It should be noted that these relationships work best for medium and high volumetric flow

rates: they may be inaccurate when flow rates are much lower than the average for a given river.

Table 11 Concentration factors of freshwater fish (muscle) in waters of low and high mineral content (adapted from Blaylock, 1982)

Element	Water of low mineral content	Water of high mineral content	CF used in the model
Hydrogen	1	1	1
Carbon	5×10^4	5×10^3	2.2×10^4
Cobalt	1×10^3	50	300
Ruthenium	100	10	-
Iodine	50	5	40
Radium	150	10	-
Uranium	20	2	50
Plutonium	50	5	50

5.1 Predicting Time of Travel of the Plume Peak

The model of Jobson (1997) for predicting plume dispersion uses the following input parameters:

- D_a – the catchment area above the river reach, m^2 ;
- S – the average slope of the reach, $m\ m^{-1}$ (note this parameter is optional);
- v – the river flow velocity, $m\ s^{-1}$;
- Q – the volumetric flow rate at the reach, $m^3\ s^{-1}$;
- Q_a – the mean annual volumetric flow rate at the reach, $m^3\ s^{-1}$;
- g – the acceleration due to gravity, $9.8\ m\ s^{-2}$;
- v_l – the velocity of the plume leading edge, $m\ s^{-1}$;
- M_i – amount of radionuclide released, Bq;
- C_w – radioactivity concentration in water, $Bq\ l^{-1}$.
- x – distance downstream, m.

Defining the following dimensionless parameters:

$$v' = \frac{vD_a}{Q}, \quad D_a' = \frac{D_a^{1.25} \sqrt{g}}{Q_a}, \quad Q_a' = \frac{Q}{Q_a} \quad (1)$$

the average water velocity may be estimated using:

$$v = 0.094 + 0.0143(D_a')^{0.919} (Q_a')^{-0.469} S^{0.159} \frac{Q}{D_a} \quad (2)$$

If slope information is not available, the following relationship is recommended:

$$v = 0.020 + 0.051(D_a')^{0.821}(Q_a')^{-0.465} \frac{Q}{D_a} \quad (3)$$

The time of travel of the plume peak to a site x metres downstream is simply x/v . Relationships are also given (Jobson, 1997) for estimating of the maximum likely velocity of the peak.

5.2 Predicting Time of Travel of the Plume Leading Edge

The time of arrival of the plume leading edge was observed (Jobson, 1997) to be closely related to the peak travel time, being approximately a factor 0.89 of the peak travel time. Thus the velocity of the plume leading edge, v_l (m s^{-1}) is given by:

$$v_l = \frac{v}{0.89} \cdot \quad (4)$$

5.3 Predicting Maximum Activity Concentrations Downstream of the Discharge

For a radionuclide release of very short duration (effectively instantaneous), the maximum concentration in water, $C_w(\text{max})$ may be estimated using:

$$C_w^{\text{max}} = \frac{M_i 857}{Q \cdot 10^9} \left(\frac{x}{3600 \cdot v} \right)^{-0.76} \left(\frac{Q}{Q_a} \right)^{-0.079} \quad (5)$$

where M_i is the amount of radionuclide released (Bq), Q is measured in $\text{m}^3 \text{s}^{-1}$ and $C_w(\text{max})$ is measured in Bq l^{-1} . The factor 3600 converts the time of travel (x/v) from units of seconds to hours.

For releases of longer duration (hours, say), superposition of a number of very short duration releases may, in principle, be used to give the maximum downstream concentration.

5.4 Testing the USGS Model

The USGS model (Jobson, 1997) was tested against the results of the dye-tracer experiments carried out in this study. Tests were carried out first assuming no prior knowledge of river flow velocity, and secondly using estimates of river flow velocity which were estimated independently of the dye-tracer experiments. The results of the USGS model tests are shown in Figures 15 and 16 below.

In general the USGS model performs very well in estimating approximate times of travel and plume concentrations. Independent measurement of the flow velocity at a site significantly improves the model predictions.

5.4.1. Example USGS model output

Output from the USGS model is given for a site 1 km downstream of a 1 MBq release of very short duration (< 5 min) in two hypothetical rivers of volumetric flow rate $10 \text{ m}^3 \text{ s}^{-1}$, one slow-flowing, with velocity 0.1 m s^{-1} , the other faster flowing with velocity 1 m s^{-1} . The USGS predictions for these scenarios are shown in Table 12.

River Colne - USGS model "blind" test

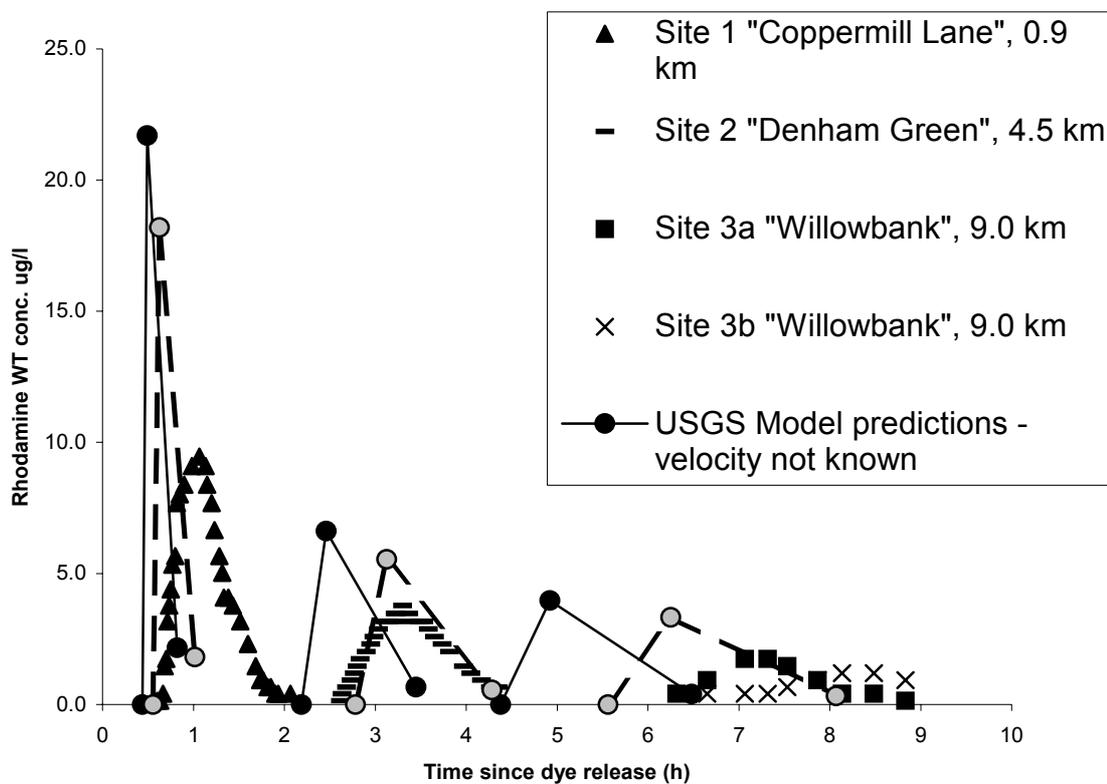
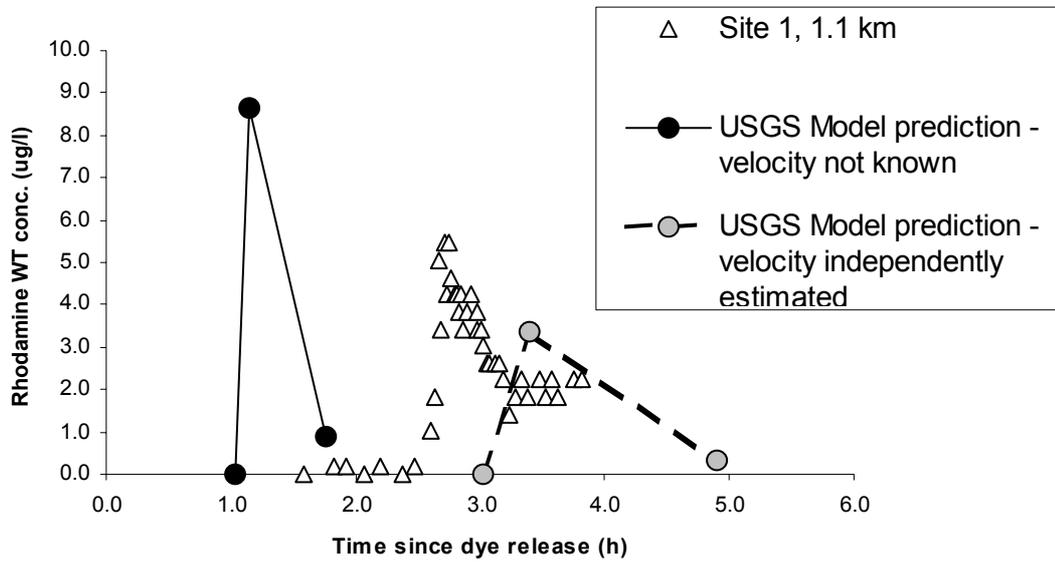


Figure 15 Test of USGS model against dye tracer data from the River Colne

(a) River Thames Pangbourne: model "blind" test

$Q = 10.5 \text{ m}^3 \text{ s}^{-1}$



(b) River Thames Pangbourne: model "blind" test

$Q = 25.5 \text{ m}^3 \text{ s}^{-1}$

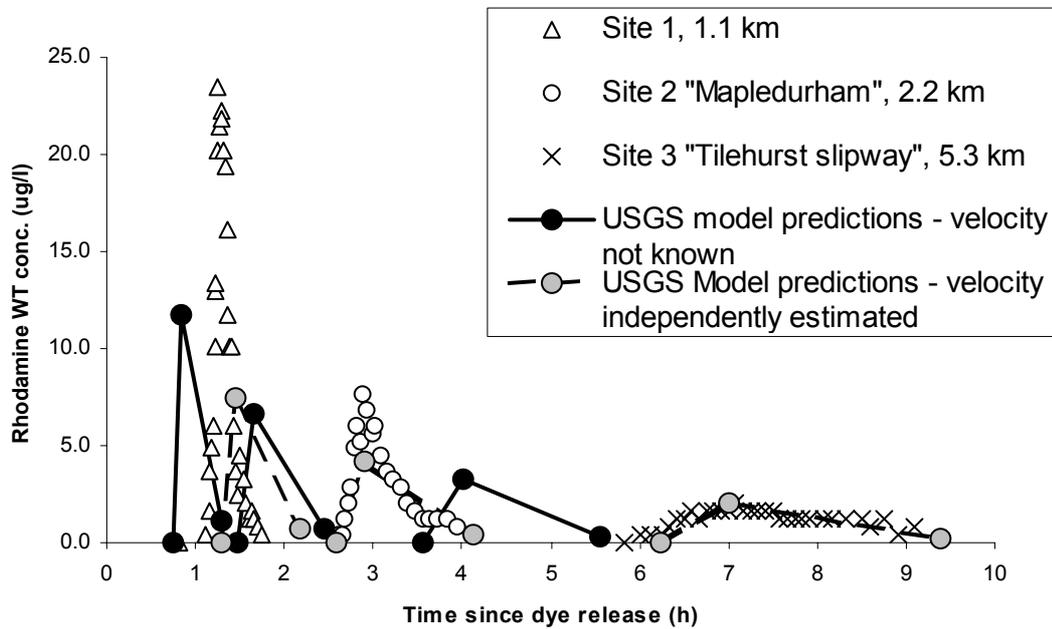


Figure 16 Test of USGS model against dye tracer data from the River Thames at two different flow velocities

Table 12 USGS model predictions for a site 1 km downstream of a 1 MBq release to hypothetical slow and fast-flowing rivers

	River Q = 10 m ³ s ⁻¹ , 1 km downstream of 1 MBq release	
	v = 0.1 m s ⁻¹	v = 1.0 m s ⁻¹
Time of arrival of peak	2 h47 min.	17 min.
Time of arrival of leading edge	2 h28 min.	15 min.
Maximum water concentration	0.039 Bq l ⁻¹	0.23 Bq l ⁻¹

5.5 Determining Input Data for Generalised Models

Methods of estimating key river characteristics at new sites for use in the generalised models are outlined below. It is emphasised, however, that further work is required to quantify uncertainty in these estimates.

Volumetric flow rate. May be estimated by extrapolation (on a catchment area weighting basis) from Environment Agency flow gauging stations on the river or on similar nearby rivers. Alternatively, rainfall-runoff models are available for estimating volumetric flow rate.

River flow velocity. May be measured directly by dye-tracer experiment, flow meters or time of travel of oranges. Alternatively, estimates may be made using volumetric flow rate and average channel cross section. It may be possible to estimate channel cross section from Environment Agency river channel charts.

River catchment area. The catchment area above the river reach may be estimated from maps, or in a GIS, using for example the CEH UK river network GIS.

River slope. The average slope may be estimated from topographic maps, or in a GIS, using, for example, the CEH UK river network GIS.

6. APPLICATION OF RESULTS TO OTHER RIVERS

The model detailed above can to a certain extent be generalised to other rivers, given the limitations discussed in Chapter 5 above. Below, the method of applying the model to other rivers, assuming a 1 MBq release, is given for each of the model target variables.

Maximum activity concentration in water at the discharge point. The cross-sectional average of the maximum activity concentration at the discharge point may be estimated using Figure 4. Input data required is the volumetric flow rate of the river. This maximum at the point of discharge will over-estimate maximum concentrations at sites downstream of the discharge point since the plume will disperse as it travels downstream.

Maximum activity concentration in water downstream of the discharge point. The model cannot directly be used to predict maximum activity concentrations in water downstream of the discharge point. If there is dye tracer data for the reach of interest, these can be analysed to determine coefficients and the simplified advection dispersion model (Appendix G) may be used to predict maximum concentrations in water. In the absence of dye tracer data, the USGS model (Chapter 5) should be used. For greatest accuracy, the flow velocity of the river should be independently estimated. Input data required is the catchment area above the river reach; the average slope of the reach (note this parameter is optional); the volumetric flow rate at the reach at the time of release, and the mean annual volumetric flow rate at the reach.

Time integrated activity concentration in water. The time integrated activity concentration in water may be estimated using Figure 9. Input data required is the volumetric flow rate of the river.

Maximum activity concentration in fish. The maximum activity concentration in fish may be estimated using Figure 10. Input data required is the volumetric flow rate of the river. Note, however, that application of Figure 10 to other rivers assumes that fish uptake parameters are the same as those in the rivers of the Thames catchment (see Chapter 5 note 4.).

Time integrated activity concentration in fish. The time integrated activity concentration in fish may be estimated using Figure 11. Input data required is the volumetric flow rate of the river. Note, however, that application of Figure 11 to other rivers assumes that fish uptake parameters are the same as those in the rivers of the Thames catchment (see Chapter 5 note 4.).

Maximum activity concentration in bed sediments. The maximum activity concentration in bed sediments may be estimated using Figure 12. Input data required is the volumetric flow rate of the river and the fraction of radioactivity sorbed to the solid phase (Table 8). Note that for radionuclides that are strongly adsorbed to sediments, for very slow flowing rivers ($<0.1 \text{ m s}^{-1}$), Figure 12 may over estimate maximum activity concentrations in sediments at downstream sites by more than the factor of 2 discussed in section 4.6, note 3.

Time integrated activity concentration in bed sediments. The time integrated activity concentration in bed sediments may be estimated using Figures 13, 14. Input data required is the volumetric flow rate of the river and the fraction of radioactivity sorbed to the solid phase (Table 8). Note that for radionuclides that are strongly adsorbed to sediments, for very slow flowing rivers ($<0.1 \text{ m s}^{-1}$), Figure 14 may over estimate maximum activity concentrations in sediments at downstream sites by more than the factor of 2 discussed in section 4.7, note 3.

7. SUMMARY AND CONCLUSIONS

A model was developed to assess the dispersion of radionuclides following short duration releases to stretches of the River Thames or its tributaries near 3 licenced nuclear sites. The model was used to estimate the following target variable:-

- Maximum activity concentration in water (total and dissolved phase);
- Integrated (one year time integral) activity concentration in water (total and dissolved phase);
- Maximum activity concentration in predatory fish;
- Integrated (one year time integral) activity concentration in predatory fish;
- Maximum activity concentration in bed sediments;
- Integrated (one year time integral) activity concentration in predatory fish.

Calculations were made for all of the radionuclides considered (Table 1) for the different release times from each site 5 min, 30 min, 3 h, 12 h, 24 h (see Table 2). Calculations were made for sites at the following distances downstream of the discharge points: 100 m, 300 m, 1000 m, 3000 m, 10000 m. The calculations were presented as graphs of radionuclide concentrations versus volumetric flow rate and distance for a nominal 1MBq radionuclide release.

It was found that the river **volumetric flow rate** is the most important environmental variable determining activity concentrations in water, fish and sediments following a radionuclide release. In general, the maximum and integrated activity concentrations in water and fish are in inverse proportion to the river volumetric flow rate, for a given amount and duration of release. The **duration of release** has little impact on integrated activity concentrations in water, fish and sediments, nor has it significant impact on maximum activity concentrations in fish and sediments. It does, however, significantly influence peak radioactivity concentrations in water, $C_w(\text{max})$, particularly at short distances from the discharge point, where $C_w(\text{max})$ is in inverse proportion to the release duration.

The **fraction of radioactivity sorbed to suspended particulates**, f_p , can significantly affect radioactivity concentrations in all three target variables. If 95% of the radioactivity is sorbed to suspended particulates, activity concentrations in water (dissolved phase) and fish are reduced by a factor of 20 or more. In the model, however, estimates of activity concentrations in dissolved water and fish always conservatively assume that the vast majority of radioactivity is in the dissolved phase.

The **water temperature** significantly influences the maximum activity concentration in fish, $C_f(\text{max})$ because fish feeding rates are much lower at lower water temperatures. At a water temperature of 17°C, $C_f(\text{max})$ is predicted to be approximately five times higher than at a water temperature of 7°C for all radionuclides except radiostrontium and tritium. The water temperature has less effect on time integrated activity concentrations in fish, there being less than a factor of three decline in ΣC_f as temperature changes from 17°C to 7°C.

Although the model for radiostrontium and tritium uptake does not account for different water temperatures (it assumes a water temperature of 25°C) it can be expected that

strontium uptake would also decline at lower temperatures as fish metabolic rates would be lower.

7.1 Uncertainties and Potential Further Research

1. This project has developed generalised models for extension of the work to other rivers and discharge scenarios. The models developed by the USGS (Jobson, 1997) are robust, easy to use and have a very strong empirical basis. It is believed therefore that these generalised models are fit-for purpose, but require further testing to assess their applicability to (generally smaller) UK rivers, and to determine the limits of their applicability. In particular, we recommend:
 - a. Further testing of the predictions of the USGS model against existing dye tracer measurements in UK rivers held by the Environment Agency and, if necessary, additional dye tracer experiments. In this study we have carried out a limited test of this model, however further testing is required for application to UK rivers, and to estimate uncertainties and the limits of model applicability (e.g. can the model be used for small streams or is it only applicable to relatively large rivers?).
 - b. Extension of the USGS model for non-instantaneous releases. Currently, the USGS model is not applicable to release times greater than a few minutes. It may, however, be developed for application to longer release times (relatively easily) using superposition of the model output for a number of instantaneous releases. Such a model could be implemented in an EXCEL spreadsheet;
 - c. Testing generic methods of estimating river flow velocities (particularly under low flow conditions) and assessing uncertainties in travel time estimates. It has been shown that predictions of the USGS model may be significantly improved by direct estimation of the river flow velocity. Simple methods of estimating flow velocity (without using expensive dye tracer experiments) need to be investigated.
2. A major uncertainty in the model predictions is the fraction of radionuclide absorbed to the solid phase. This uncertainty has been accounted for in the model predictions by making conservative assumptions of the value of this parameter. This may lead to significant over-estimates of doses, though the review of K_d values (Table 8 and Appendix D) suggest that this is more likely in the case of activity concentrations in sediments, than in estimates for water and fish. Similarly, resuspension and transport of bed sediments is uncertain, so conservative assumptions have been made: again these are likely to have the most influence on the sediment phase.

Given the inherent difficulties in estimating particulate-sorbed fractions of radionuclides, and resuspension and transport of bed sediments, it is not clear that further research would necessarily reduce model uncertainties. This is particularly true for application of a generalised model where there may be little site specific information. Further research to test the model assumptions and better assess the variability of sediment sorption and resuspension parameters would, however, significantly strengthen the empirical basis of the model.

3. It is further assumed that all radioactivity is in the dissolved form at the time of release. This may not always be true: for example, in some cases discharged radioisotopes may be bound to organic molecules. This could potentially affect both particle sorption and uptake through the aquatic food chain. Further research on chemical forms of discharged radionuclides, and their influence on environmental mobility, would be valuable.
4. For the particular reaches studied here, there is some uncertainty regarding the volumetric flow rate of the Mill Stream. Radionuclide concentrations in fish, water and sediments in the Mill Stream are likely to be up to one order of magnitude higher than those in the main river, at least for temporary low flow periods. Although there is only a short distance between the discharge and the confluence of the Mill Stream and the main river, it would be useful to better quantify the Mill Stream volumetric flow rate as a function of that of the whole river. It is noted that at the minimum volumetric flow in the Mill Stream, the volumetric flow rate of the pipeline represents a significant proportion (up to 25%) of that of the Mill Stream.
5. The general applicability of the model for radionuclides in fish would be strengthened by estimates of fish uptake parameters for river chemical conditions which are significantly different to those of the Thames (particularly for low nutrient, low mineral content rivers).

7.2 Further Technical Developments

If the Environment Agency wishes to strengthen its scientific basis for response to accidental river pollution (radioactive and non-radioactive), it would benefit from an integrated pollution dispersion model. A GIS-based river network model could be linked with a pollution dispersion model to provide estimates of key model parameters such as water velocity and volumetric flow rate for all UK rivers. Such a system is feasible on the basis of current models such as those discussed above. It would be capable of giving rapid estimates of pollutant time of travel, duration of an event, and maximum concentrations downstream of the source. A generalised model such as this would give useful predictions, requiring only the amount of pollutant released and the release duration as user input data.

REFERENCES

- AWE (1999) *Report following the inspection of Pangbourne pipeline discharge system in the river Thames*. AWE report, ref: DSE07/B/L/RP/A12N/13.06.89.04/99-26.
- Baudin J P and Fritsch A F (1989) *Relative contributions of food and water in the accumulation of ⁶⁰Co by a freshwater fish*. Water Research 23 No. 7 817-823.
- Bencala K E and Walters R A (1983) *Simulation of solute transport in a mountain pool-and-riffle stream: A transient storage model*. Water Resources Research 19, 718-724.
- Blaylock, B.G. (1982) *Radionuclide data bases available for bioaccumulation factors for freshwater biota*. Nuclear Safety 23, 427-438.
- Chowdhury M J and Blust R (2001) *A mechanistic model for the uptake of waterborne strontium in the common carp (Cyprinus Carpio L)*. Environ. Sci. Technol. 35, 669-675.
- Denyer S (2001) *Short term uptake of phosphorus and zinc in pike (Esox lucius) in the Grand Union Canal*. Nycomed Amersham Health Physics and Safety Technical Note 00/15.
- Elliott J M (1975a) *Number of meals per day, maximum weight of food consumed per day and maximum rate of feeding of brown trout, Salmo Trutta*. Freshwater Biology Vol. 5, 287-303.
- Elliott J M (1975b) *Growth rate of brown trout (Salmo Trutta) fed on maximum rations*. J. Animal Ecology Vol. 44, 805-821.
- Evans D W, Alberts J J and Clark R A (1983) *Reversible ion-exchange fixation of cesium-137 leading to mobilization from reservoir sediments*. Geochimica et Cosmochimica Acta 47, pp. 1041-1049.
- Gharbi S and Verrette J-L (1998) *Relation between longitudinal and transversal mixing coefficients in natural streams*. Journal of Hydraulic Research, 38 43-53 (in French).
- Hesslein R H (1987) *Whole-lake radiotracer movement in fertilized lake basins*. Canadian Journal of Fisheries and Aquatic Sciences 44 (suppl. 1), 74-82.
- House W A, Farr I S, Orr D R and Welton J S (1989) *The interaction between pesticides and particulates in rivers*. Institute of Freshwater Ecology.
- House W A, Denison F H, Smith J T and Armitage P D (1995) *An investigation into the effects of water velocity on inorganic phosphorus influx to a sediment*. Journal of Environmental Pollution. 89 No. 3, 263-271.
- Jobson H E (1997) *Predicting travel time and dispersion in rivers and streams*. J. Hydraulic Engineering, Vol. 123, 971-978.

- Kryshchuk I I and Ryazantsev (2000) *Ecological safety of atomic energy complexes in Russia*. Izdat, Moscow ISBN 5-86656-097-6, in Russian.
- Liu H, Asce M and Cheng A H D (1980) *Modified Fickian model for predicting dispersion*. Journal of the Hydraulics Division, Vol. 106, No. 6, 1021-1040.
- Neal C and Robson A J (2000) *A summary of river water quality data collected within the LOIS study: core data for Eastern UK rivers draining to the North Sea*. Science of the Total Environment vols. 251, 252, pp 585-665.
- Partheniades E and Asce M (1965) *Erosion and deposition of cohesive soils*. Journal of the Hydraulics Division, Vol. 91, 105 – 139.
- Rowan D J and Rasmussen J B (1994) *Bioaccumulation of radiocaesium by fish: the influence of physicochemical factors and trophic structure*. Canadian Journal of Fisheries and Aquatic Sciences 51, 2388-2410.
- Simmonds J R, Lawson G and Mayall A (1995) *Methodology for assessing the radiological consequences of routine releases of radionuclides to the environment*. Report EUR15760 EN, European Commission, Luxembourg.
- Smith J T and Comans R N J (1996) *Modelling the diffusive transport and remobilisation of Cs-137 in sediments: the effects of sorption kinetics and reversibility*. Geochimica et Cosmochimica Acta 60 No. 6, 995-1004.
- Smith J T and Elder D G (1999) *A comparison of different methods of characterising radionuclide activity-depth profiles in soils*. European Journal of Soil Science, 50, 295-307.
- Smith J T, Comans R N J and Elder D G (1999) *Radiocaesium removal from European lakes and reservoirs: key processes determined from 16 Chernobyl contaminated lakes*. Water Research 33(18) 3762-3774.
- Smith J T, Comans R N J, Ireland D G, Nolan L and Hilton J (2000) *Experimental and in situ study of radiocaesium transfer across the sediment/water interface and mobility in lake sediments*. Applied Geochemistry, 15 (6), 831-846.
- Smith J T, Kudelsky A V, Ryabov I N, Hadderingh R H (2000b) *Radiocaesium concentration factors of Chernobyl-contaminated fish: a study of the influence of potassium, and "blind" testing of a previously developed model*. Journal of Environmental Radioactivity, 48, 359-369.
- Stonell G (1999) *Methodology for assessing radiological impacts of Harwell liquid effluents: discharges at Sutton Courtenay*. UKAEA report HSED/LDA/DPUD/1.
- Titley J G, Carey A D, Crockett G M, Ham G J, Harvey M P, Mobbs S F, Tournette C, Penfold J S S, Wilkins B T (2000) *Investigation of the Sources and Fate of Radioactive Discharges to Public Sewers*. Environment Agency report TR P288.

Vinogradov A P (1953) *The elementary chemical composition of marine organisms*. Sears Foundation for Marine Research, Memoir Number II, Yale University.

Young P C and Wallis S G (1993) *Solute transport and dispersion in channels*. In: Channel network hydrology, K. Beven and M J Kirkby (eds.), Wiley, Chichester.

Won Seo I., Asce M. and Sung Cheong T (1998) *Predicting longitudinal dispersion coefficient in natural streams*. Journal of Hydraulic Engineering, 124 No. 1, 25-32.

APPENDIX A: RELEASE SCENARIOS

A1 Releases of liquid effluents from AWE Aldermaston

There are currently three aqueous disposal routes from Aldermaston: to the Thames at Pangbourne (the “Aldermaston pipeline”), to Silchester sewage treatment works (STW), and to Aldermaston stream. The latter discharge is a continuous release of tritiated groundwater. Only the discharge to the Thames at Pangbourne via the Aldermaston pipeline will be considered in this project. The median volumetric flow rate of the Thames at Reading in 2000 was $39.2 \text{ m}^3 \text{ s}^{-1}$. The volumetric flow rate is not expected to change significantly between Pangbourne and Reading.

Radioactive liquid effluents generated on site are collected at the Radioactive Effluent Treatment Plant. Following treatment they are discharged to the Thames via the Aldermaston pipeline. The pipeline is 20 km long and discharges underwater directly to the Thames. The river forms a single channel at this point. Typically, two discharges per week are made, each of volume 150 m^3 . Because of the length of the pipeline (20 km), each discharge takes about 18 h to pass through it giving an average volumetric flow rate of the discharge of approximately $2.2 \times 10^{-3} \text{ m}^3 \text{ s}^{-1}$, approximately 0.025% of the minimum volumetric flow rate of the river. Because of the length of the pipeline, AWE Aldermaston consider it to be impossible that any release could be of duration less than 18 hours. Currently, discharges of Pu-241, H-3, total alpha emitters, and total beta emitters are authorised by the Environment Agency via this route.

A field visit to the discharge site has been made, and the discharge point identified. The discharge occurs via two parallel pipelines lying on the river bed (AWE, 1999). Each pipe has six nozzles of 12 mm diameter, spaced at approximately 2.45 m intervals. The pipelines stretch approximately 30 m, half way across the 60 metre wide river. Discharges are made from each pipe alternately. Build up of silt over the nozzles of between 15-76 cm was observed during a diving study commissioned by AWE, though the discharge rate was not adversely affected by the silt build up (AWE, 1999). Fresh water is now being pumped through the pipeline once per week in order to keep the nozzles clear and prevent silt build up.

A2 Releases of liquid effluent from UKAEA Harwell

There are currently two liquid effluent discharge routes from Harwell, to the Thames at Sutton Courtney via a pipeline, and to Lydebank Brook (surface water runoff). Authorised discharges via surface water runoff (primarily tritiated water) to Lydebank Brook are most appropriately modelled as a continuous rather than a short-term release and therefore will not be considered in this project. Therefore, only the discharge to the Thames at Sutton Courtney will be considered in this project. Currently, discharges of H-3, Cs-137, Co-60, total alpha emitters, and total beta emitters are authorised by the Environment Agency via this route. Liquid effluents arising from the active area of the site are routed to the Liquid Effluent Treatment Plant (LETP). Liquid effluents from all other areas of the site (known as trade wastes) are routed to holding tanks for monitoring prior to discharge. If necessary the trade waste effluents in the holding tanks can be diverted to the LETP for treatment. Effluents from the LETP and trade wastes from the holding tanks are discharged via a pipeline to the River Thames at Sutton Courtenay. Two types of tank are emptied into the pipeline, of volume 500 m^3 for low level effluent and of volume 1800 m^3 for trade waste. The rate of tank

emptying is approximately $0.083 - 0.125 \text{ m}^3 \text{ s}^{-1}$, and the travel time of effluent through the pipeline is approximately 35-40 minutes.

At the point of discharge, the main Thames river is divided into three channels, a canal (the Culham Cut), Sutton Pools, and the Mill Stream to the south of Sutton Pools. The median volumetric flow rate of the Thames at Sutton Courtney in 2000 was $27.9 \text{ m}^3 \text{ s}^{-1}$. The flow rate of the canal is negligible in comparison to that of the main river, and can be ignored. The discharge of liquid effluent from UKAEA Harwell is via nozzles in a pipeline which stretches underneath a weir, all the way across the Mill Stream channel to the south of Sutton Pools. Before discharges are made, UKAEA ensures that there is at least 76 mm head of water over the weir, which equates to a minimum flow in the Mill Stream of $0.44 \text{ m}^3 \text{ s}^{-1}$ (Stonell, 1999). This is approximately 7% of the total river flow under low flow conditions (Table 5). At the minimum volumetric flow in the Mill Stream, the volumetric flow rate of the pipeline represents a significant proportion (up to 25%) of the Mill Stream volumetric flow.

The Mill Stream and Sutton Pools branches rejoin approximately 100 m below the discharge.

A3 Releases from Amersham

There is currently one aqueous disposal route from Amersham, to the Grand Union Canal (GUC) via Maple Lodge sewage treatment works (STW). Active discharges are routed through final discharge tanks: 6 of volume 54.6 m^3 and 4 of volume 27.3 m^3 . The tanks take approximately four hours to discharge.

At the STW discharge point, the GUC and river Colne run parallel, with approximately 80% of the river flow going down the GUC. The combined volumetric flow rate of the GUC/Colne at Denham (ca. 5 km downstream of Maple Lodge STW) in 2000 had median value $4.35 \text{ m}^3 \text{ s}^{-1}$. The discharge from Maple Lodge is into the GUC, with a storm overflow into the Colne. The GUC joins the river Colne approximately 1.3 km downstream of the Maple Lodge discharge point. The Colne enters the Thames north west of Staines, which is approximately 25 km downstream of Maple Lodge. The Thames is tidal up to Teddington Lock, which is approximately 25 km downstream of where the Colne enters the Thames. The nearest water supply intake to the discharge point is on the river Thames, more than 25 km downstream of Maple Lodge STW.

A4 Radionuclides to be considered and release scenario

Radionuclides authorised for release from the three sites are shown in Table A1. The model release scenario will assume a nominal 1 MBq release of each of the radionuclides at the point of discharge to the river. For all three sites, releases for the following periods will be considered: 10 min, 30 min, 1 h, 12 h, 24 h. For each radionuclide, therefore, the total release of 1 MBq will be divided by the release time to give the rate of release over each release period. For Amersham, mixing of the release as it flows through Maple Lodge STW will also be modelled to give estimates of the input function to the river Colne.

Table A.1 List of radionuclides considered and their half-lives

Radionuclide	Half-life	Site potentially discharging
H-3	12.3 yrs	Amersham, Harwell, Aldermaston
C-14	5730 yrs	Amersham
P-32	14.3 days	Amersham
Co-60	5.27 yrs	Amersham, Harwell
Zn-65	244.3 days	Amersham, Harwell
Sr-89	53 days	Amersham, Harwell
Sr-90	28.8 yrs	Amersham, Harwell
I-125	59.4 days	Amersham, Harwell
I-131	8.05 days	Amersham, Harwell
Cs-134	2.065 yrs	Amersham, Harwell
Cs-137	30.2 yrs	Amersham, Harwell
Pu-238	87.7 yrs	Amersham, Harwell, Aldermaston
Pu-239	2.4×10^4 yrs	Amersham, Harwell, Aldermaston
Pu-240	6.5×10^3 yrs	Amersham, Harwell, Aldermaston
Am-241	432.2 yrs	Amersham, Harwell, Aldermaston
U-234	2.45×10^5 yrs	Amersham, Harwell, Aldermaston
U-235	7.08×10^8 yrs	Amersham, Harwell, Aldermaston
U-238	4.47×10^9 yrs	Amersham, Harwell, Aldermaston

A5 Mixing of release from Amersham through Maple Lodge STW

The typical transfer time of effluent through Maple Lodge STW is 8 hours (Colin Delve, Thames Water, pers. comm.) and the average effluent inflow is $120 \times 10^3 \text{ m}^3 \text{ d}^{-1}$. Treating the STW as a completely mixed reactor of water residence time 8 hours, the concentration exiting the STW, C_e (Bq m^{-3}) at time t resulting from an inflow of radionuclide of concentration C_i (Bq m^{-3}) of duration T (hr) is given by:

$$C_e = C_i(1 - e^{-t/T_w}) \quad 0 < t < T$$

$$C_e = C_i(e^{-(t-T)/T_w} - e^{-t/T_w}) \quad t > T \quad (\text{A1})$$

where T_w is the residence time of the mixed reactor. Figure A.1 shows an example of the estimated exit concentration resulting from a continuous inflow of 1 MBq of activity over a 12 hour period, at an effluent inflow rate equal to the average for the STW, $120 \times 10^3 \text{ m}^3 \text{ dy}^{-1}$. The effects of mixing in the STW may be approximated by altering the timing and duration of release to account for the different river input terms resulting from given releases at the Amersham site (Table A.2)

Table A.2 Hypothetical release durations from Amersham and resulting release duration to the river

Duration of hypothetical release from Amersham	Estimated duration of discharge from Maple Lodge STW to GUC	Approximate time lag between release from Amersham and inflow to GUC
5 minute	8 hours	
30 minute	8 hours	
3 hours	8 hours	> 8 hours
12 hours	12 hours	
24 hours	24 hours	

A6 Loss of radionuclides during treatment at Maple Lodge STW

For the purpose of this study, we will assume that all radionuclides remain in the solution and are not lost to sludge as they pass through Maple Lodge STW. It is, however, noted that this may lead to over-estimation of river activity concentrations resulting from discharges of certain radionuclides from Amersham. This is likely to be particularly important for C-14 and P-32 because of carbon and phosphorus removal to solids during the sewage treatment process. Removal of radionuclides to sludge during sewage treatment has been estimated by Titley et al. (2000).

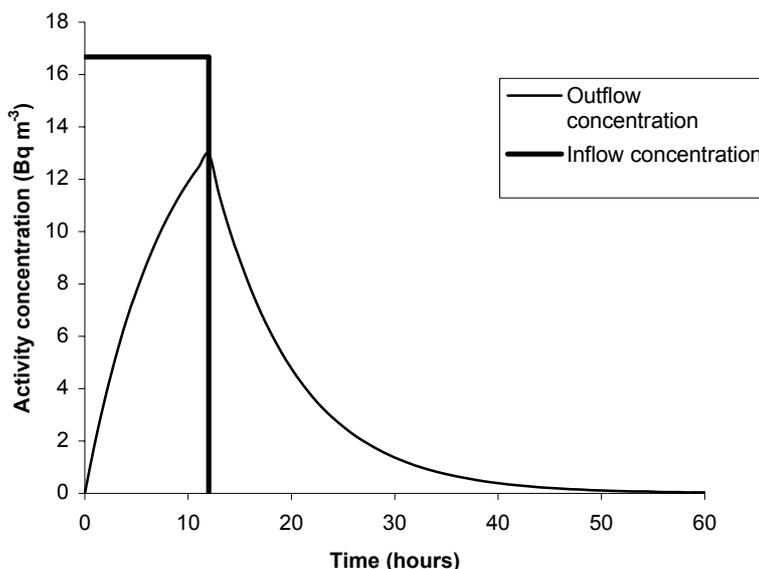


Figure A.1 Estimated outflow activity concentration from Maple Lodge STW following a total input of 1 MBq over a period of 12 hours

APPENDIX B: MODEL SELECTION

The aim of the physical transport modelling (ie modelling movement within the non-biological parts of the systems) is to predict activity concentrations of the above radionuclides in river water and sediments up to 10 km downstream of the discharge points, and for time periods up to 1 year after release. In the initial phase of the release (ie on time scales less than, or of the order of, the travel time along the 10 km stretch), modelling of the advection-dispersion of the plume and transfer of radioactivity to sediments is required. On longer time scales than this, detailed modelling of advection and dispersion in the river is not required, but transport of contaminated bottom sediments, and remobilisation of radioactivity to the water column must be considered.

B1 Physical transport modelling – transport of the release plume

One-dimensional compartmental models. In compartmental (or “box”) models, the river is modelled by a series of appropriately sized boxes representing different river stretches. These models are simple, require few, easily available, input parameters, and can relatively easily include transfers of the pollutant to and from sediments, and transfers to biota. They are appropriate for modelling slowly-varying discharges to a river, and to model behaviour of a pollutant at long-time scales after a pollution incident, i.e. in cases when short-term transport processes may be ignored. They are not appropriate for modelling short-term transport of a plume down a river since their spatial and temporal resolution is too low.

One-dimensional advection-dispersion models. In 1-D advection-dispersion (ADE) type models, the concentration of pollutant is averaged over the river cross section and longitudinal (down river) advection and dispersion of the averaged concentration estimated. Typically, advection of the discharge plume is proportional to mean flow rate and dispersion is a function of mean flow rate and, potentially, river and bed sediment characteristics. The 1-D advection-dispersion approach offers greater resolution than compartmental models without significant increase in model parameterisation requirements. In addition, there is a large body of measurements of ADE model parameters for a wide range of rivers available in the literature (for example, data summarised in Young and Wallis, 1993). This is a considerable advantage when attempting to make predictions of pollutant transport in systems for which tracer data for model calibration is not available.

In common with all 1-D models, the 1-D ADE model assumes complete, instantaneous, cross-sectional mixing of a point source tracer. Pollutant discharges often, however take some time (distance downstream) before they are completely mixed across the river. Thus for some distance downstream of the discharge, there will be significant differences in concentrations across the river. In other words, the mean cross sectional concentration predicted by the ADE will underestimate concentrations within the plume, and over-estimate concentrations outside the plume, even though the cross sectional average may be correctly predicted. In addition, tracer studies (e.g. Liu et al. 1980) show that dispersion is not constant with distance from release, but increases with distance. This is primarily a result of incomplete cross-sectional mixing of the tracer at injection. Methods are available, however, for predicting such effects of transverse mixing of the source.

In the advection-dispersion model, incorporation of rates of transfer to and from bottom sediments and biota is slightly more mathematically complex than for compartmental models. In general, computing time requirements of the ADE model are high when (as in most cases in which we are interested) numerical evaluation is required.

B2 Skewness of tracer concentration profiles

A further limitation of the ADE model is that in many cases observed tracer concentration profiles are skewed. Skewness is expected in measurements of tracer concentrations at different times passing a point of given distance from source, even when they are distributed spatially according to the ADE. This is because the tracer at the trailing edge of the profile (at a given spatial sampling point) has had a longer time to diffuse than that in the leading edge. However, even when this is accounted for, the tracer profile often remains skewed (examples of non-skewed and skewed distributions are shown in Figure B.1). Approximately 9 out of the 26 tracer profiles we studied showed evidence of skewed distributions, though this did not significantly affect the model fits. Two of the more seriously skewed profiles are shown in Figure B.1.

Skewness of tracer profiles may be caused by a number of factors including the way in which the tracer is injected into the river, the number of slow flowing “dead zones” in the river reach, and the presence of weirs in the reach. A literature search has not thus far found any quantitative criteria for determining the conditions under which profiles are likely to be skewed. Skewness of the observed tracer profiles, however, is not large (Figure B.1) and will not significantly affect the model target variables (i.e. maximum and integrated concentrations in water, fish and sediments).

One-dimensional dead zone models. The observed “skewness” of many tracer concentration profiles (Figure B.1) is often attributed to the effects of zones of slow-moving eddying water (“dead zones”). This skewness may be modelled (fitted) either by modifying the ADE model (e.g. Bencala and Walters, 1983), or by using an alternative fitting function. The Aggregated Dead Zone (ADZ) model (e.g. Young and Wallis, 1993) uses a transfer function approach to model the change in concentration of a solute over a whole river reach. It has been argued (Young and Wallis, 1993) that the ADZ model better fits tracer concentration profiles in rivers than the ADE. It is also numerically much more efficient than numerical solutions to the ADE.

In common with all 1-D models, the ADZ model does not account for incomplete cross-sectional mixing of a point source tracer. The ADZ also requires calibration using a tracer experiment – there are at present no generally applicable relationships between ADZ model parameters and measurable river characteristics such as water velocity and volumetric flow rate. In addition, the ADZ only gives a transfer function between input and output points. In other words, it only gives predictions for the points downstream of the release at which tracer measurements have been made. Interpolation between those points would therefore be required for the purposes of this project.

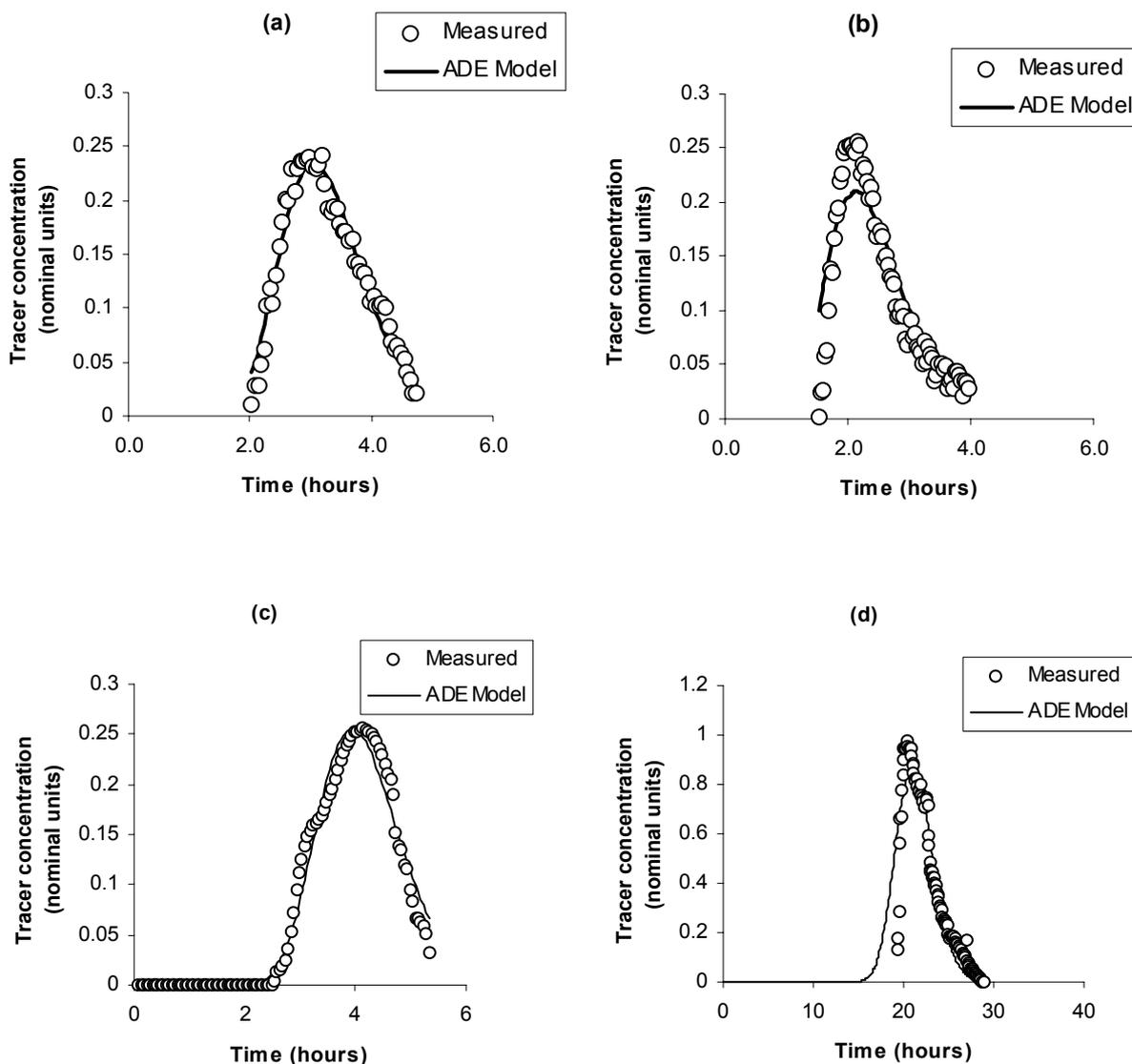


Figure B.1 Examples of least-squares fits of the advection-dispersion equation (ADE) to measurements of Rhodamine WT dye tracer concentrations in the river Thames (data supplied by the Environment Agency). The distributions on the left (a) UTHH16 and (c) UTHH18 are fitted well by the ADE, the ones on the right (b) UTHH17 and (d) UTHL19 are skewed (showing a “tail” in the tracer distribution at long times after the peak has passed) and are less well fitted by the ADE

Two- and 3-dimensional models. In 2-D and 3-D numerical models, pollutant concentrations are predicted at each depth and location in the river (3-D model) or are depth-averaged for each location (2-D model). They can provide accurate estimates of the dispersion of a plume both across the river from a discharge point and longitudinally. They therefore account explicitly for cross sectional mixing of the tracer from a point source. However, 2- and 3-D river models are extremely data intensive and therefore difficult to calibrate for such large stretches of river (up to 10 km) as will be considered in this project. They also require considerable computer time and resources to run over long distances and times.

B3 Transverse mixing of pollutants in rivers

Mixing of a plume across the river section occurs at rate determined by the transverse dispersion coefficient, D_t and increases with time as the plume moves downstream. The transverse mixing may be simulated using a solution to the diffusion equation for diffusion between two boundaries at $y = 0$ and $y = W$, where W is the river width and y is the distance across the river. A solution for these boundary conditions is given in Crank (1975) (his Equation 2.17):

$$C = \frac{1}{2} C_0 \sum_{n=-\infty}^{\infty} \left(\operatorname{erf} \frac{h + 2nW - y}{2\sqrt{D_t t}} + \operatorname{erf} \frac{h - 2nW + y}{2\sqrt{D_t t}} \right) \quad (\text{B1})$$

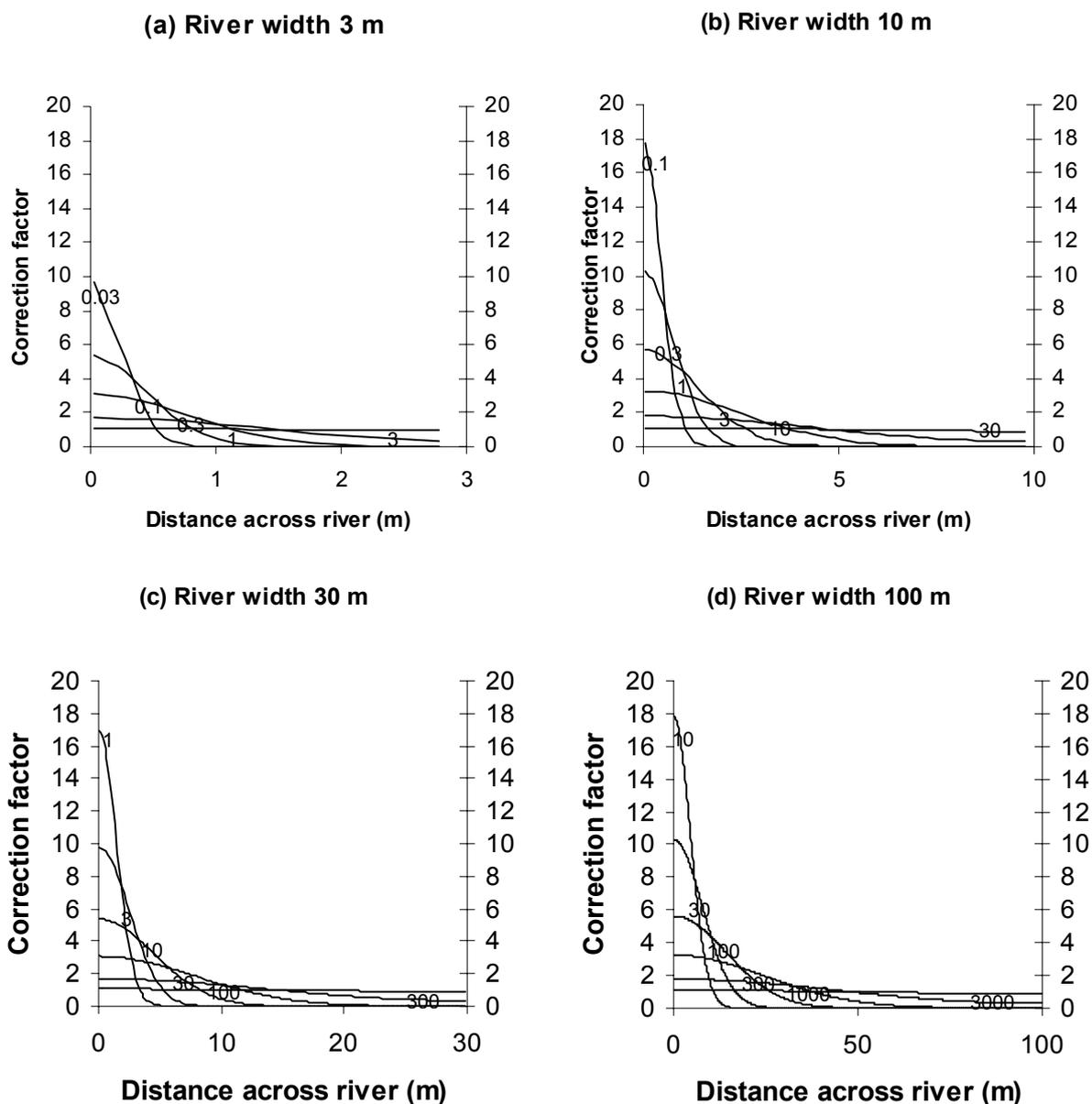
where h is one half the width of the pollution source. According to Equation (B1) transverse mixing increases as the product $D_t t$ increases, where t is the time of travel ($t = x/v$) of the plume. Equation (B1) was used to generate transverse mixing profiles for rivers of different widths and for different values of the product $D_t t$ (Figure B.2). In Figure B.2 it is assumed that the inflow of pollutant occurs at the side of the river (*i.e.* h is small). This is not the case for the particular sites we are studying, where the discharge is made some way across the river. Transverse mixing will therefore be more rapid in the sites we are studying than estimated in Figure B.2 (and see Table 7).

The graphs in Figure B.2 are normalised to a unit mean cross sectional activity concentration (*i.e.* for complete transverse mixing the activity concentration is 1 Bq m^{-3} at each distance y across the river). The graphs therefore give correction factors for estimating the activity concentration at each point y for a given mean cross sectional activity concentration as determined from the longitudinal dispersion model. This is done by calculating the value of $D_t t$ then, for the particular river width, looking up the value of the correction factor for the transverse distance y . The average cross-sectional activity concentration is then multiplied by the correction factor to give the estimated activity concentration at point y .

The value of the transverse dispersion coefficient, D_t , may be estimated using the empirical model of Gharbi and Verrette (1998) which relates the transverse dispersion coefficient to the coefficient of longitudinal dispersion, D :

$$D_t = 0.00035 \left[\frac{(Q/d)^{1.75} (W/d)^{0.25}}{D^{0.75}} \right] + 0.0005 \quad (\text{B2})$$

where Q is the volumetric flow rate ($\text{m}^3 \text{ s}^{-1}$), d is the average depth (m), W is the average width (m). Two typographical errors found in the paper of Gharbi and Verrette have been corrected (S. Gharbi, pers. comm.), so the above Equation (B2) is slightly different to that given in Gharbi and Verrette (1998) (their Equation 13).



Reach	Sutton Courtney			Pangbourne			GUC/Colne		
	Low	Medium	High	Low	Medium	High	Low	Medium	High
Vol Disch									
$D_t \text{ m}^2 \text{ s}^{-1}$	0.00645	0.007714	0.006609	0.012689	0.05566	0.067561	0.003018	0.002731	0.002262
$V \text{ m s}^{-1}$	0.1	0.29	0.71	0.08	0.29	0.83	0.16	0.23	0.38
Distance m	Estimated values of $D_t t$								
100	6.45	2.66	0.93	15.86	19.19	8.14	1.89	1.19	0.60
300	19.35	7.98	2.79	47.59	57.58	24.42	5.66	3.56	1.79
1000	64.50	26.60	9.31	158.62	191.93	81.40	18.87	11.88	5.95
3000	193.51	79.80	27.93	475.85	575.80	244.20	56.60	35.63	17.86
10000	645.04	265.98	93.09	1586.18	1919.32	813.99	188.65	118.76	59.53

Figure B.2 Transverse mixing correction factors for a point source release at the side of rivers of different width, for different values of $D_t t$. Values of $D_t t$ for the three study reaches are given for illustrative purposes. Actual transverse mixing at these sites is given in Table 7

For illustrative purposes, Figure B.2 shows estimates of the product D_{it} at different volumetric flow rates and at different distances downstream of the three sites being studied here. The estimates indicate that for a pollutant released at the side of a wide river such as the Thames, transverse mixing may not occur until the plume has travelled several kilometres downstream. They also indicate that the distance downstream at which complete transverse mixing occurs appears to increase with increasing volumetric flow rate.

It is emphasized that there are many uncertainties in estimating transverse mixing rates from limited field data, but Equations (B1, B2) and Figure B.2 do indicate patterns of transverse mixing in rivers. Transverse mixing at the three sites studied here is given in Table 7.

B4 Choice of plume transport model

For the purpose of this project we have decided to use the 1-D ADE model. Compartmental models have too low spatial resolution for our purpose of modelling transport of the plume resulting from short-term releases. Two- and 3-dimensional models are much too difficult and expensive to calibrate for the required purpose of modelling large river stretches, and for development of generic modelling approaches for new sites. It should also be noted that the releases from all three sites are expected to be relatively rapidly mixed across the river channel. For Harwell, the discharge is spread across the channel and for Aldermaston, the release pipes stretch half way across the channel. Visible material in the release from Maple Lodge was observed to be relatively rapidly distributed (within 100 m) across the stream. Therefore, the assumption of cross-sectional averaging of the release, inherent in the 1-D models is not expected to lead to large deviations from reality, even relatively close to the discharge points. Transverse mixing for the three sites is given in Table 7 of the main text.

The ADZ model has some advantages over the ADE: it is computationally simpler, and it better fits skewed concentration distributions. The ADZ, however, models river reaches, whereas (at short times after release) the intention here is to model tracer concentrations at any distance downstream, which is more easily done using the ADE. In addition, the ADE much better fits the second objective of developing a generic model for dispersion in rivers. Because of the large quantity of existing data on dispersion parameters in rivers, empirical relationships have been developed between ADE parameters and easily measurable river characteristics (Won Seo et al. 1998). Such relationships allow estimation of ADE parameters for new sites and new rivers in cases where dye tracer measurements are not available.

B5 Physical transport modelling – transfers to sediments

Plume transport models, as discussed above, describe the movement of substances in solution down the river. Modelling non-conservative substances (i.e. material associated with sediments) requires estimation of transfers of the substance to suspended and bottom sediments. Uptake of radionuclides by suspended sediments is a dynamic process, often involving a number of separate sorption, speciation and precipitation processes. Although models for the kinetics of radionuclide uptake by suspended sediments are available, they are specific to certain environmental and chemical conditions, and to certain radionuclides. They are therefore inappropriate for a study such as this which requires simple, robust predictive estimates of radionuclide partitioning for a range of radionuclides. The distribution coefficient (K_d , $l\text{ kg}^{-1}$) approach will therefore be used. This assumes an equilibrium ratio between activity concentration on suspended solids to that in solution:

$$K_d = \frac{\text{Activity Concentration per unit mass of solids (Bq kg}^{-1}\text{)}}{\text{Activity Concentration per unit volume of solution (Bq l}^{-1}\text{)}} \quad (\text{B3})$$

A literature study of K_d values for the selected radionuclides has been carried out. The results of this study are reported in Annex 1, which includes a full discussion of the assumptions behind K_d 's, and their application to river transport modelling. Table B.1 gives a summary list of recommended parameters for the different radionuclides. The K_d approach may be used despite known time-dependent changes in sorption of the radionuclide since we will here use best estimate K_d 's selected from measurements over the timescale appropriate for this study (i.e. approximately 2 days). Given the known large range in possible K_d values, we will, in addition, account for potential errors by giving output for a range of K_d values (using variation in the particulate sorbed fraction, see below).

The estimate for C-14 given in Table B1 assumes a maximum K_d of 10^4 , as observed for a range of organic substances (House et al., 1989). The estimate for P-32 was made using measurements of Total Dissolved Phosphorus and Particulate Phosphorus in the Thames (Neal and Robson, 2000). The K_d values estimated for other radionuclides are from a literature review carried out within this project. The K_d values given in Table B1 are best estimates of the likely sorption K_d 's based on the range of values available in the literature and a knowledge of the general chemistry of the river Thames, and also include a range of values associated with the inherent uncertainty of the system. The best estimates emphasise estimates made for short contact times between water and sediment, as is appropriate for the relatively short-term modelling required for this project where sorption equilibrium is not likely to be achieved.

Table B.1 Estimated K_d (l kg^{-1}) values for freshwater systems. Best estimate values emphasise short (<48 h) contact times between radionuclide and sediment, and where possible account for environmental conditions prevailing in the Thames. Range is estimated for all contact times and environmental conditions

Element	Best estimate K_d l kg^{-1}	Range of K_d l kg^{-1}
^3H	1	0 – 10
^{14}C	1×10^4	$< 1 \times 10^4$
^{32}P	5.7×10^3	$10^2 - 10^4$
Cs	5×10^3	$5 \times 10^2 - 5 \times 10^4$
Sr	10^2	$10 - 10^3$
Zn	5×10^2	$10^2 - 10^4$
I	50	0 – 10^2
Co	10^3	$10^2 - 10^5$
U	50	$10 - 10^3$
Pu	10^3	$10^2 - 10^5$
Am	5×10^3	$10^2 - 10^5$

Transfers of radioactivity from the river water to bottom sediments can occur either by sorption to and subsequent settling of suspended particulates, or by direct diffusion of the dissolved radionuclide across the sediment-water interface. The latter process has been observed for radionuclides in lakes (Hesslein, 1987) and for nutrients in experimental channels (House et al. 1995).

Transfers to sediments via settling particles are modelled using the K_d to estimate the fraction, f_p ([dimensionless]) of the radionuclide attached to suspended particles:

$$f_p = \frac{1}{1 + 1/sK_d} \quad (\text{B4})$$

where s (kg l^{-1}) is the suspended solids concentration of the river. Estimates of the particle settling velocity, v_p (m dy^{-1}) are then used to estimate the rate of transfer of suspended particulates to bottom sediments. At high water velocities, sediment will resuspend, rather than settle: the velocity at which this occurs may be calculated by estimating the velocity at which the shear stress on the river bed exceeds some critical value. Estimates of this critical shear stress for fine silt/clay sediments have been made in flume experiments (e.g. Partheniades and Asce, 1965).

Direct diffusion of a radionuclide to bed sediments is commonly modelled by estimating (or measuring) the thickness of the benthic laminar flow layer and assuming that the radionuclide diffuses across this layer into the sediment by molecular diffusion at a rate determined by the molecular diffusion coefficient of the dissolved ion. Clearly, as concentrations of the radionuclide in the sediment increase, the concentration gradient, and therefore the rate of diffusion across the boundary will decrease. In practice, however, it is usually assumed that the activity concentration in the sediment interstitial water is maintained at zero, resulting in transfer of a constant fraction of the dissolved activity in the water column into the sediment per unit time (e.g. Smith et al. 2000).

Whilst transfers of radionuclides to sediment via particle settling has been observed in many situations, the direct diffusion mechanism has only been observed in experimental channels. In a study of radiocaesium removal from European lakes, Smith and coworkers (1999) showed the particle settling process to be more important than direct diffusion. It is believed that no studies have determined the relative importance of the two processes in rivers. Both particle settling and direct diffusion processes are dependent on the K_d : in general, both predict a greater rate of transfer of high K_d radionuclides than those with low K_d . This makes the relative importance of the two processes difficult to determine, but it also means that the particle settling model also simulates the behaviour of the direct diffusion model (i.e. both respond to the key environmental variable, K_d in the same way). For the purpose of this study, the direct diffusion process will be ignored, in view of the difficulty of accurately quantifying it, its high input data requirements, and uncertainty as to its importance. The particle settling model, we believe, is much more simply and accurately quantified, and the addition of a direct diffusion component is unlikely to improve model predictive power, nor would it have a significant effect on model output. The particle settling model therefore best suits our purpose of developing a simple and robust predictive model.

B5.1 Comparison with the Schaeffer coefficient approach

An alternative approach to modelling transfers of radionuclides to sediments assumes that, for a constant discharge of radioactivity to a river, the resultant activity concentration in the river water declines exponentially with distance from the discharge point as a result of dilution and transfers to sediments (Simmonds et al. 1995). The transfer to sediment component is modelled using an empirically determined “Schaeffer coefficient”. This coefficient is dependent on the sorption properties of the radionuclide. Whilst this approach may be

appropriate for routine releases, it is not appropriate for our purposes since it assumes that sediment deposition is independent of water velocity. This assumption does not hold since it is well known that sediment deposition is strongly dependent on water velocity. Comparison of the particle settling velocity model with the Schaeffer model is shown in the model validation section, Appendix F, section F3.

B6 Physical transport modelling – resuspension and movement of bottom sediments

The resuspension of bottom sediments is, like particle settling, determined by water shear stress on the bed sediments. Above a critical shear stress, bed sediment particles will be entrained into the river flow, carrying with them the absorbed radionuclide. River bed sediments tend to consolidate over (often long) periods of low to medium flow, then resuspension of large areas of the river bed can occur during short flood periods. This high dependence of resuspension on river flow rate makes it extremely difficult to model accurately at high temporal resolution (to predict, for instance, daily resuspension rates). Chemical remobilisation of radionuclides by diffusion from bottom sediments is possible, and has been observed under anoxic conditions in lakes (e.g. Evans, 1983). Though modelling diffusional remobilisation is possible (Smith and Comans 1996), such models are complex and require data on sediment chemical and physical composition, as well as detailed information on sorption kinetics of particular radionuclides. Inclusion of a remobilisation component is therefore not suited to our purpose, and, given the present state of knowledge of its parameter values, would not improve model predictive power.

In view of the difficulties in modelling sediment uptake and transfer of radionuclides, conservative bounding value assumptions will be made regarding these processes:

- *Activity concentrations in water and fish.* It will be assumed that there is no transfer of radionuclides to bed sediments. Under this assumption, the model would over estimate maximum and integrated activity concentrations for water and fish if in reality there was significant storage of radioactivity in bed sediments;
- *Activity concentrations in bed sediments.* It will be assumed that there is transfer of radioactivity to bed sediments, but that there is no subsequent movement of bed sediments. Under this assumption, the model would over-estimate maximum and integrated activity concentrations in bed sediments if in reality there was significant resuspension and movement of bed sediments.

APPENDIX C: MODEL DEVELOPMENT

C1 Physical transport model

Summary of model parameters

D	Dispersion coefficient	$\text{m}^2 \text{s}^{-1}$
v	Advection coefficient (mean water velocity)	m s^{-1}
Q	Volumetric flow rate	$\text{m}^3 \text{s}^{-1}$
v_p	Settling velocity of suspended solids	m d^{-1}
s	<i>Suspended solids concentration</i>	mg l^{-1}
ρ_s	Bed sediment density	kg m^{-3}
d_s	Sediment mixing depth	cm

An “off the shelf” code which meets these modelling needs was not found. The model was therefore implemented in-house using a code written in FORTRAN F77 and run on a SUN workstation.

There are two distinct time scales appropriate to our requirements for modelling the transport of radionuclides in the river system. Two separate models have been developed for these two time periods:

- *Early phase.* The period from the beginning of radionuclide release to the movement of the discharge plume out of the 10 km stretch. This model incorporates advection and dispersion of the discharge plume as well as transfers of radionuclides to bottom sediments.
- *Late phase.* The period following the movement of the discharge plume out of the 10 km stretch. Models movement and resuspension of bed sediment.

The two separate transport models and the fish model were implemented in a single code in order to give output for the whole period after release which is being considered.

C1.1 Early phase

A simple ADE model was developed using a backward difference explicit numerical simulation of the advection-dispersion equation. The effects of numerical dispersion were nullified by setting the time and distance steps (Δt , Δx) such that the numerical dispersion coefficient was equal to the true dispersion coefficient. Thus, the dispersion term in the ADE was not explicitly evaluated, but was simulated using numerical dispersion (see, e.g. Smith and Elder 1999). The transfer of radionuclide to bed sediments is evaluated using the transfer rate, k_1 (dy^{-1}):

$$k_1 = \frac{f_p v_p}{d} \quad (\text{C1})$$

where f_p ([dimensionless]) is the particulate sorbed fraction (Equation B4), v_p (m dy^{-1}) is the particulate settling velocity and d (m) is the river mean depth.

Upon deposition to the bed sediment, the radionuclide is mixed (by physical mixing and diffusion) within the surface layers of sediment. The average activity concentration of sediment, C_s (Bq kg⁻¹) within a layer of depth d_s (m) is then given by:

$$C_s = \frac{A_s}{d_s \rho_s} \quad (C2)$$

where A_s (Bq m⁻²) is the total activity of radionuclide deposited per square metre of sediment and ρ_s (kg m⁻³) is the dry mass per unit wet volume of sediment.

C1.2 Late phase

After the release, the plume will travel out of the 10 km stretch we are considering. Following this period, the movement and resuspension of bed sediments containing radionuclides can be modelled using a compartmental model. The concentration, $C_s^{i,j}$ (Bq kg⁻¹) of radionuclide on sediment in river stretch i at time step j is given by:

$$C_s^{i,j} = C_s^{i,j-1} + \frac{\Delta t}{\Delta x} v_s (C_s^{i-1,j-1} - C_s^{i,j-1}) \quad (C3)$$

where Δt , Δx are the time and distance (compartment length) steps in the numerical scheme and v_s (m s⁻¹) is the average velocity of movement of bed sediment. Given an initial distribution of radionuclide in the sediment (from the plume transport model above), the concentration of radionuclide in sediment can then be calculated for each river stretch (compartment) and each time step.

The activity concentration of radionuclide in the river water is estimated by calculating the resuspension rate of sediment travelling down river with average velocity v_s (m s⁻¹). Given velocity of sediment movement, v_s , the fraction of sediment resuspended from a stretch of length Δx and width W , during time period Δt is:

$$v_s \Delta t / \Delta x \quad (C4)$$

and assuming that all of this resuspended sediment comes from the surface layer of depth d_s , the amount of resuspended radionuclide activity (Bq) in time Δt is given by:

$$\frac{v_s \Delta t}{\Delta x} . d_s \Delta x . W \rho_s C_s \quad (C5)$$

hence the activity concentration (Bq m⁻³) in river water (dissolved and particulate phases) is:

$$\frac{1}{Q \Delta t} \frac{v_s \Delta t}{\Delta x} . d_s \Delta x . W \rho_s C_s \quad (C6)$$

This model assumes that sediment moves from one river stretch (of length Δx) to the next with average velocity v_s via the overlying river water, thus the resuspension increases the activity concentration in the river water.

C2 Fish uptake and retention model

The level of radioactive contamination of aquatic biota is commonly defined in terms of a concentration factor (CF) where

$$CF = \frac{\text{Activity concentration per kg of fish (wet wt)}}{\text{Activity concentration per litre of water}} \text{ l kg}^{-1} \quad (C7)$$

Previous studies on the accumulation of radionuclides in fish have focused on the prediction of *CF* (sometimes termed the bioaccumulation factor, *BAF*, or aggregated concentration factor, *ACF*). Some models (e.g. Rowan and Rasmussen, 1994; Smith et al., 2000b) predict the water-fish *CF* for certain radionuclides using relationships with the water chemistry. using an inverse relation between the *CF* and the potassium concentration of the surrounding water. For most radionuclides, however, single "best estimate" *CF* values are assumed to apply to all freshwater bodies.

Estimates of the fish-water *CF* may be made from measurements (in the lab or field) reported in the literature. Alternatively, in appropriate cases, *CF* may be estimated from the concentration factor of the stable isotope:

$$CF \approx \frac{\text{mass of stable isotope per kg of fish (w.w)}}{\text{mass of (available) stable isotope per l of water}} \quad (C8)$$

The equilibrium *CF* modelling approach is appropriate for cases in which the radionuclide activity concentration in fish can be assumed to be in equilibrium with that in water, for example at long times (years) after radionuclide fallout, or for continuous releases of radionuclides to a river. At short times after radioactive contamination of an aquatic system, or where activity concentrations in water are changing relatively rapidly, a dynamic modelling approach may be more appropriate. Dynamic models for radiocaesium accumulation in freshwater fish have been developed using the results of controlled laboratory experiments (for example, Garnier-Laplace et al., 1997).

It is known that the bioaccumulation of radioactivity in fish is determined by numerous ecological and environmental factors such as the trophic level of the fish species, the length of the food chain, water temperature and the water chemistry. Uptake may be via ingestion of contaminated food or direct transfers from the water via the gills. For most radionuclides the food chain is the primary uptake pathway, so a food uptake model will be used to estimate uptake rates. For strontium isotopes, which are primarily absorbed through the gills, a model for direct uptake via the water (Chowdhury and Blust 2001) was used to estimate the intake rate. Where the primary uptake pathway is uncertain, as in the case of Co for example, a food uptake model was used, but a conservative approach was taken which over-estimates the uptake rate to account for possible influence of the direct uptake (gill) pathway.

Summary of main symbols with units as used

C_f	Concentration of the RN in fish	Bq kg^{-1}
CF	Fish-water concentration factor	l kg^{-1}
CF_{food}	Food-water concentration factor	l kg^{-1}
C_{food}	Concentration of the RN in prey fish	Bq kg^{-1}
C_w	Activity concentration of the RN in water	Bq l^{-1}
k_f	Rate constant of the RN uptake in fish	$\text{l kg}^{-1} \text{d}^{-1}$
k_b	Rate constant of RN excretion from fish	d^{-1}
D_{max}	Maximum daily intake of food by trout	mg d^{-1}
α	Radionuclide assimilation efficiency	[]
w	Wet weight of fish	g

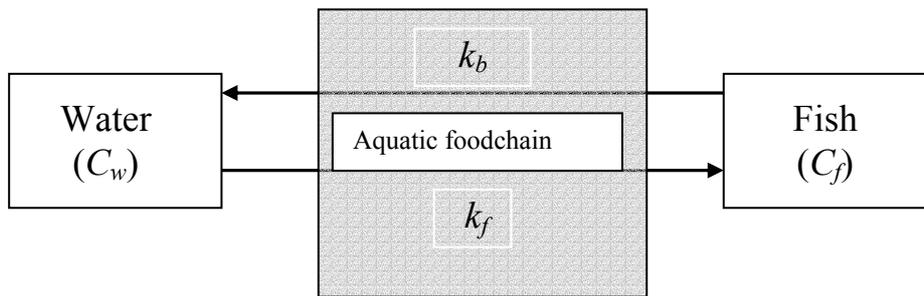


Figure C.1 Illustration of model for uptake in fish via the food chain

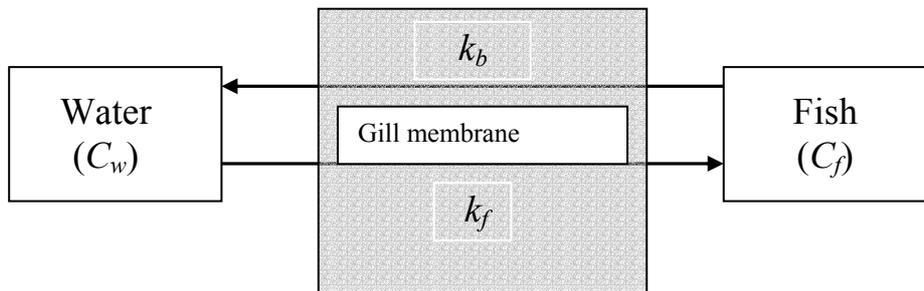


Figure C.2 Illustration of model for uptake in fish via the gills. Mathematically the model is identical to the food chain model, but the uptake parameter is calculated using a Michaelis-Menten type model (Chowdhury and Blust 2001) for transfer of Sr across the gill membrane

The activity concentration of a radionuclide in fish, C_f (Bq kg^{-1}) may be modelled by a simple “two-box” model describing uptake from the water C_w (Bq l^{-1}) and release from the fish (Figure C.1, C.2):

$$\frac{dC_f}{dt} = k_f C_w - (k_b + \lambda) C_f \quad (\text{C9})$$

where k_f ($\text{l kg}^{-1} \text{d}^{-1}$) is the rate constant describing transfers of ^{137}Cs to fish through its food and k_b (d^{-1}) is the backward rate constant describing excretion of radioactivity from the fish. The ratio of these rate constants gives the equilibrium concentration factor, CF (l kg^{-1}), of the radionuclide in fish relative to water:

$$\frac{k_f}{k_b} = \frac{C_f}{C_w} (\text{at equilibrium}) = CF \quad (C10)$$

For a constant activity concentration in the water phase, C_w , equation (C9) has solution:

$$C_f = \frac{k_f C_w}{k_b + \lambda} (1 - \exp(-(k_b + \lambda)t)) \quad (C11)$$

C3 Estimating uptake and excretion rates

For a number of radionuclides under study it is possible to estimate rates of uptake and excretion, k_f , k_b using field and/or laboratory studies of these rates. It is rare for both the uptake and excretion rate to be measured in a form useable for this predictive modelling. If, however, either the uptake or the excretion rate is known, Equation (C10) may be used to estimate the excretion rate using the uptake rate and CF (or, conversely, the uptake rate may be estimated from the excretion rate and CF).

In cases where uptake is principally via ingestion we can estimate the uptake rate by:

$$k_f = \frac{C_{food} \times D_{max} \times \alpha \times 10^{-3}}{C_w \times w \times 10^{-3}} = \frac{CF_{food} \times D_{max} \times \alpha \times 10^{-3}}{w \times 10^{-3}} \quad (C12)$$

where D_{max} (g d^{-1}) is the maximum daily intake (wet weight) of food by trout, w is the wet weight of fish in grammes and α is the assimilation efficiency (the fraction of amount ingested which is absorbed by the fish). The factor 10^{-3} in the numerator is required to convert D_{max} from g d^{-1} to kg d^{-1} and the factor 10^{-3} in the denominator is required to convert w from grammes to kg. C_{food} is the activity concentration of the food and CF_{food} is the concentration factor of the food (e.g. plankton for herbivores or herbivorous fish for piscivores). In this study model predictions will be made for piscivorous fish. It is assumed that uptake to the prey of piscivorous fish is instantaneous, thus the activity concentration in the prey, C_{food} , is estimated from the concentration of radioactivity in the water, C_w , using the concentration factor, CF_{food} . CF_{food} is in most cases assumed to be equal to the concentration factor of the predatory fish unless it is known to be significantly different (for example, for ^{137}Cs , it is known that the CF of predatory fish is approximately two times higher than their herbivorous prey, Rowan and Rasmussen 1994).

It was assumed that fish feed at their maximum daily rate and calculations were made for trout, a predatory fish about which there is good data on feeding rates. Elliot (1975a) developed an empirical model which estimates trout feeding rate for fish of different wet weight w (grammes) at different water temperatures, T :

$$D_{max} = (4 \times 10^{-3}) A_D \times w^{b_1} \times \exp(b_3 T) \quad (C13)$$

where A_D , b_1 and b_3 are empirically determined constants whose values are given in Table C.1. The factor 4 converts the dry weight feeding rates estimated by the Elliot (1975a) model to feeding rate expressed in terms of wet weight as used in equation 16. The factor 10^{-3} converts D_{max} in mg per day estimated by the Elliot model to g per day used in Equation C12.

Table C.1 Values of constants for different temperature ranges in model for estimating fish feeding rates (Elliot 1975a)

$T^{\circ}C$	A_D	b_1	b_3
3.8 - 6.6	0.654	0.762	0.418
6.6 - 13.3	3.384	0.759	0.172
13.3 - 18.4	5.956	0.767	0.126

C4 Estimating uptake of Sr via the gills

For strontium isotopes, which are primarily absorbed through the gills, we will use a model for direct uptake via the water (Chowdhury and Blust 2001) to estimate the intake rate.

The Chowdhury and Blust model estimates the stable strontium uptake rate, j_{Sr} ($\mu\text{mol kg}^{-1} \text{h}^{-1}$) as a function of the H^+ and Ca^{2+} concentration:

$$j_{Sr} = J_{\max Sr} \frac{\beta_{Sr}(H^+) + K_{iH}}{(H^+) + K_{iH}} \frac{(Sr^{2+})}{(Sr^{2+}) + K_{mSr} [1 + (Ca^{2+}) / K_{iCa}]} \quad (C14)$$

where the constants of the model have values given in Table C.2 and Sr^{2+} , H^+ , Ca^{2+} are measured in μM . The Sr uptake rate, j_{Sr} , is converted to the uptake rate constant in the model developed here, k_f ($1 \text{ kg}^{-1} \text{d}^{-1}$):

$$k_f = \frac{24 \cdot j_{Sr}}{(Sr^{2+})} \quad (C15)$$

where the uptake rate is divided by the Sr^{2+} concentration to convert the absolute Sr^{2+} uptake rate (i.e. μM of Sr per hour per kg of fish) to the uptake rate per unit concentration (in Bq or μM) in water. The factor 24 accounts for the change of units from h^{-1} to d^{-1} .

Table C.2 Parameter values for the Sr uptake model (Chowdhury and Blust 2001)

Parameter	Estimated value
$J_{\max Sr}$	$293.0 \mu\text{mol kg}^{-1} \text{h}^{-1}$
K_{mSr}	$96.3 \mu\text{M}$
K_{iCa}	$28.5 \mu\text{M}$
K_{iH}	$0.54 \mu\text{M}$
β_{Sr}	0.35

APPENDIX D: MODEL INPUT DATA AND PARAMETER VALUES

D1 Physical transport model

D1.1 Advection and dispersion rates

For two of the reaches, Sutton Courtney and GUC/Colne, the Environment Agency have supplied data from dye tracer experiments, some at different river flow rates. We have fitted a solution to the ADE to these data in order to obtain advection and dispersion rates. Parameter values obtained from these model fits are given in Tables D.1 and D.2. As shown in Figure D.1, for the Upper Thames data, strong correlations are observed between advection and dispersion rates and river volumetric flow rate. In addition, different river reaches show similar advection and dispersion parameters.

The following relationships between volumetric flow rate and velocity and dispersion were determined by least squares regression to the dye-tracer measurements for the Upper Thames (Figure D.1):

$$D = 0.0148Q^2 + 0.33Q \quad R^2 = 0.93 \quad (D1)$$

$$v = 0.028Q^{0.7} \quad R^2 = 0.93 \quad (D2)$$

The data for the Colne/Grand Union Canal is not so easily analysed, there were only data for medium flow conditions. The data for the Colne/GUC was therefore supplemented by a dye tracer experiment carried out during this study (see below).

Table D.1 Advection and dispersion coefficients derived from dye tracer experiments on the Upper Thames (data supplied by the Environment Agency). The Sutton Courtney reach corresponds approximately to the reach studied in experiment UTH19 which goes from Culham Lock to Clifton Lock

Reach	Length km	Flow, $Q \text{ m}^3 \text{ s}^{-1}$	Mean velocity, $v \text{ m s}^{-1}$	Dispersion, $D \text{ m}^2 \text{ s}^{-1}$
UTHL16 Sandford Lock	4.4	3.27	0.062	0.67
UTHM16 Sandford Lock	4.4	19.8	0.17	13.2
UTHH16 Sandford Lock	4.4	42.5	0.39	40.9
UTHL17 Culham Intake	3.05	3.27	0.055	0.84
UTHM17 Culham Intake	3.05	19.8	0.24	7.3
UTHH17 Culham Intake	3.05	42.5	0.38	46.4
UTHL18 Abingdon Lock	4.89	3.27	0.086	0.088
UTHM18 Abingdon Lock	4.89	19.8	0.23	16.5
UTHH18 Abingdon Lock	4.89	42.5	0.34	28.5
UTHL19 Culham Lock	5.3	5.45	0.070	1.73
UTHM19 Culham Lock	5.3	19.8	0.35	14.0
UTHH19 Culham Lock	5.3	44.1	0.36	48.8

Table D.2 Advection and dispersion coefficients derived from dye tracer experiments on the River Colne/Grand Union Canal (data supplied by the Environment Agency). The GUC/Colne reach corresponds approximately to the reach studied in experiment CLNM4 “Maple Lodge STW”

Reach	Length km	Flow, $Q \text{ m}^3 \text{ s}^{-1}$	Mean velocity $v \text{ m s}^{-1}$	Dispersion $D \text{ m}^2 \text{ s}^{-1}$
CLNM2 Little Munden Farm	14	3.06	0.11	13.8
CLNM3 Batchworth	6.4	3.06	0.15	4.7
CLNM4 Maple Lodge STW	6.2	3.06	0.15	4.7
CLNM5 The Lea	5.0	3.06	0.13	1.1
CLNM15 Poyle	2.25	0.23	0.15	2.0
CLNM16 Staines WS Intake	3.0	0.23	0.096	0.66

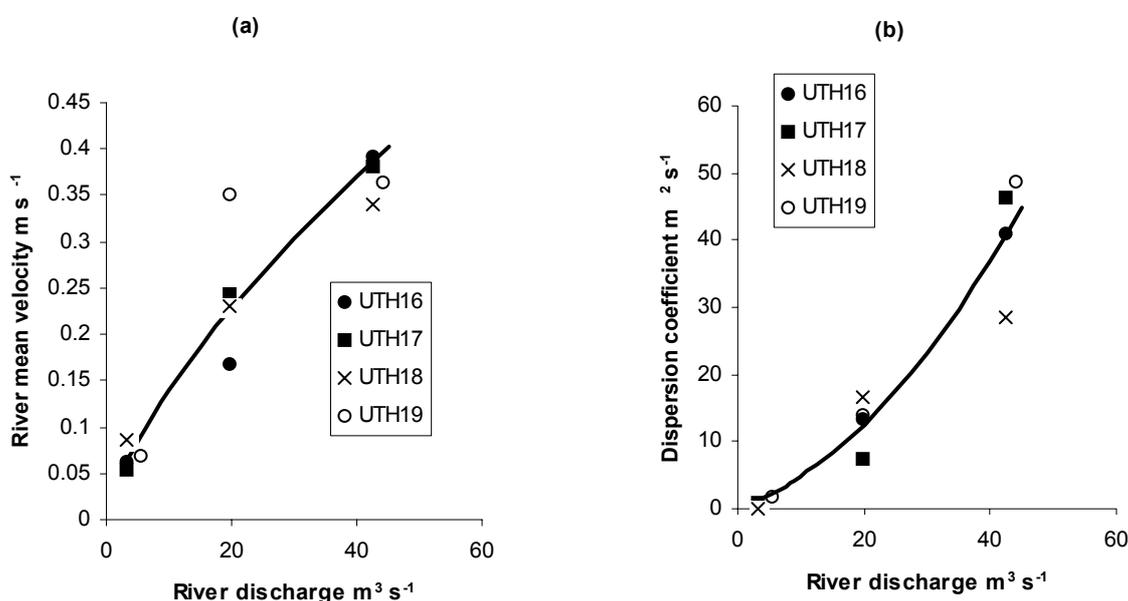


Figure D.1 Relationships between (a) advection and (b) dispersion of a conservative dye tracer and river volumetric flow rate for four reaches of the Upper Thames (from data in Table 5)

D2 Field studies for model parameterisation

Rhodamine WT dye was used to estimate the advection and dispersion of a short term release of radionuclides from Maple Lodge STW into the GUC/River Colne and into the Thames at Pangbourne. For the Sutton Courtney reach, previous data obtained by the Environment Agency for advection and dispersion in this part of the river was used. The pilot field study, planned for August 2000, was postponed at the request of the Environment Agency. The re-scheduled study, planned for 9-11 April was postponed because of concerns over foot-and-mouth, and flooding. Three experiments were finally carried out, one on the Colne/GUC on 05/06/01 and two on the Pangbourne-Reading reach of the Thames on 19/06/01 and 28/08/01.

D2.1 Calculating the amount of Rhodamine WT to inject

The limit for Rhodamine WT concentration at drinking water intakes is $1 \mu\text{g l}^{-1}$. The amount of Rhodamine WT to be injected given a limiting concentration C_p ($\mu\text{g l}^{-1}$) may be calculated using Church's equation (Church 1974):

$$W = 3.8 \times C_p \left(\frac{QL}{v} \right)^{0.93} \quad (\text{D3})$$

where W (milligrammes) is the amount of 20% solution to be injected (i.e. 5 times the mass of Rhodamine WT added), Q is the volumetric flow rate of the river ($\text{m}^3 \text{s}^{-1}$), v is the mean water velocity (m s^{-1}) and L is the length of reach (m).

In addition, for aesthetic reasons, Thames Region Environment Agency have (by trial and error) set a limit of 15 mg/l of Rhodamine WT at the injection point per second of volume flow (Andrzej Nowosielski, Thames Region EA, pers. comm.). Thus the maximum mass of Rhodamine WT to be added is given by:

$$M = Q \times 15 \quad (\text{D4})$$

where M is in grammes and Q is in $\text{m}^3 \text{s}^{-1}$. Note that the mass of 20% solution of Rhodamine WT would be 5 times this amount, $W = 5 \times M$. In practice, Equation (D4) determined the maximum amount of dye to inject. Since drinking water abstractions were a long way from the injection points, the water quality limit did not constrain the amount added.

The amounts of Rhodamine WT added and volumetric flow rate data are given in Table D.3.

Table D.3 Amounts of Rhodamine WT added and volumetric flow rate for each dye tracer experiment

Site	Date	Injection time	Vol. flow rate $Q \text{ m}^3 \text{s}^{-1}$	Amount of 20% solution added, g.
Maple Lodge	05/06/01	06:36	8.035*	600
Pangbourne	19/06/01	05:56	25.5**	1500
Pangbourne	28/08/01	06:15	10.5 ⁺	600

* Flow rate for Colne/GUC at Denham at 09:00 on 05/06/01.

** Average for Thames at Reading 06:00 – 10:00 on 19/06/01.

+ Average for Thames at Reading 06:00 – 10:00 on 28/08/01.

The 20% solution of Rhodamine WT dye was mixed with river water in a bucket, then poured into the river. At Maple Lodge, the Rhodamine WT dye was poured from the side of the river into the stream of effluent from the STW. At Pangbourne, the dye was poured from a boat above the pipeline, spread in a line stretching half way across the river (from 5 m from the right bank to the centre of the river). The concentration of dye at various points downstream of the injection point was measured as a function of time using a Chelsea Instruments Minitracka II Fluorimeter. The results of the experiments are summarised in Tables D.4-D.5 and Figures D.1-D.2.

Table D.4 Advection and dispersion parameters for sites downstream of Maple Lodge STW on 05/06/01. Volumetric flow rate of the river was $8 \text{ m}^3 \text{ s}^{-1}$

Site	Distance (m)	Velocity v (M s^{-1})	Dispersion D ($\text{m}^2 \text{ s}^{-1}$)
1. Coppermill Lane	900	0.25	5.9
2. Denham Green	4500	0.38	13.2
3a. Willowbank	9000	0.35	13.2
3b Willowbank	9000	0.30	10.7
Predicted before expt.		0.4	14.9

River Colne - results and model fits

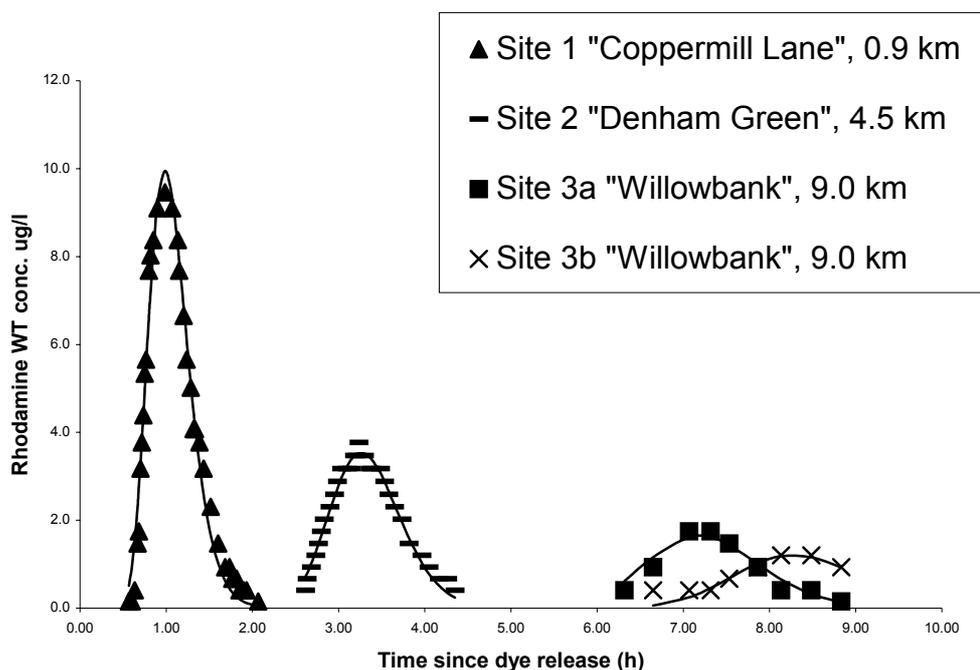


Figure D.2 Fit of the ADE model (solid lines) to field data from the River Colne/Grand Union Canal. Parameters determined from these data are given in Table D.5

Table D.5 Advection and dispersion parameters for sites downstream of the assumed discharge point, Pangbourne reach

Site	Distance (m)	Velocity v (m s^{-1})	Dispersion D ($\text{m}^2 \text{ s}^{-1}$)	Flow rate Q , $\text{m}^3 \text{ s}^{-1}$
1. Boathouse	1100	0.24	0.38	25.5
2. Mapledurham	2200	0.21	0.96	25.5
3. Tilehurst	5300	0.20	13.2	25.5
Predicted before expt.		0.21	18.0	
1. Boathouse	1100	0.11	0.97	10.5
Predicted before expt.		0.09	5.1	

**River Thames Pangbourne:
Results and model fits, $Q = 25.5 \text{ m}^3 \text{ s}^{-1}$**

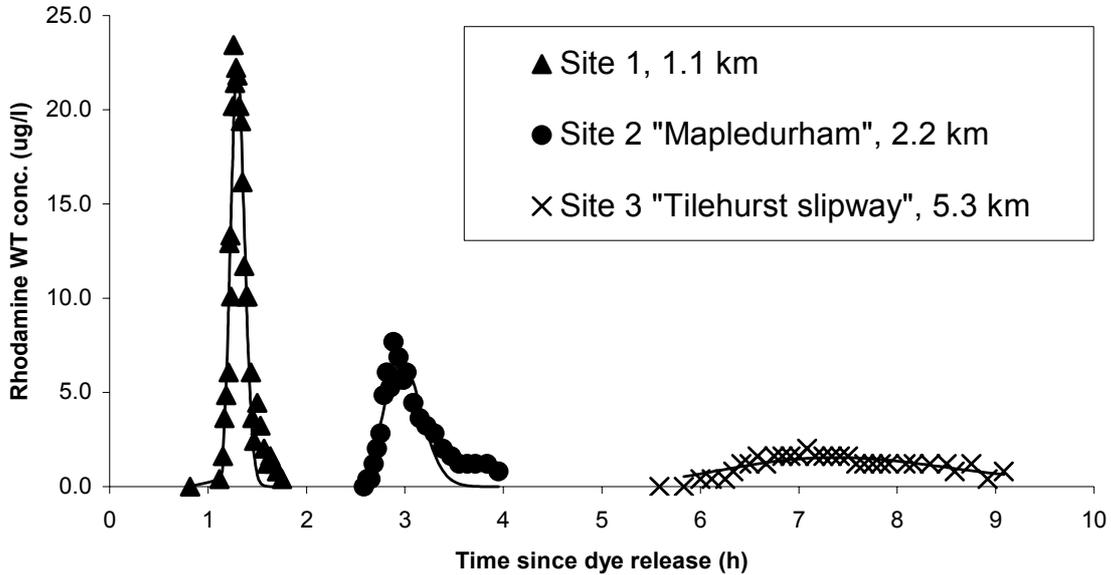


Figure D.3 Fit of the ADE model (solid lines) to field data from the River Thames at Pangbourne (medium flow). Parameters determined from these data are given in Table 10

**River Thames Pangbourne:
Results and model fits, $Q = 10.5 \text{ m}^3 \text{ s}^{-1}$**

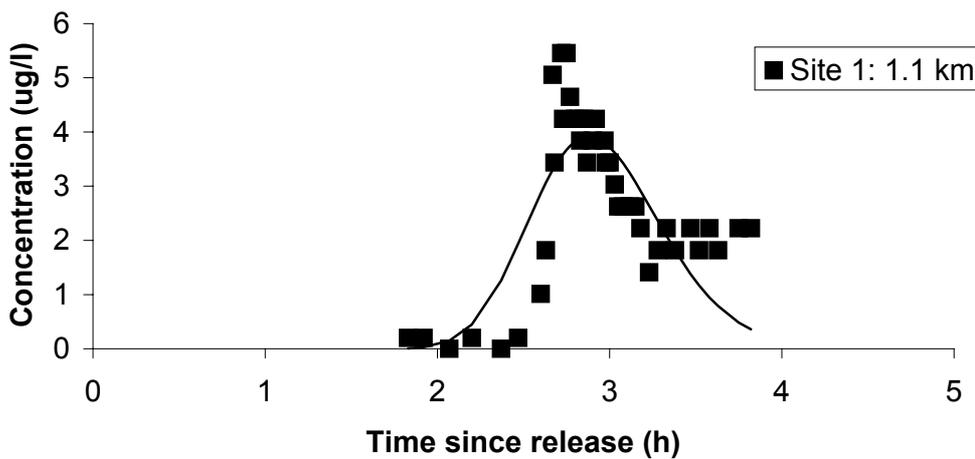


Figure D.4 Fit of the ADE model (solid lines) to field data from the River Thames at Pangbourne (low flow). Parameters determined from these data are given in Table D.5

D3 Sediment model

D3.1 Fraction of radionuclide adsorbed to suspended solids and settling velocity

The concentration of suspended sediments in the Thames is extremely variable, being in the range 1.3 – 50.1 mg l⁻¹ with the mean of 108 samples being 13.0 mg l⁻¹ over a two year sampling period (Neal and Robson, 2000: data for Thames at Day's Lock). One measurement made in this study gave a value of 8.7 mg l⁻¹ in the GUC at Coppermill Lane, within the range observed for the Thames.

Table D.6 shows the fraction, f_p , of the different radionuclides estimated to be adsorbed to suspended sediments at a suspended solids concentration, s , of 13.0 mg l⁻¹ showing that the majority of all of the radionuclides are expected to be found in solution. Given the high uncertainty in K_d values, however, for the modelling we will assume three categories of radionuclide sorption: $f_p = 0.0, 0.05$ and 0.95 . In Table D.6, however, we give recommended best estimates for use in the model: these are given separately for estimation of activity concentrations in water and fish (where lower f_p gives higher activity concentrations) and for activity concentrations in sediment (where higher f_p gives higher activity concentrations).

Table D.6 Estimated (short term) K_d (l kg⁻¹) values and fraction of radionuclide sorbed to solid phase assuming a suspended solids concentration, s , of 13.0 mg l⁻¹

Element	Best estimate K_d [l kg ⁻¹]	Fraction in solid phase, f_p []	Selected value for model (water, fish)	Selected value for model (sediment phase)
³ H	0	0.00001	0	0.05
¹⁴ C	1×10 ⁴	0.12	0	0.95
³² P	5.7×10 ³	0.07	0	0.95
Cs	5×10 ³	0.04	0	0.95
Sr	10 ²	0.0013	0	0.05
Zn	5×10 ²	0.0065	0	0.05
I	50	0.00065	0	0.05
Co	10 ³	0.013	0	0.05
U	50	0.00065	0	0.05
Pu	10 ³	0.013	0	0.05
Am	5×10 ³	0.04	0	0.95

To estimate the settling velocity of suspended sediments, we will use a typical value for freshwater sediments of $v_p = 1$ m dy⁻¹ (Smith et al. 1999, from measurements in lakes).

D3.2 Bed sediment parameters

The dry mass per unit wet volume of bed sediments, ρ_s (kg m⁻³), is taken to be equal to that observed in the River Blackwater, a tributary of the Thames, $\rho_s = 500$ kg m⁻³ (F H Denison, unpublished results). The sediment mixing depth, d_s , is taken as 0.02 m.

D4 Parameter values for the fish uptake model

Table D.7 shows values of the CF obtained from a review of the literature with recommended values shown in bold. These are generally conservative estimates, particularly for those radionuclides about which there is relatively little information available. Uptake and excretion rates were determined for each radionuclide, as described below. Where possible, we have tested these models against independent laboratory or field data.

D4.1 Caesium-137

Rates of uptake of radiocaesium have been extensively studied both in the laboratory and in the field following the Chernobyl accident. The principal uptake route is via food, there being relatively low rates of transfer via the gills (e.g. Coughtrey and Thorne 1983). For piscivorous trout eating herbivorous food (e.g. small roach) a value of CF_{food} is taken as $CF/2$ since typically the concentration factor of herbivorous fish is half that of piscivorous (Rowan and Rasmussen 1994; Smith et al. 2000b). The assimilation efficiency, α , is taken as the average value for Atlantic salmon and brook trout of $\alpha = 0.44$ (Tucker and Rasmussen 1999). Using equation (C12) this gives a value of $k_f = 4.4 \times 10^{-4} \times CF_{fish} \times D_{max}$ for a 500 g fish and, from equation (C10), $k_b = k_f / CF_{fish}$.

The ^{137}Cs uptake model has been tested against measurements of ^{137}Cs in trout, pike, eel and perch in Windermere following Chernobyl (Camplin et al. 1989), as shown in Figure D.5. The model of Rowan and Rasmussen (1994) was used (see Table D.7) to predict the CF for Windermere for $[\text{K}^+] = 0.6 \text{ mg l}^{-1}$ and $s = 1.1 \text{ mg l}^{-1}$: changes in activity concentration in water are determined from Smith et al. 1997, and water temperature (required for estimating D_{max}) is estimated from Davison et al. (1993).

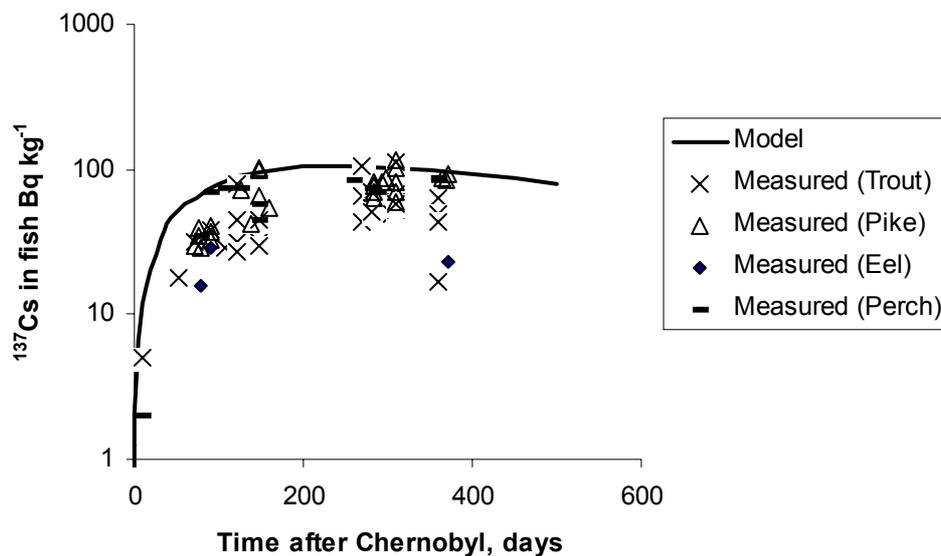


Figure D.5 Test of the ^{137}Cs uptake model against measurements in Windermere after Chernobyl

Table D.7 Concentration factors of radionuclides in whole fish

Element	Uptake pathway	Estimated CF used in model 1 kg^{-1}	Ranges and other estimates of CF 1 kg^{-1}	Notes and references
^3H	Water/food	1	1 (0 – 10)	IAEA (1994) Estimate and range.
^{14}C	Food	2.2×10^4	2.2×10^4 5×10^4 $5 \times 10^3 - 5 \times 10^4$	Estimate from stable carbon in fish (10% of wet weight, Vinogradov, 1953) and DOC in the Thames of 4.4 mg l^{-1} (Neal and Robson 2000). Likely to be over-estimate. IAEA (1994) Blaylock (1982), value for fish muscle, high value is for low mineral content water, low value for high mineral content water.
^{32}P	Food	1×10^4	5×10^4 ($3 \times 10^3 - 1 \times 10^5$) 3300 4020	IAEA (1994) Estimate and range. This study, from $\text{P} = 0.3\%$ of wet weight (Vinogradov, 1953) and SRP 0.91 mg l^{-1} for the Thames (Neal and Robson, 2000). Denyer (2001) estimate for Grand Union Canal for stable P.
Cs	Food.	2×10^3	2×10^3 ($3 \times 10^1 - 3 \times 10^3$) $3 \times 10^3/[\text{K}^+]$ 450^* $4880/[\text{K}^+]$ 730^+ $\log(\text{CF}) = 3.3 - 0.72\log[\text{K}^+] + 0.29 - 0.23\log[s]$ 570^{**}	IAEA (1994). Estimate and range. Coughtrey and Thorne (1983), for turbid water. *Estimate assuming $[\text{K}^+] = 6.7 \text{ mg l}^{-1}$ in the Thames (Neal and Robson, 2000). Smith et al.(2000), inverse relation with $[\text{K}^+]$. +Estimate assuming $[\text{K}^+] = 6.7 \text{ mg l}^{-1}$ in the Thames (Neal and Robson, 2000). Rowan and Rasmussen (1994), $[\text{K}^+]$ is in mg l^{-1} , s is the suspended solids conc. in mg l^{-1} . **Estimate assuming $[\text{K}^+] = 6.7 \text{ mg l}^{-1}$, s = 13.0 mg l^{-1} in the Thames (Neal and Robson, 2000).

Element	Uptake pathway	Estimated CF used in model l kg ⁻¹	Ranges and other estimates of CF l kg ⁻¹	Notes and references
Sr	Sr is taken up from water by the same mechanisms as Ca which (Vinogradov, 1953) is primarily absorbed direct from water via the gills.	60	60 (1 – 1 × 10 ³) CF = exp(5.2 - 1.2 ln[Ca ²⁺]) 0.54* 56 (0.82 – 198) 3.9 (0.74 – 10) 10.8	IAEA (1994) Best estimate and range. Vanderploeg et al. (1975), quoted in Blaylock (1982) for edible portions. Edible portions of fish have much lower CF's than whole fish. *Estimate for [Ca ²⁺] (121 mg l ⁻¹) in the Thames (Neal and Robson, 2000) for edible portions. Blaylock (1982) for edible portions of fish; calcium concentration in water <20 mg l ⁻¹ . Estimate and range. Blaylock (1982) for edible portions of fish; calcium concentration in water in range 20-60 mg l ⁻¹ . Estimate and range. Estimate from relations given by Vanderploeg et al. (1975), quoted in Blaylock (1982) (see text) for whole fish for [Ca ²⁺] (121 mg l ⁻¹) in the Thames (Neal and Robson, 2000).
Zn	Coughtrey and Thorne (1983) cite 3 reports referring to the importance of zinc intake from food, but note other reports have referred to passive sorption via the gills.	5 × 10³	1 × 10 ³ (1 × 10 ² – 3 × 10 ³) 2 × 10 ³ (2.8 × 10 ² – 2 × 10 ⁴) 4600 1250	IAEA (1994) Estimate and range. Coughtrey and Thorne Vol. 2 (1983). Estimate and range. CF likely to be related to stable Zn concentration. This study, from Zn(stable) = 0.003% of wet weight of fish (from data in Vinogradov, 1953) and [Zn] 6.5 × 10 ⁻⁶ g/l for the Thames (Neal and Robson, 2000). Denyer (2001) estimate for Grand Union Canal from stable Zn.
I	Limited data, but uptake is expected to be rapid (Kryshev 1995)	40	40 (20 – 6 × 10 ²) 40 (10 – 132) 10 30	IAEA (1994). Estimate and range. Blaylock (1982), Estimate and range for fish muscle Kryshev (1995) Coughtrey and Thorne vol 3 (1983) CF expected to be related to stable element concentration. Expected to concentrate somewhat in the thyroid.

Element	Uptake pathway	Estimated CF used in model 1 kg ⁻¹	Ranges and other estimates of CF 1 kg ⁻¹	Notes and references
Co	Evidence for significant uptake from both water and food (Baudin and Fritsch 1989).	3 × 10²	3 × 10 ² (10 – 300) 2.5 × 10 ² 5-280 20	IAEA (1994). Estimate and range. Coughtrey and Thorne vol. 2 (1983) Blaylock (1982), whole fish. Blaylock (1982), value for fish muscle. Greatest accumulation in internal organs, particularly kidneys.
U	Not known	50	10 (2 – 50) 0.3 - 0.6 8 0.7-38	IAEA (1994) Estimate and range. Blaylock (1982) planktivorous fish L. Michigan. Blaylock (1982) planktivorous fish, Lake Issyk-kul, Soviet Union. Blaylock (1982) omnivorous fish, Zirovski mining area, Yugoslavia.
Pu	Not known	50	30 (4 – 300) 35 (10 – 1000) 0.4 - 7	IAEA (1994) Estimate and range. Coughtrey et al. Vol 4 (1984) Estimate and range. Blaylock (1982) Accumulation decreases with increasing trophic level. Edible portions have lower CF's than whole fish.
Am	Not known	1000	30 (30 – 300) 840 (700-1000)	IAEA (1994) Estimate and range. Coughtrey et al. (1984) Vol 5. Best estimate and range.

Concentration factors (CF) are given for wet weight of freshwater (whole) fish unless stated otherwise. Where distinction between piscivorous and non-piscivorous fish has been possible, values for piscivorous fish were chosen. CF values were chosen for the Thames which has high mineral and nutrient content of the water. CF values for rivers of lower nutrient and/or mineral content are likely to be much higher

D4.2 Strontium-89, 90

There is less quantitative information available for uptake and retention rates of ^{90}Sr in fish than ^{137}Cs , however it is known that uptake is mainly direct from water via the gills. A direct uptake model will be used for predicting uptake rates for ^{90}Sr . Like calcium, strontium is primarily absorbed in the bony parts of the fish (skeleton, head, fins, scales). Measurements made by Vanderploeg et al. (1975), quoted in Blaylock (1982) have determined relationships between fish-water CF for ^{90}Sr , and $[\text{Ca}]$ (mg l^{-1}) in the surrounding water:

$$CF(\text{muscle}) = \exp(5.2 - 1.2 \ln[\text{Ca}]) \quad (\text{D6})$$

$$CF(\text{bone}) = \exp(9.7 - 1.2 \ln[\text{Ca}]) \quad (\text{D7})$$

Assuming that 20% of the wet weight of a fish is composed of bony parts (I I Ryabov, Severtsov Institute, Moscow, pers. comm.) this gives a whole fish CF :

$$CF(\text{whole fish}) = \exp(8.13 - 1.2 \ln[\text{Ca}]) \quad (\text{D8})$$

The uptake rate for strontium is estimated using a model for transfers by direct uptake across the gills (Chowdhury and Blust, 2001) for $[\text{Ca}]$; $[\text{Sr}(\text{stable})]$ and pH in the Thames from data in Neal and Robson (2000) gives $k_f = 0.675 \text{ l kg d}^{-1}$.

The ^{90}Sr uptake model (Chowdhury and Blust, 2001) was tested against measurements of ^{90}Sr in bream and pike-perch in the Kiev Reservoir following Chernobyl (Kryshev and Ryazantsev 2000), as shown in Figure D.6. This data is believed to be for whole fish so the whole fish CF_{fish} ($= 53 \text{ l kg}^{-1}$, for $[\text{Ca}] = 32 \text{ mg l}^{-1}$ in the Kiev Reservoir, I I Ryabov, Severtsov Institute, Moscow, pers. comm.) has been used (Table D.7). Activity concentration of ^{90}Sr in water is calculated using data from the Ukrainian Hydrometeorological Institute (Voitsekhovitch et al. 1997).

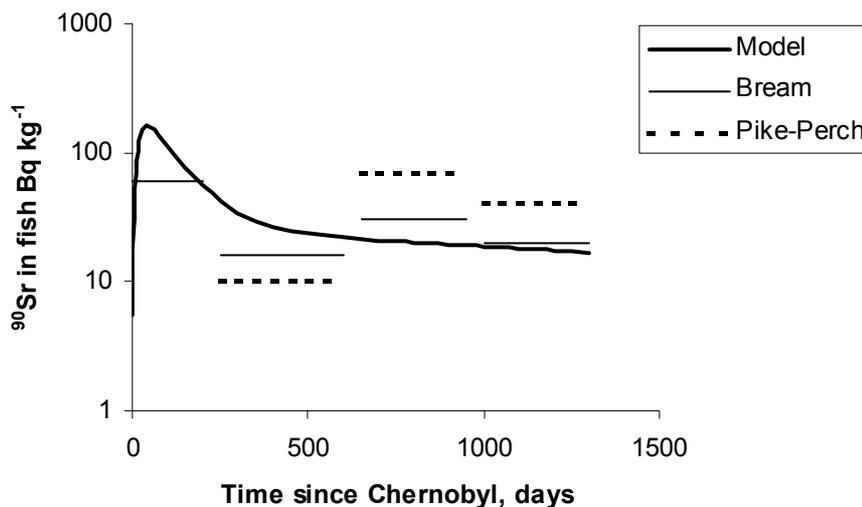


Figure D.6 Test of the ^{90}Sr uptake model against measurements in the Kiev Reservoir after Chernobyl

D4.3 Iodine-125, 131

There is less quantitative information available for uptake and retention rates of ^{131}I in fish than ^{137}Cs , so we will make the assumption that the assimilation efficiency is $\alpha = 1.0$ (i.e. all ^{131}I ingested is assimilated) and that $CF_{\text{food}} = CF_{\text{fish}}$. Using Equation C12 this gives a value of $k_f = 2.0 \times 10^{-3} \times CF_{\text{fish}} \times D_{\text{max}}$ for a 500 g fish and, from Equation C10, $k_b = k_f / CF_{\text{fish}}$. We have tested the ^{131}I uptake model against measurements in fish in the Kiev Reservoir following Chernobyl (Kryshev and Ryazantsev 2000), as shown in Figure D.7. The whole fish CF_{fish} ($= 40 \text{ l kg}^{-1}$) has been used (Table D.7). Summer temperature for the Kiev Reservoir is assumed to be 15°C for estimation of feeding rate. Activity concentration of ^{131}I in water is calculated using data from Kryshev and Ryazantsev (2000).

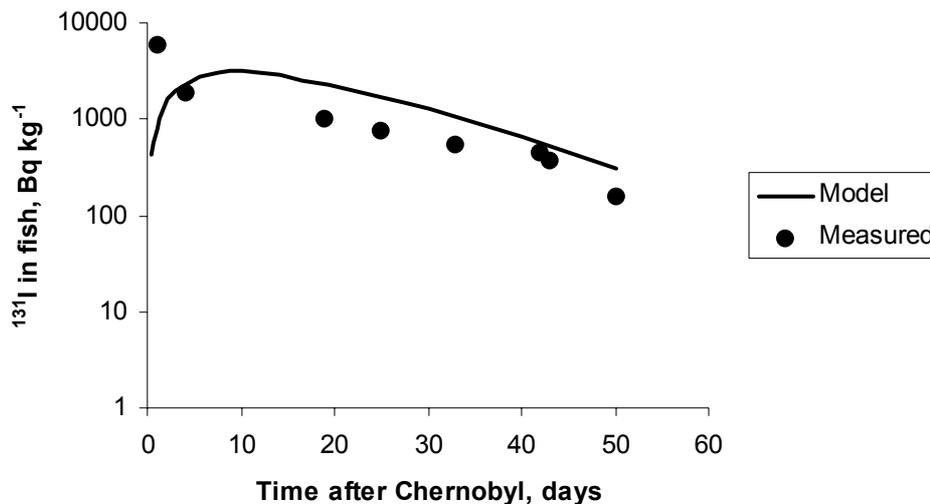


Figure D.7 Test of the ^{131}I uptake model against measurements in the Kiev Reservoir after Chernobyl

D4.4 Carbon-14

The CF of C-14 in fish has previously been estimated from the stable carbon content of fish (typically 10% of body wet weight, Vinogradov 1953) and the carbon ($\text{HCO}_3 + \text{CO}_3$) content of water (IL Ophel quoted in Blaylock 1982). This (Blaylock 1982) gave estimates of $CF = 5 \times 10^3 \text{ l kg}^{-1}$ for waters of high mineral content and $5 \times 10^4 \text{ l kg}^{-1}$ for waters of low mineral content. This compares with an estimate of $22 \times 10^3 \text{ l kg}^{-1}$ from stable carbon in fish (10% of wet weight, Vinogradov, 1953) and DOC in the Thames of 4.4 mg l^{-1} (Neal and Robson 2000). This latter is likely to be an over estimate of the CF (under estimate of available carbon) since it does not include organic particulate carbon and inorganic carbon in the water. The assimilation efficiency, α , of stable carbon in food may be estimated from data presented in Elliott (1975b) on growth rates of brown trout. For a 500 g trout at temperature 12°C equations given in Elliott (1975b) estimate a growth rate of 1.65 g dy^{-1} (wet weight) at a maximum feeding rate of 11.8 g dy^{-1} (wet weight). Assuming that the carbon content of food is approximately equal to that of the fish, this gives an assimilation efficiency, $\alpha \approx 14\%$.

D4.5 Phosphorus-32

The phosphorus content of fish is approximately 0.3% of wet weight (Vinogradov, 1953) and soluble reactive phosphorus (SRP) is 0.91 mg l^{-1} for the Thames (Neal and Robson, 2000), giving a CF of 3300 l kg^{-1} . Uptake and excretion rates are estimated by assuming that $CF_{food} = CF_{fish}$ and assuming an assimilation efficiency is $\alpha = 1.0$ (i.e. all radioactivity ingested is assimilated).

D4.6 Cobalt-60

There is less quantitative information available for uptake and retention rates of ^{60}Co in fish than ^{137}Cs . Laboratory studies, by Baudin *et al.* (2000) observed assimilation of ^{60}Co from food at a rate approximately 6 times lower than ^{137}Cs . We will make the (slightly conservative) assumption that the assimilation efficiency is $\alpha = 0.1$, approximately four times lower than for ^{137}Cs , and that $CF_{food} = CF_{fish}$. Using Equation (C12) this gives a value of $k_f = 2.0 \times 10^{-4} \times CF_{fish} \times D_{max}$ for a 500 g fish and, from Equation (C10), $k_b = k_f / CF_{fish}$.

The ^{60}Co uptake model was tested against laboratory measurements by Baudin and Fritsch (1989), as shown in Figure 12. The whole fish CF_{fish} ($= 300 \text{ l kg}^{-1}$) has been used (Table 12). Temperature of the experiments was $20 \text{ }^\circ\text{C}$ (Baudin and Fritsch 1989) for estimation of feeding rate. Activity concentration of ^{60}Co in water was approximately constant during the experiment having value $5 \times 10^4 \text{ Bq l}^{-1}$. Average fish mass during the experiment was approximately 1.4 g. As shown in Figure D.8 the model significantly over estimates activity concentrations in the fish. This may be because (as is typical for laboratory studies) very small fish (initial body mass 1 g) were used, and we have used an estimated CF_{fish} largely from field data where larger fish are typically studied. Our over-estimates of uptake rates are appropriate because it is also known (Baudin and Fritsch 1989) that water is also an important uptake pathway for ^{60}Co .

D4.7 Tritium

The biological half life of tritium is $< 1\text{d}$ so the concentration of H-3 in fish will closely follow the concentration of H-3 in ambient water (Vanderploeg *et al.* 1975 quoted in Blaylock, 1982). Assuming a biological half life of 1 d gives $k_b = 0.69 \text{ d}^{-1}$ and, using Equation C10 a value of $k_f = 0.69 \text{ d}^{-1}$. We have thus far found no empirical data against which to test ^3H uptake models.

D4.8 Americium-241, Uranium-234, 235, 238, Plutonium-238, 239, 240

Concentration factors of these radionuclides are given in Table D.7. Uptake and excretion rates are estimated by assuming that $CF_{food} = CF_{fish}$ and assuming an assimilation efficiency is $\alpha = 1.0$ (i.e. all radioactivity ingested is assimilated).

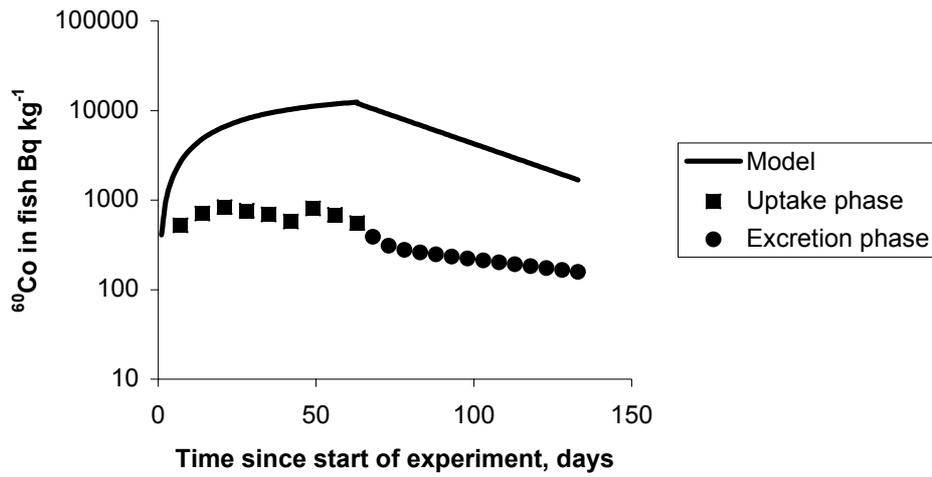


Figure D.8 Test of the ⁶⁰Co uptake model against measurements in a laboratory experiment (Baudin and Fritsch 1989)

Estimated uptake and excretion rates for a 500 g trout at water temperatures of 7, 12, 17°C are summarised in Tables D.8, D.9, D.10, respectively.

Table D.8 Estimated uptake and excretion rates of radionuclides in fish at 7 °C. All isotopes are assumed to be assimilated via the food pathway except strontium for which uptake via the gills is modelled and ³H for which no uptake mechanism is specified (measured uptake rates are used rather than an uptake model)

Element	Pathway	Relevant water chemistry	$CF(fish)$ l kg ⁻¹	$CF(food)$ l kg ⁻¹	Assimilation efficiency, α	Feeding rate, w.w g dy ⁻¹ @ 7 °C	Uptake rate k_f l kg ⁻¹ d ⁻¹ @ 7 °C	Excretion rate, k_b d ⁻¹ @ 7 °C
¹³⁷ Cs	Food	[K] 6.7 ¹ mg l ⁻¹	2×10^3	= $CF(fish)/2$	0.44 ²	5.0 ³	4.40	0.0022
⁹⁰ Sr	Water	[Ca] 121 ¹ mg l ⁻¹ [Sr] 0.36 ¹ mg l ⁻¹ pH 8.1 ¹	60	N/A	N/A	N/A	0.68	0.011
¹³¹ I	Food		40	= $CF(fish)$	1.0	5.0 ³	0.4	0.01
⁶⁰ Co	Food		3×10^2	= $CF(fish)$	0.1	5.0 ³	0.3	0.001
³ H	N/A		1.0	1.0	N/A	N/A	0.69	0.69
³² P	Food	[SRP] stable = 0.91 ¹ mg l ⁻¹	1.0×10^4	= $CF(fish)$	1.0	5.0 ³	100	0.01
¹⁴ C	Food	DOC 4.4 ¹ mg l ⁻¹	2.2×10^4	= $CF(fish)$	0.14 ⁴	5.0 ³	30.9	0.0014
Pu, U	Food		50	= $CF(fish)$	1.0	5.0 ³	0.50	0.01
⁶⁵ Zn	Food	[Zn] stable = 6.5 × 10 ⁻³ mg l ⁻¹ 1	5000	= $CF(fish)$	1.0	5.0 ³	50	0.01
²⁴¹ Am	Food		1000	= $CF(fish)$	1.0	5.0 ³	10	0.01

1. Neal and Robson, 2000; 2. Tucker and Rasmussen (1999); 3. Elliott (1975a); 4. From data in Elliott (1975b).

Table D.9 Estimated uptake and excretion rates of radionuclides in fish at 12 °C. All isotopes are assumed to be assimilated via the food pathway except strontium for which uptake via the gills is modelled and ³H for which no uptake mechanism is specified (measured uptake rates are used rather than an uptake model)

Element	Pathway	Relevant water chemistry	$CF(fish)$ l kg ⁻¹	$CF(food)$ l kg ⁻¹	Assimilation efficiency, α	Feeding rate, w.w. g dy ⁻¹ @ 12 °C	Uptake rate k_f l kg ⁻¹ d ⁻¹ @ 12 °C	Excretion rate, k_b d ⁻¹ @ 12 °C
¹³⁷ Cs	Food	[K] 6.7 ¹ mg l ⁻¹	2×10^3	= $CF(fish)/2$	0.44 ²	11.8 ³	10.4	0.0052
⁹⁰ Sr	Water	[Ca] 121 ¹ mg l ⁻¹ [Sr] 0.36 ¹ mg l ⁻¹ pH 8.1 ¹	60	N/A	N/A	N/A	0.68	0.011
¹³¹ I	Food		40	= $CF(fish)$	1.0	11.8 ³	0.94	0.024
⁶⁰ Co	Food		3×10^2	= $CF(fish)$	0.1	11.8 ³	0.71	0.0024
³ H	N/A		1.0	1.0	N/A	N/A	0.69	0.69
³² P	Food	[SRP] stable = 0.91 ¹ mg l ⁻¹	1.0×10^4	= $CF(fish)$	1.0	11.8 ³	236.0	0.024
¹⁴ C	Food	DOC 4.4 ¹ mg l ⁻¹	2.2×10^4	= $CF(fish)$	0.14 ⁴	11.8 ³	72.7	0.0033
Pu, U	Food		50	= $CF(fish)$	1.0	11.8 ³	1.18	0.024
⁶⁵ Zn	Food	[Zn] stable = ¹ 6.5 × 10 ⁻³ mg l ⁻¹	5000	= $CF(fish)$	1.0	11.8 ³	118.0	0.024
²⁴¹ Am	Food		1000	= $CF(fish)$	1.0	11.8 ³	23.6	0.024

1. Neal and Robson, 2000; 2. Tucker and Rasmussen (1999); 3. Elliott (1975a); 4. From data in Elliott (1975b).

Table D.10 Estimated uptake and excretion rates of radionuclides in fish at 17°C. All isotopes are assumed to be assimilated via the food pathway except strontium for which uptake via the gills is modelled and ³H for which no uptake mechanism is specified (measured uptake rates are used rather than an uptake mode)

Element	Pathway	Relevant water chemistry	$CF(fish)$ l kg ⁻¹	$CF(food)$ l kg ⁻¹	Assimilation efficiency, α	Feeding rate, w.w. g dy ⁻¹ @ 17 °C	Uptake rate k_f l kg ⁻¹ d ⁻¹ @ 17 °C	Excretion rate, k_b d ⁻¹ @ 17 °C
¹³⁷ Cs	Food	[K] 6.7 ¹ mg l ⁻¹	2×10^3	$= CF(fish)/2$	0.44 ²	23.8 ³	20.9	0.010
^{89, 90} Sr	Water	[Ca] 121 ¹ mg l ⁻¹ [Sr] 0.36 ¹ mg l ⁻¹ pH 8.1 ¹	60	N/A	N/A	N/A	0.68	0.011
^{125, 131} I	Food		40	$= CF(fish)$	1.0	23.8 ³	1.9	0.048
⁶⁰ Co	Food		3×10^2	$= CF(fish)$	0.1	23.8 ³	1.43	0.0048
³ H	N/A		1.0	1.0	N/A	N/A	0.69	0.69
³² P	Food	[SRP] stable = 0.91 ¹ mg l ⁻¹	1.0×10^4	$= CF(fish)$	1.0	23.8 ³	476	0.048
¹⁴ C	Food	DOC 4.4 ¹ mg l ⁻¹	2.2×10^4	$= CF(fish)$	0.14 ⁴	23.8 ³	147	0.0067
Pu, U	Food		50	$= CF(fish)$	1.0	23.8 ³	2.38	0.048
⁶⁵ Zn	Food	[Zn] stable = 6.5 × 10 ⁻³ mg l ⁻¹	5000	$= CF(fish)$	1.0	23.8 ³	238	0.048
²⁴¹ Am	Food		1000	$= CF(fish)$	1.0	23.8 ³	47.6	0.048

1. Neal and Robson, 2000; 2. Tucker and Rasmussen (1999); 3. Elliott (1975a); 4. From data in Elliott (1975b).

APPENDIX E: IDENTIFICATION OF MAJOR ENVIRONMENTAL VARIABLES

Using the Pangbourne reach as an example, the main environmental variables affecting radionuclide transport and uptake to fish are illustrated below.

E1 Effect of flow rate on radioactivity in water and fish

Figure E1 illustrates the effect of flow rate on integrated activity concentrations of ^{137}Cs in water (Bq days m^{-3}) and fish (Bq days kg^{-1}) for a site at the end of the Pangbourne reach. Parameter values are as follows: settling velocity, $v_p = 1 \text{ m d}^{-1}$; sorbed fraction $f_p = 0.05$; velocity of sediment movement = 27.4 m dy^{-1} ; river average width = 40 m ; discharge time = 30 min ; integration time = 1 month ; water temperature = 12°C .

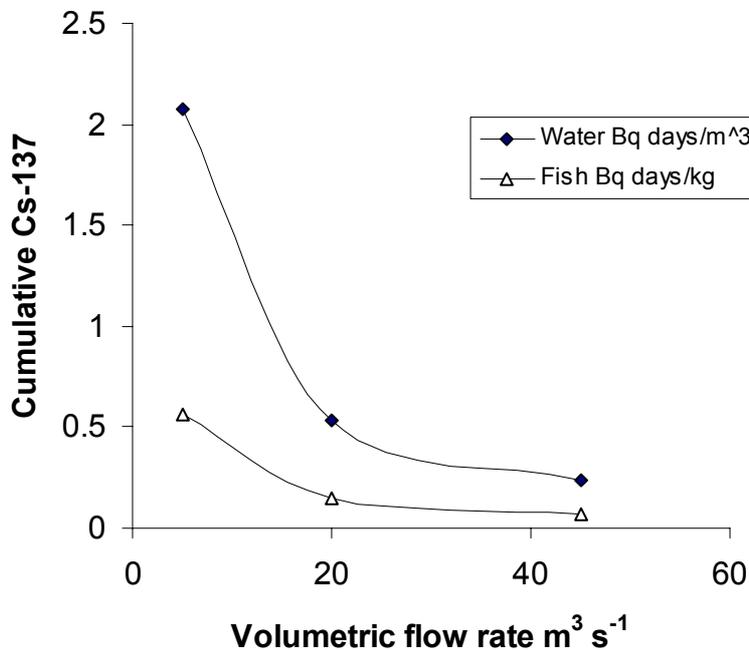


Figure E.1 Effect of volumetric flow rate on cumulative ^{137}Cs in water and fish

In general the radionuclide activity concentrations in the target variables are inversely proportional to volumetric flow rate.

E1.1 Maximum concentrations in water

Figure E2 shows maximum water activity concentrations for the case $f_p = 0$, input time = 30 min , and for radionuclides with decay times significantly longer than the travel time out of the 10 km stretch (i.e. $1/\lambda > 10,000/v$). In practice, these are good estimates of maximum concentrations for all the radionuclides being considered. Even for the fastest decaying radionuclide, ^{131}I , at the lowest water velocity, the over-estimation assuming no decay is only approximately 10% at the furthest site (Figure E.2). It can be seen from Figure E.2 that flow rate and distance from the discharge have a major influence on maximum concentrations at a given site.

The discharge time also has an important influence on maximum concentrations downstream. Figure E.3 shows maximum water activity concentrations for various discharge times under medium flow conditions.

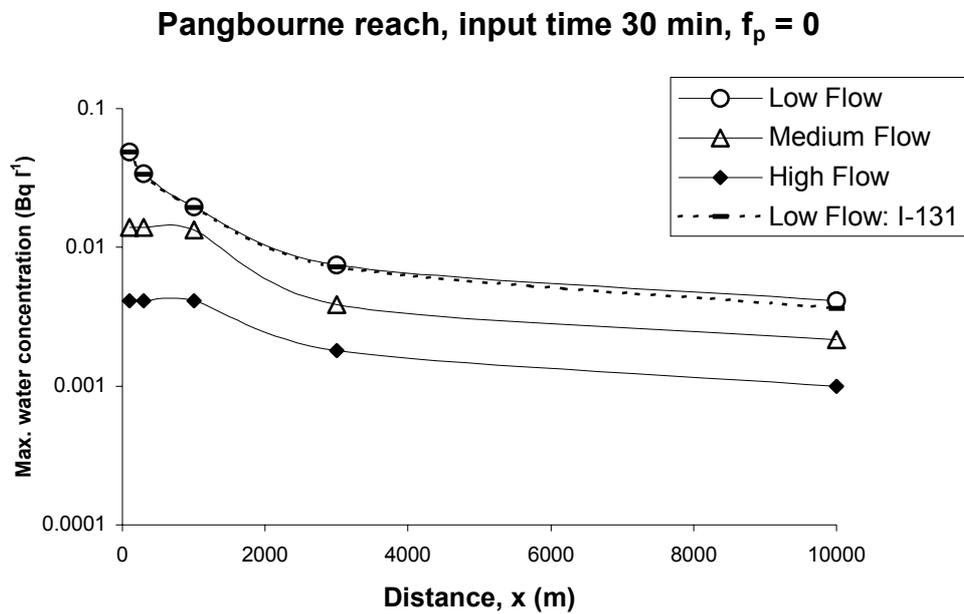


Figure E.2 Maximum water concentrations downstream of a 1 MBq release (over a 30 min period) from the assumed discharge point. The estimates apply to any radionuclide since in this scenario radioactive decay makes no significant difference to maximum concentrations

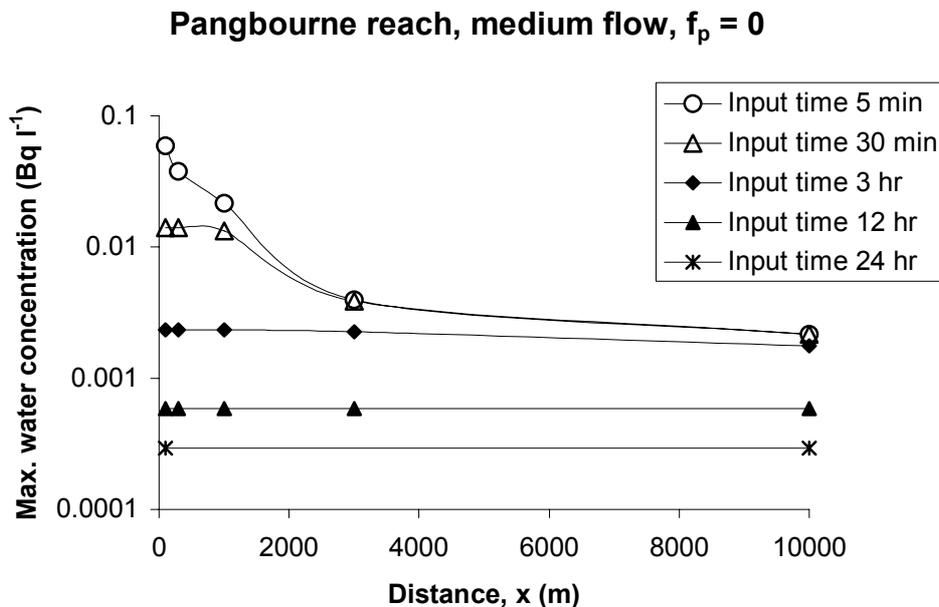


Figure E.3 Maximum water concentrations downstream of a 1 MBq release from the assumed discharge point over different time periods. Medium flow conditions are assumed

The influence of the particulate sorbed fraction f_p on maximum water activity concentrations was investigated for low flow conditions (under low flow conditions, particle settling would have the most influence on water activity concentrations). As illustrated in Figure E.4, particle sorption has relatively little influence on total activity concentrations in water (solid lines), but at high values ($f_p > 0.1$) makes a significant difference to dissolved-phase activity concentrations. Therefore, the dissolved phase activity concentrations for radionuclides with high particle affinity can be relatively simply estimated by estimating the maximum activity concentration (both suspended and dissolved phases) assuming no particle sorption and estimating the dissolved phase activity concentration from

$$C_w(\text{dissolved}) \approx C_w(\text{total}) \cdot (1 - f_p) \quad (\text{E1})$$

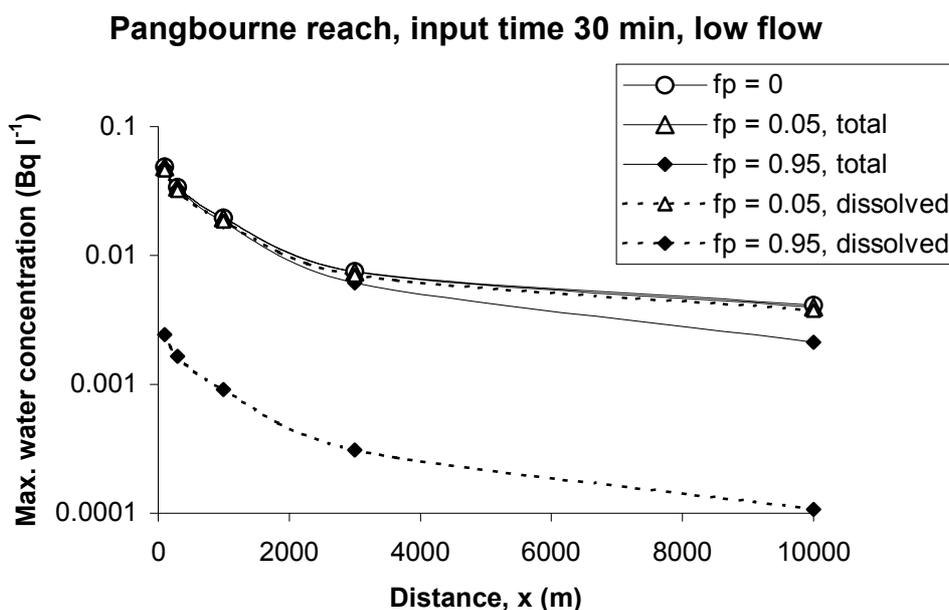


Figure E.4 Maximum water concentration as a function of particle sorbed fraction, f_p , for a 1 MBq release

E1.2 Integrated water concentration over a one year period

For the case of zero sorption of the radionuclide to suspended solids, Figure E.5 gives the activity concentration in water integrated over a one year period as a function of river volumetric flow rate. The integrated water concentration (for $f_p = 0$ and $\lambda = 0$) is not influenced by distance from the discharge point, if as in this case, volumetric flow rate is approximately constant over the study reach. Also, for ¹³¹I (the radionuclide with the highest decay rate), at low volumetric flow rate, the integrated water concentration at 10 km from the discharge point is only 12% lower than that for $\lambda = 0$. Thus, Figure 9 gives a relatively accurate upper limit for the integrated water concentration for all radionuclides we are studying.

E1.3 Maximum concentrations in fish

For the case of zero sorption of the radionuclide to suspended solids, Figure E.5 gives the maximum activity concentration of ^{137}Cs in fish as a function of river volumetric flow rate. The maximum activity concentration in fish for this scenario was independent of the distance from the discharge point.

E1.4 Integrated concentrations in fish

For the case of zero sorption of the radionuclide to suspended solids, Figure E.5 gives the activity concentration of ^{137}Cs in fish integrated over a one year period as a function of river volumetric flow rate. The integrated activity concentration for this scenario was independent of the distance from the discharge point.

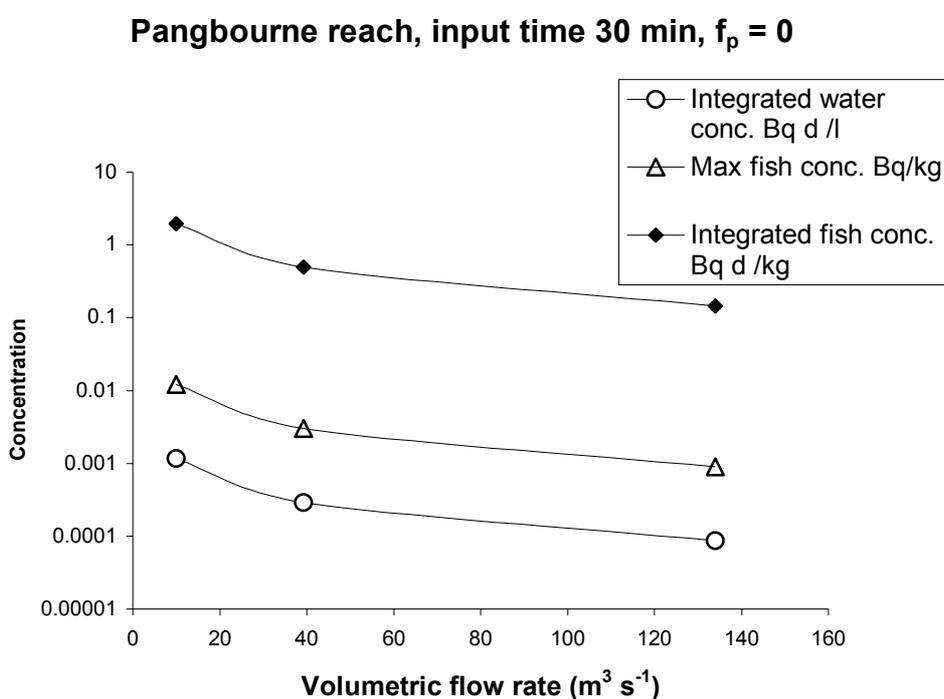


Figure E.5 Integrated water concentration, integrated fish concentration and maximum fish concentration as a function of volumetric flow rate for a 1 MBq release

E2 Seasonal variation in fish uptake rates via the food pathway

Rates of radionuclide uptake by fish are strongly influenced by feeding rates which in turn are strongly influenced by water temperature. As an example of the influence of seasonal variation on uptake rates, we have calculated ^{137}Cs , ^{32}P and ^{131}I activity concentrations in a 500 g fish at three different temperatures representative of the water temperature of the Thames during winter (7°C), spring or autumn (12°C), and summer (17°C). Figure E.6 shows the results of this analysis for a constant water temperature, and constant radionuclide concentration in water of 1 Bq l^{-1} . The activity concentrations of ^{137}Cs tend towards those predicted using the *CF* model, but for ^{131}I and ^{32}P the steady state concentrations are significantly lower than those predicted using the *CF* model. This is because the *CF* does not

account for radioactive decay of the radionuclide in the fish. From Equation C9 it can be seen that at steady-state:

$$C_f = \frac{k_f C_w}{k_b + \lambda} \quad (E2)$$

so in cases where λ is of the order of or greater than k_b (as for ^{131}I , ^{32}P), the CF model ($C_f = CF.C_w$) will over-estimate the steady state activity concentration. Note, however, that where CF has been determined from field measurements of a given radionuclide *in situ*, it is likely that decay is accounted for. The over-estimation is only likely to occur when a stable analogue or longer lived radionuclide is used to estimate the CF of a short-lived radionuclide.

It is likely that the uptake of radiostrontium via the gills will also be influenced by temperature, however this influence is not quantified in the model of Chowdhury and Blust (2001). The model was developed using experiments conducted at a water temperature of 25°C which is expected to give a maximum uptake rate: uptake rates in winter are therefore likely to be lower than our model predicts, however we cannot currently quantify this effect.

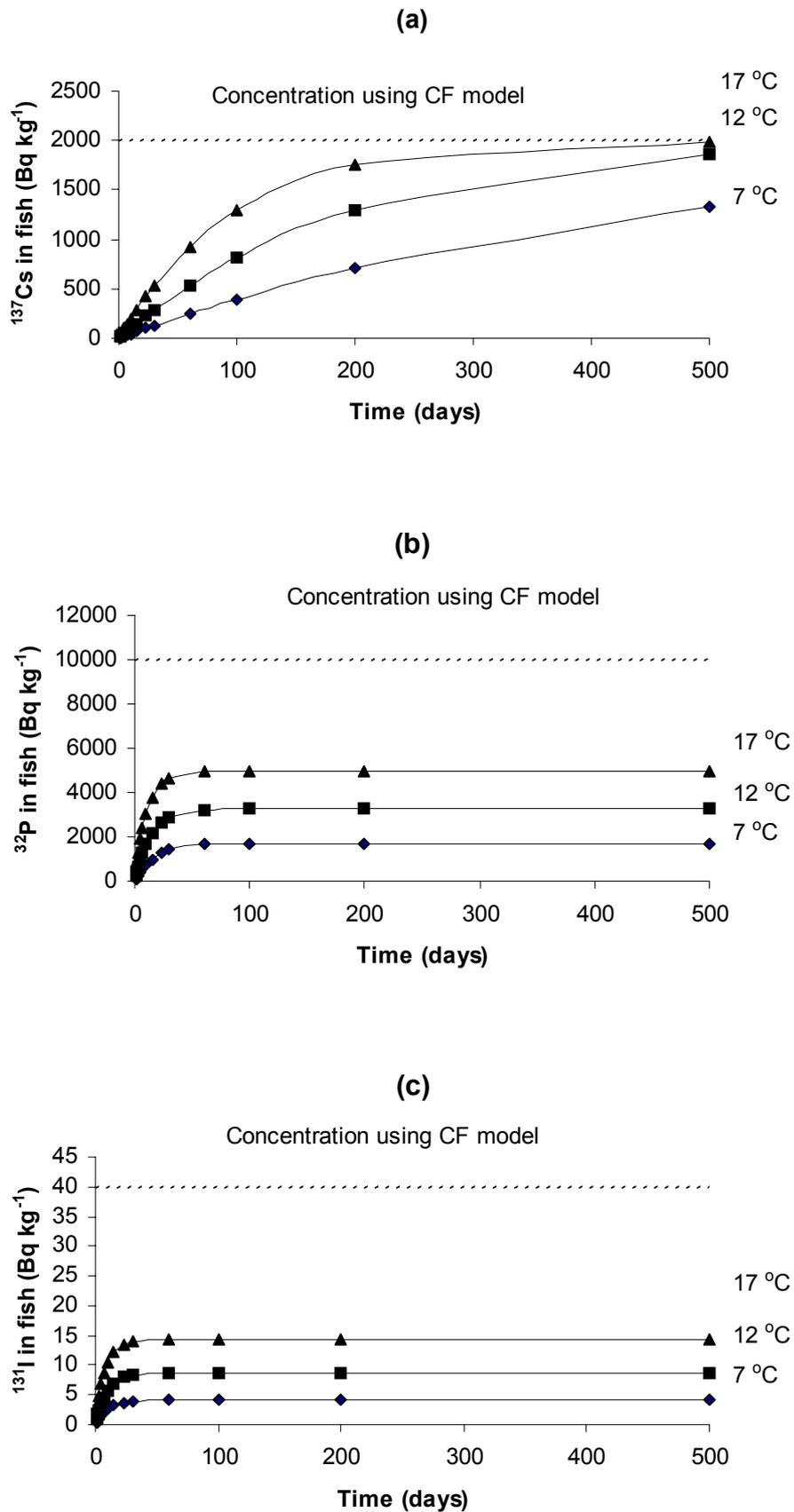


Figure E.6 Variation in uptake of (a) ¹³⁷Cs; (b) ³²P and (c) ¹³¹I in fish as a function of water temperature

APPENDIX F: TESTING THE NUMERICAL CODE

The numerical implementation of the model and the FORTRAN code was tested in the following ways:

1. Mass balance check;
2. Comparison of predicted activity concentration in water with analytic solution to the ADE for the case of zero transport of radioactivity to bottom sediments with delta-function discharge of radioactivity (i.e. instantaneous “spike” of radioactivity to the river);
3. Comparison of predicted activity concentration in water with analytic solution for the case of a continuous discharge of radioactivity to the river including transport of radioactivity to bottom sediments and radioactive decay;
4. Comparison of predicted concentration in sediment with analytic estimates;
5. Comparison of predicted concentration in fish with analytic estimates.

F1 Mass balance check

Conservation of mass is an important test of a numerical code. The model covering the initial phase (whilst the plume is in the 10 km reach) conserves mass: the sum total of radioactivity in the water and sediment of the 10 km reach plus that which has passed out of the reach plus losses by radioactive decay equals the sum total of radioactivity discharged. Conservation of mass in the numerical scheme and code was checked by summing these amounts and comparing them with the total amount input. In all cases mass was conserved to within 2% of the mass input, the slight discrepancy being due to rounding errors and truncation errors in the Taylor Series on which the explicit numerical solution is based. The error could be reduced by reducing the time and distance step size of the numerical solution (which would significantly increase the run time of the code) but it was decided that this was not necessary given the very small errors observed.

The model for the secondary (sediment transport) phase conserves mass in the sediment: the sum total of radioactivity in the sediment of the 10 km reach plus that which has passed out of the reach plus losses by radioactive decay equals that transferred to the sediment. A mass balance check was again carried out on the total amount of radioactivity in the sediment and mass was shown to be conserved to within 0.1% of mass input to the sediment. In the secondary phase the activity concentration in the water is approximated by assuming that it is equal to the amount of radioactivity resuspended at a give point per unit time, divided by the amount of water passing that point per unit time. Since the sediment is assumed to move by resuspension, the resuspended radioactivity is calculated from the rate of sediment transport. This method of calculating activity concentration in the water means that, during the secondary phase, the sediment being transported is effectively counted twice. Therefore, during the secondary phase the total amount of radioactivity is slightly over-estimated.

The conservation of mass checks showed no significant conservation of mass errors in the model. Therefore the results of this test are believed to be acceptable.

F2 Comparison with analytic solution for “spike” input

The numerical model was tested by comparison with an analytic solution of the ADE for the case of an instantaneous “spike” input of a conservative tracer to the river (Figure F.1).

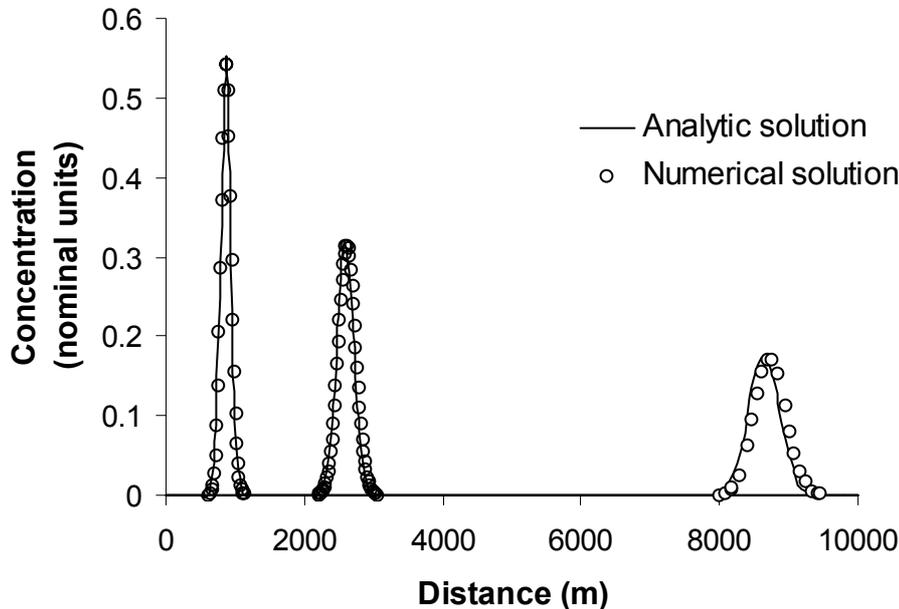


Figure F.1 Comparison of the numerical model with the analytic solution of the ADE for a “spike” input to a river of a conservative tracer for different times (0.01 day, 0.03 day and 0.1 day) after injection. Mean flow velocity, v was 1 m s^{-1} and dispersion coefficient, D , was $3 \text{ m}^2 \text{ s}^{-1}$

The results of this test showed excellent agreement between the numerical and analytic solutions.

F3 Comparison with analytic solution for transport to sediment for continuous discharge scenario

For a continuous discharge of radioactivity to the river, the transfer of radioactivity to sediments during the early phase results in an activity concentration in the river water which is constant over time. This activity concentration declines exponentially with distance downstream of the discharge point as a result of radioactive decay at rate λ , and transfer to bottom sediments at rate k_1 :

$$C_w(x) = C_w(0) \exp\left(-\frac{(\lambda + k_1)}{v}x\right) \quad (\text{F1})$$

where x is the distance downstream of the discharge point, λ is the radioactive decay constant, v is the water velocity and k_1 is the rate of transfer of radioactivity to sediments (Equation C.1). Note that this model is mathematically identical to that developed by Schaeffer, though

the sediment transfer constant is calculated differently. Comparison of the model with the Schaeffer model is carried out below.

We have compared the numerical code output to the above analytic solution for the case of no radioactive decay and assuming a very high radioactive decay rate ($\lambda = 0.69 \text{ d}^{-1}$, approximately 8 times higher than ^{131}I , the most rapidly decaying radionuclide considered in this study). The test of the code against the analytic solution is shown in Figure F.2, illustrating good agreement between analytic and numerical models.

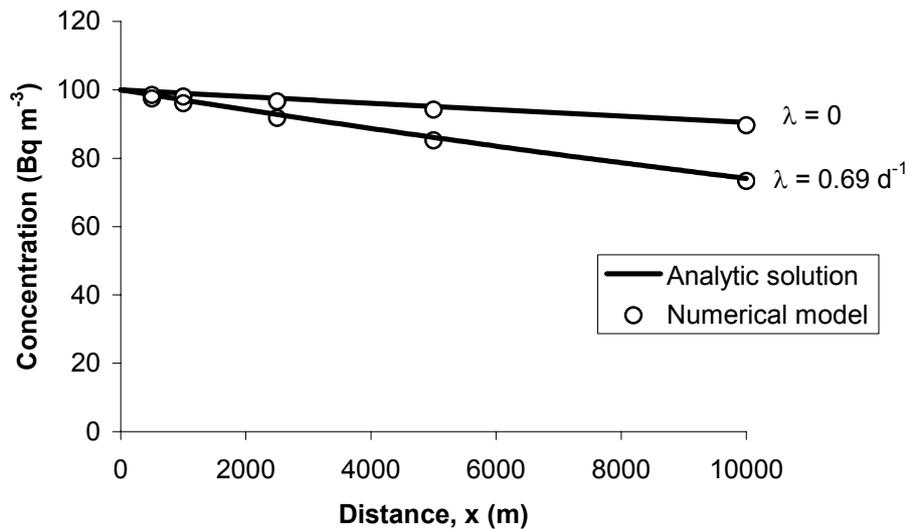


Figure F.2 Test of the numerical code against an analytic solution for continuous discharge and transfers of radioactivity to sediment. The test was carried out for the case of no radioactive decay and for a decay rate corresponding to a half-life of one day. The rate of transfer of radioactivity to sediments was calculated from Equation C1 with $f_p = 0.95$, $v_p = 1.0 \text{ m d}^{-1}$ and $d = 2.75 \text{ m}$ giving $k_I = 4.0 \times 10^{-6} \text{ s}^{-1}$

F3.1 Comparison with Schaeffer model for transport of radioactivity to bed sediments

The Schaeffer model (as described in Simmonds et al., 1995) for transport of radioactivity from river water to bed sediments was developed for modelling a continuous discharge of radioactivity to a river. The total activity concentration in the river water, $C_w(x)$, decreases exponentially with distance, x , downstream of the discharge:

$$C_w(x) = C_w(0) \exp\left(-\left(\frac{\lambda}{v} + k'\right)x\right) \quad (\text{F2})$$

where k' (m^{-1}) is an empirical factor, the ‘‘Schaeffer coefficient’’, which simulates the transfer of radioactivity to bed sediment. The Schaeffer model is mathematically identical to the model we have used (Equation F1), however the model parameterisation differs significantly. By comparing Equation F2 to Equation F1 we can relate the sedimentation parameter used in the present study to the Schaeffer coefficient:

$$k' = \frac{k_l}{v} = \frac{f_p v_p}{dv} \quad (\text{F3})$$

The key difference between the two models is that in the model presented here the rate of transfer of radioactivity to sediments (and therefore the decline in C_w) is inversely proportional to the river flow velocity.

Using Equation (F3) we can compare the removal rate constant, k_l in the model to recommended values of the Schaeffer coefficient. For the Pangbourne-Reading reach (Aldermaston pipeline), we compare these values in Table F.1. The parameter values are of the same order, though the particulate settling model estimates quite large variation in settling as volumetric flow rate changes, whereas the Schaeffer coefficient is independent of river flow velocity.

Table F.1 Illustrative comparison of the settling velocity model parameter values for the Thames (Reading-Pangbourne) with the Schaeffer coefficient model parameter values for the river Rhone (quoted in Simmonds et al. 1995). The Schaeffer coefficients were estimates for high K_d ($>10^5$ l kg⁻¹) and medium K_d ($>10^4$ l kg⁻¹). We have compared these with the settling velocity model ($v_p = 1$ m d⁻¹) for $f_p = 0.95$ and 0.05 respectively. For both models, low K_d radionuclides were assumed not to settle to bed sediments (i.e. $f_p = 0$, $k' = 0$)

Vol. flow rate m ³ s ⁻¹	River mean depth, m.	Mean water velocity, m s ⁻¹	Particulate sorbed fraction, []	Equivalent Schaeffer coeff. m ⁻¹	Schaeffer coeff. for Rhone m ⁻¹
High sediment-water distribution coefficient					
134	2.7	0.83		4.9×10^{-6}	
39.2	2.3	0.29	0.95	1.6×10^{-5}	1×10^{-5}
9.9	2.1	0.08		6.5×10^{-5}	
Medium sediment-water distribution coefficient					
134	2.7	0.83		2.6×10^{-7}	
39.2	2.3	0.29	0.05	8.7×10^{-7}	2×10^{-6}
9.9	2.1	0.08		3.4×10^{-6}	

F4 Comparison of concentration in sediment with analytic estimates

A further test of the numerical code was carried out using a simple model to simulate the activity concentration in the sediment within the first distance step of the numerical code. For the test scenario it was assumed that 1 MBq of radionuclide is released during a 30 minute period, and that the river parameters are those of the Pangbourne-Reading reach at medium flow (Table 4). The particle settling velocity was assumed to be 1 m d⁻¹ and the fraction of radioactivity sorbed to the solid phase, f_p , was 0.95.

The radioactivity concentration at the release point during the release (averaged over the river cross section and ignoring dispersion) is given by:

$$C_w = \frac{C_i}{Q.T_i} \quad (\text{F4})$$

where C_i is the total amount of radioactivity discharged, Q is the volumetric flow rate and T_i is the duration of the discharge. The maximum activity concentration in sediments (Bq kg^{-1}) may be estimated using:

$$C_s(\text{max}) = \frac{C_w \cdot k_l \cdot T_i \cdot d}{\rho_s d_s} \quad (\text{F5})$$

where d is the mean depth of the river and k_l is the rate of transfer of the radionuclide to the bed sediments. The change in activity concentration in the bed sediments within the first distance step of the numerical code may be estimated by:

$$C_s(t) = C_s(\text{max}) \cdot \exp\left[-\left(\frac{v_s}{\Delta x} + \lambda\right)t\right] \quad (\text{F6})$$

where Δx is the length of the first distance step of the numerical code and v_s is the rate of movement of bed sediment. For the case $\lambda = 0$ (no physical decay) and $v_s = 0$ (no sediment movement) the integrated activity concentration over a period τ (d) is simply $\tau \times C_s(\text{max})$ Bq d kg^{-1} . For the case λ and/or $v_s > 0$, the integrated activity concentration is given by the solution to the integral of Equation F6 which is:

$$\sum C_s(t) \approx \frac{C_s(\text{max})}{\frac{v_s}{\Delta x} + \lambda} \left[1 - \exp-\left(\frac{v_s}{\Delta x} + \lambda\right)\tau\right] \quad (\text{F7})$$

where it is assumed that the discharge time and travel time of the plume out of the first distance step of the numerical code is short in comparison with the integration time τ .

The numerical code was tested against these analytic estimates for three scenarios (Figure F.3) showing good agreement between the two models.

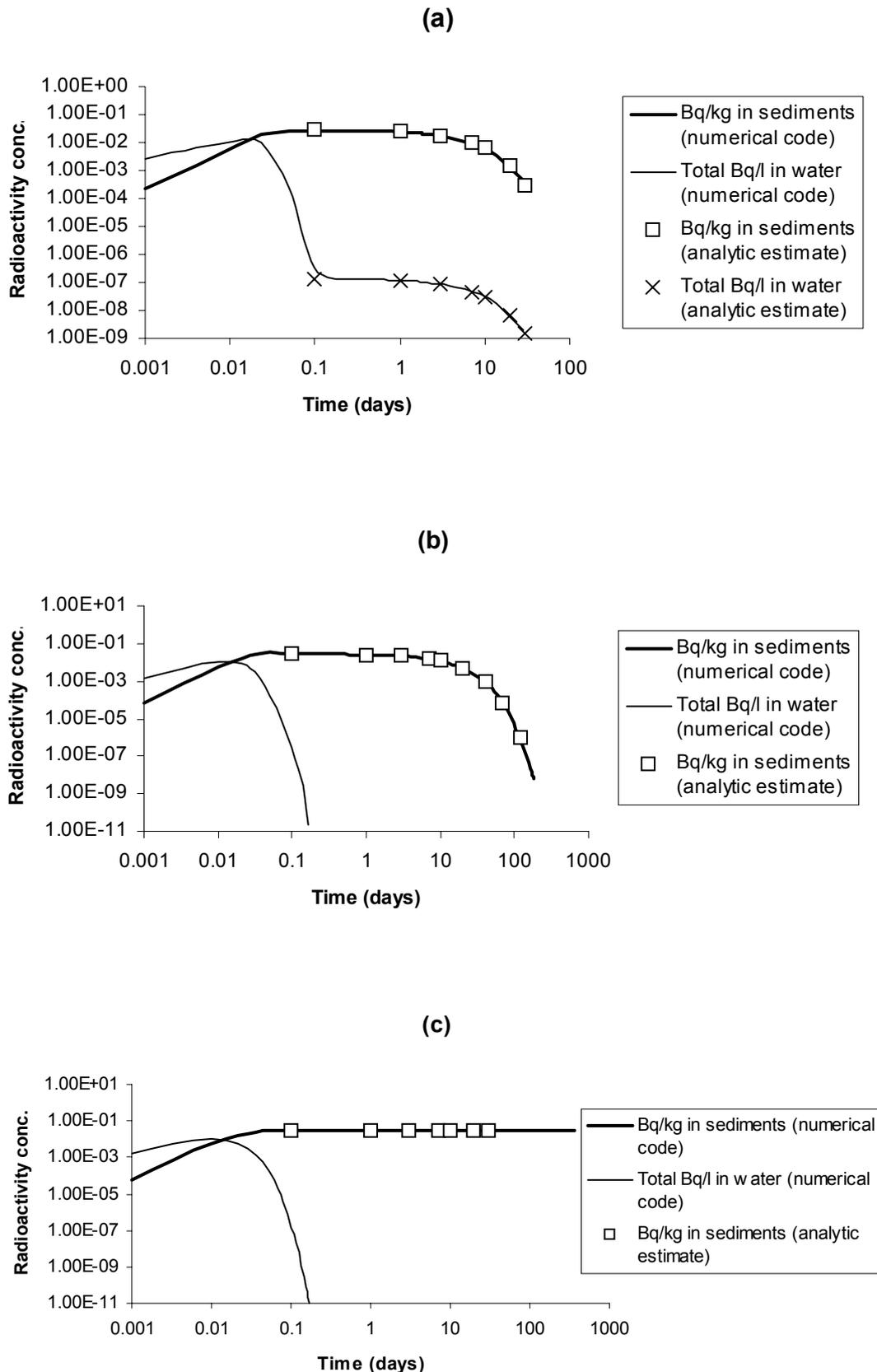


Figure F.3 Comparison of numerical code output with analytic estimate for activity concentrations in sediment during the sediment transport phase. The different graphs represent the following scenarios: (a) no radioactive decay, sediment velocity $v_s = 27.4 \text{ m d}^{-1}$; (b) no sediment movement, radioactive decay with $T_{1/2} = 8.05 \text{ d}$; (c) no sediment movement or radioactive decay

F5 Comparison of predicted concentration in fish with analytic estimates

The component of the numerical code simulating uptake and excretion of radionuclides in fish was tested against estimates using an analytical solution for a scenario in which radioactivity was discharged to the river at a constant rate for a three day period, after which no radioactivity was discharged. During the discharge phase, the activity concentration in the fish in the first distance step of the numerical code (i.e. close to the discharge) can be approximated by Equation C11. This equation describes uptake and excretion of radioactivity in fish for a constant activity concentration in water. Following the discharge phase, it was assumed that the activity concentration in water was zero and therefore the activity concentration in fish declined exponentially with rate constant $(k_b + \lambda)$. The comparison of the numerical code output with this analytical estimate is shown in Figure F.6, showing good agreement. The estimate of fish activity concentration using a concentration factor approach (i.e. $C_f = CF \cdot C_w$) is also shown in Figure F.6 illustrating the difference between the dynamic and equilibrium models.

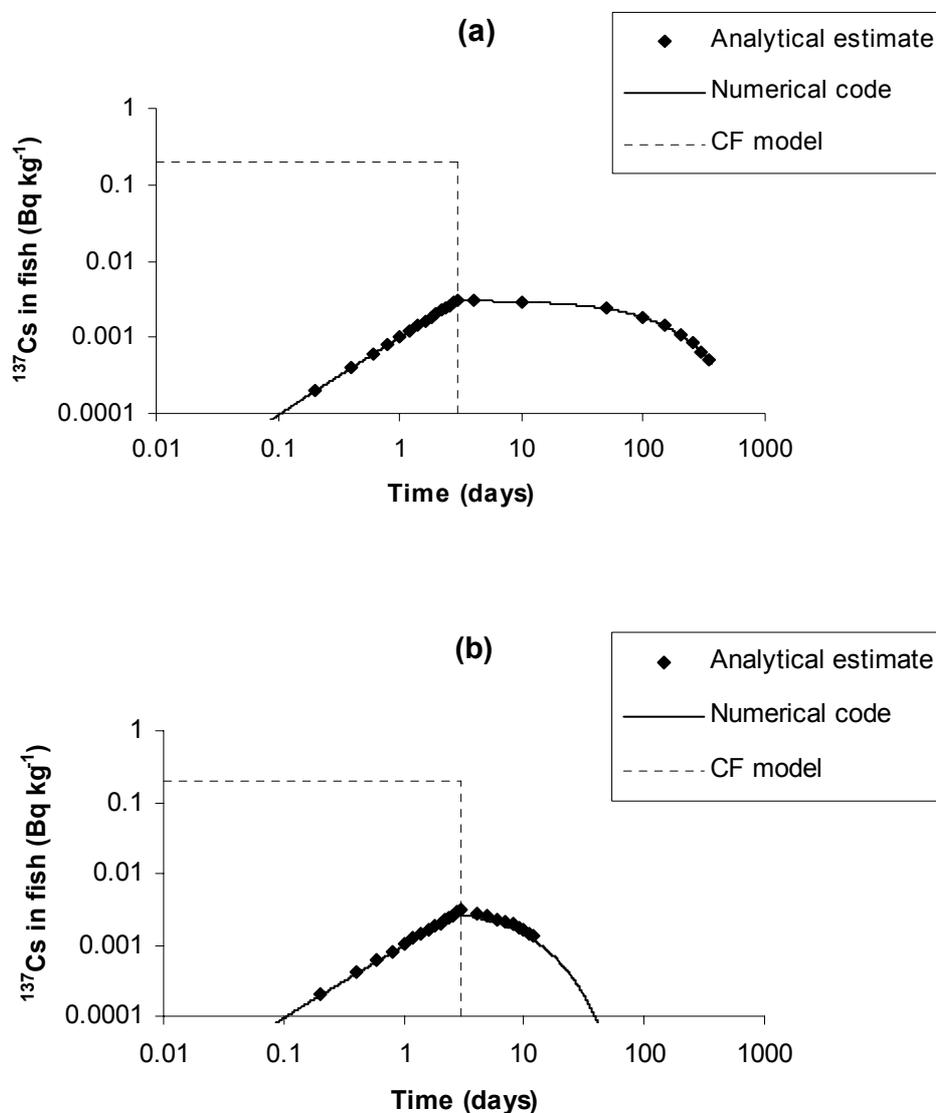


Figure F.6 Comparison of numerical code output with analytic estimate for activity concentrations in fish. The different graphs represent the following scenarios: (a) no radioactive decay; (b) radioactive decay with $T_{1/2} = 8.05$ d

APPENDIX G: SIMPLIFYING THE MODEL

G1 Simplifying the model

G1.1 Parameters of the simplified model

C_T	Total radionuclide activity concentration in water	Bq l ⁻¹
C_w	Radionuclide activity concentration in dissolved phase of water	Bq l ⁻¹
C_p	Radionuclide activity concentration in particulate phase of water	Bq l ⁻¹
C_s	Radionuclide activity concentration in bed sediments	Bq kg ⁻¹
Q	Volumetric flow rate of river	m ³ s ⁻¹
k_i	Rate of transfer of radionuclides to bed sediments	s ⁻¹
λ	Radionuclide decay constant	s ⁻¹
T_i	Duration of release	s

Estimates of pollutant maximum and integrated concentrations may be made using relatively simple relationships between the key model parameters. The simplified relationships outlined below have been tested against the output of the numerical model.

G1.2 Estimating maximum water concentrations

The maximum dissolved-phase water concentration (expressed as a cross-section average), $C_w^{\max}(x)$ (Bq l⁻¹) for a site close to the discharge point may be estimated from the amount of radioactivity discharged, C_i (Bq), the time over which the discharge takes place, T_i (s), and the volumetric flow rate, Q (m³ s⁻¹):

$$C_w^{\max}(x) \approx \frac{C_i \times 10^{-3}}{Q.T_i} (1 - f_p) \quad (\text{G1})$$

where the factor 10^{-3} changes the units of concentration from Bq m⁻³ to Bq l⁻¹.

The maximum activity concentration (total and dissolved phase) at distance x downstream of the discharge may be estimated from the following relationships:

$$C_T^{\max}(x) \approx \frac{C_i \times 10^{-3}}{Q.T_i} \operatorname{erf}\left(\frac{T_i v}{4\sqrt{Dx/v}}\right) \exp\left[-\frac{(\lambda + k_1)}{v} x\right]$$

and

$$C_w^{\max}(x) \approx (1 - f_p) C_T^{\max}(x) \quad (\text{G2})$$

where erf is the error function. The exponential term accounts for radioactive decay and transfers to sediments as the plume moves downstream.

G1.3 Estimating integrated water concentrations

The integrated (infinite time integral, approximately equal to the 1 year integral since the plume travel time is $\ll 1$ year) total, $\Sigma C_T(x)$ (Bq d l⁻¹), and dissolved-phase water concentrations (expressed as a cross-section average), $\Sigma C_w(x)$ (Bq d l⁻¹) may be estimated from the amount of radioactivity discharged, C_i (Bq), and the volumetric flow rate, Q (m³ s⁻¹):

$$\Sigma C_T(x) \approx \frac{C_i \times 10^{-3}}{Q \cdot 24 \cdot 3600} \exp\left[-\frac{\lambda x}{v}\right]$$

and

$$\Sigma C_w(x) \approx \Sigma C_T(x)(1 - f_p) \quad (G3)$$

where the factor 10^{-3} changes the units of concentration from Bq m⁻³ to Bq l⁻¹ and the factor 24×3600 changes the units of volumetric flow rate from seconds to days. No correction is made for transfer of radioactivity to bed sediments, leading to an over-estimate of activity concentrations. Given the relatively low potential transport rates to sediments within the 10 km stretch, this slightly conservative assumption is appropriate for estimating radioactivity in the water and for transfers to fish.

If it is assumed that there is significant uptake of radioactivity to sediments, and subsequent delay in the sediments we can use the following relations for estimating the integrated activity concentration in water:

$$\Sigma C_T(x) \approx \frac{C_i \times 10^{-3}}{Q \cdot 24 \cdot 3600} \exp\left[-\frac{(\lambda + k_1)x}{v}\right] \quad (G4)$$

and

$$\Sigma C_w(x) \approx \Sigma C_T(x)(1 - f_p) \quad (G5)$$

The latter equations (Equations G4, G5) are appropriate where there is significant uptake to sediments and the radioactivity in the sediments is not expected to be remobilised within the integration time period (in this case one year), or within the period of radioactive decay. The former condition may apply in slow flowing rivers which are dredged (i.e. little scouring of the river bed). The latter scenario may apply to short-lived radionuclides which are absorbed by sediments since the isotope may decay before it is remobilised from the sediment.

In the present study, to estimate activity concentrations in water and fish, we will conservatively use Equations G3, i.e. assuming no uptake to sediments.

G1.4 Estimating sediment concentrations

The maximum activity concentration in bottom sediments may be estimated from the transfer rate of the radionuclide to the sediment and assuming no sediment movement:

$$C_s^{\max}(x) = \frac{1 \times 10^3 k_l \cdot 24.3600 \cdot d \cdot \Sigma C_T(x)}{\rho_s d_s} \quad (\text{G6})$$

where the factor 1×10^3 converts the units of $\Sigma C_T(x)$ from Bq d l⁻¹ to Bq d m⁻³ and the factor 24.3600 converts the units of k_l from s⁻¹ to d⁻¹ (this conversion is not required in the exponential term). In the above Equation (G6), $\Sigma C_T(x)$ is calculated using Equation (G4), i.e. for the case of uptake and no subsequent movement of bottom sediments.

Again assuming no release of radionuclides from the bed sediment, the integrated activity concentration in the sediment $\Sigma C_s(x)$ (Bq d kg⁻¹) may be approximated by:

$$\Sigma C_s(x) \approx \frac{C_s^{\max}(x)}{\lambda \cdot 24.3600} [1 - \exp(-\lambda \cdot 24.3600 \cdot \tau)] \quad (\text{G7})$$

where τ is the integration time (in this case 365 days).

G1.5 Estimating fish concentrations

Where the excretion rate of radionuclides in fish has a significantly longer time scale than the travel time of the plume, i.e.:

$$3600.24 / k_b \gg L / v \quad (\text{G8})$$

(where 3600.24 changes k_b to units of s⁻¹, L is the reach length, 10,000 m in this case, and v is the flow velocity), the maximum activity concentration in fish may be estimated by:

$$C_f^{\max}(x) \approx \Sigma C_w(x) \cdot k_f \quad (\text{G9})$$

(i.e. it is assumed that there is only accumulation and no excretion). The excretion rate condition (Equation G6), for the scenarios considered here, holds for all radionuclides except tritium, for which a concentration factor approach may be more appropriate.

Given the above condition, the integrated activity concentration in fish, $\Sigma C_f(x)$, may be estimated by:

$$\Sigma C_f(x) \approx \frac{\Sigma C_w(x) \cdot k_f}{k_b + \lambda \cdot 24.3600} (1 - \exp(-(k_b + \lambda \cdot 24.3600) \tau)) \quad (\text{G10})$$

where τ is the time (days) over which the concentration is to be integrated.

G1.6 Comparison with CF approach

Under the concentration factor approach, the maximum concentration in fish is given by:

$$C_f^{\max}(x) \approx C_w^{\max}(x).CF \quad (G11)$$

which leads to significant over-estimates of the maximum activity concentrations in fish for short term releases.

The integrated activity concentration in fish is estimated by the *CF* approach using:

$$\Sigma C_f(x) \approx \Sigma C_w(x).CF \quad (G12)$$

This usually only slightly over-estimates integrated activity concentrations since, for long integration times ($\tau \times (k_b + \lambda) \gg 1$) and $k_b \gg \lambda$ so Equation (G10) tends to Equation (G12). The concentration factor approach will be used for H-3.

G2 Testing the simplified model

The simplified model output was compared with that of the numerical model for 55 model runs at different sites under different environmental conditions and for different radionuclides. The comparison showed good agreement between the simplified model and the numerical model (Figure G.1). Concentrations of H-3 in fish are over-estimated by the simplified model since the concentration factor approach was conservatively used for this radionuclide.

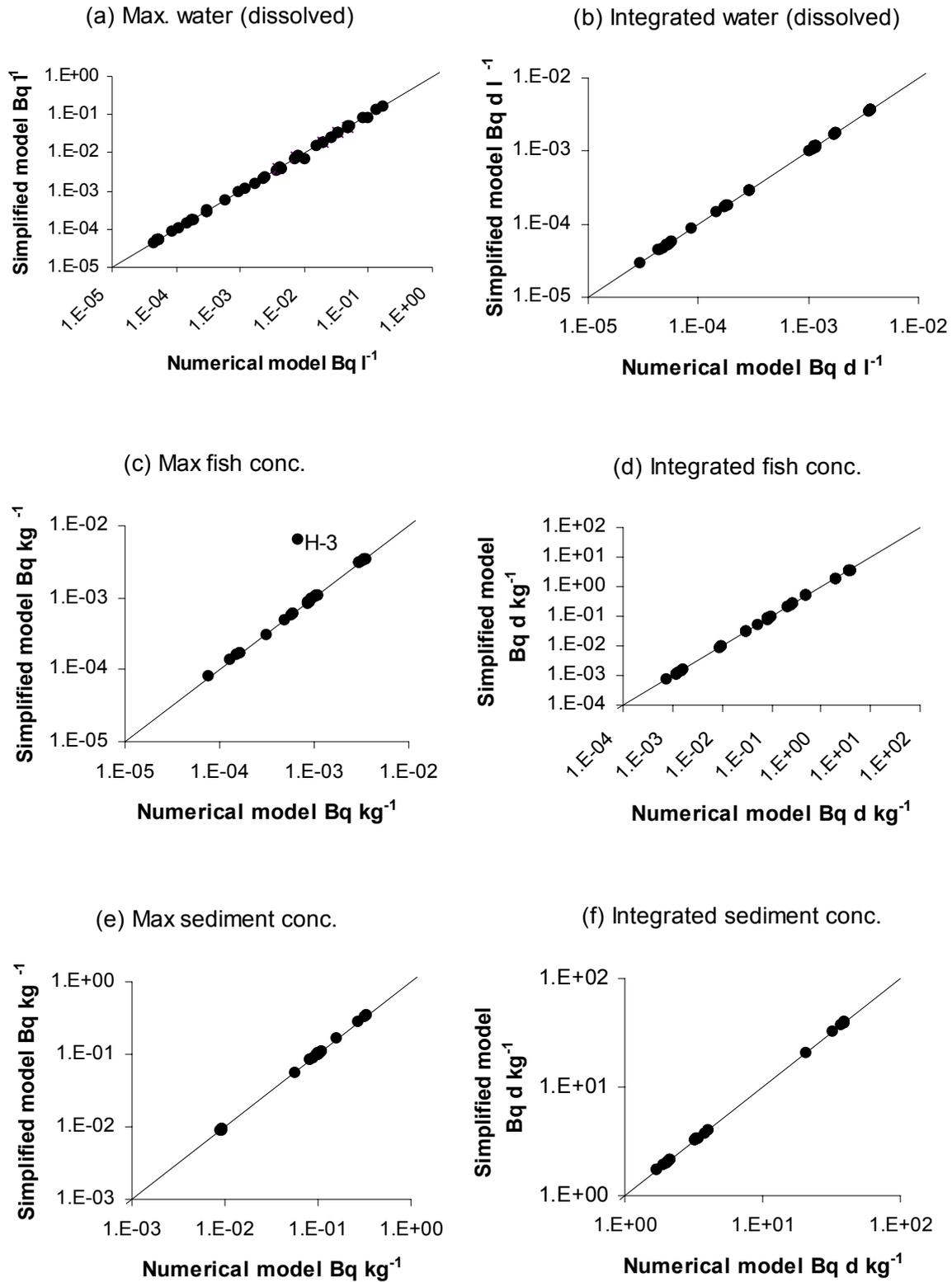


Figure G.1 Comparison between the simplified and numerical model output for different target variables (a) – (f). Lines show 1:1 correspondence

APPENDIX H: TABULATED OUTPUT

Tables are presented below of instantaneous concentrations and time integrated concentrations at different times (1 day – 50 years) after the release. Activity concentrations are given for filtered river water, total water (filtered plus unfiltered), fish at 12°C and sediment, for a hypothetical 1 MBq release to a river flowing at 10 m³/s, release duration of 3 hours, 1 km downstream.

The total and filtered water concentration and the bed sediment concentration may be estimated from Tables H.1 to H.7 depending on the site being considered and the fraction of radioactivity sorbed to particulates, given in Table 8 of the main text (reproduced here as Table H.8). It is recommended that the assumed f_p value is that given for modelling bed sediments (i.e. the upper estimate of f_p given in Table H.8). The values in Tables H.1 to H.7 apply to all radionuclides of half life >5 years (according to their f_p value). For radionuclides of half-life <5 years the values given in the tables must be corrected using the formulae given in Table H.9. For instantaneous activity concentrations in water, upper limits are given for activity concentrations resulting from resuspension of all of the bed sediment during a single 24 hour duration flood event.

Activity concentrations in fish are given in Tables H.10 – H.12.

The values in the tables were estimated as follows:

Total water concentrations. After one day the plume has in all cases passed the 1 km site. Residual radioactivity in the water will therefore only be due to resuspension of radioactivity from the bed sediments. Because of the random nature of resuspension events, it is not possible to predict instantaneous water concentrations after the passage of the contaminant plume with a reasonable degree of accuracy. An upper limit to the instantaneous water concentration will therefore be estimated. As a worst case scenario, it will be assumed that all of the radioactivity deposited on the bed sediment upstream of the 1 km reach is resuspended during a single 24 h duration flood event where volumetric flow rate is at the upper 10 percentile value for the river. The instantaneous total water concentration would then be:

$$C_T(1000) = \frac{1000.W.C_s(\max)\rho_s d_s e^{-\lambda T}}{24.3600.Q.1000} \text{ Bq l}^{-1}$$

where the value 1000 in the numerator is the distance x downstream and the value 1000 in the denominator is to convert the units from m⁻³ to l⁻¹. W is the river width and T is the time since discharge of the plume (i.e. 1 day, 1 week, 1 month, etc.).

It is further assumed that after a period of one year, all of the radioactivity will have been removed from the sediment by resuspension or dredging.

Filtered water concentrations. Calculated from the total water concentrations using the fraction of radioactivity sorbed to suspended particulates $C_w = C_T \times (1 - f_p)$.

Bed sediment concentrations. Calculated from the maximum activity concentration in bed sediments corrected for radioactive decay. It is assumed that there is no resuspension for a

time period up to one year. After a one year period it is also assumed that all of the radioactivity on the bed sediments is removed by resuspension or dredging.

Activity concentrations in fish. Calculated assuming no sorption of radionuclides to suspended and bed sediments. Maximum activity concentrations in fish occur during the first day following the discharge, then decline according to the rate of excretion of the radionuclide and its decay constant.

Table H.1 Water and sediments, all three study reaches: $f_p = 0.0$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	0	1.1×10^{-3}	0	1.1×10^{-3}	0	0
1 week	0	1.1×10^{-3}	0	1.1×10^{-3}	0	0
1 month	0	1.1×10^{-3}	0	1.1×10^{-3}	0	0
1 year	0	1.1×10^{-3}	0	1.1×10^{-3}	0	0
10 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	0
50 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	0

Table H.2 Water and sediments Pangbourne reach: $f_p = 0.05$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides, half life > 5 y

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	$< 2.9 \times 10^{-7}$	1.1×10^{-3}	$< 3.0 \times 10^{-7}$	1.1×10^{-3}	5.8×10^{-3}	5.8×10^{-3}
1 week	$< 2.9 \times 10^{-7}$	1.1×10^{-3}	$< 3.0 \times 10^{-7}$	1.1×10^{-3}	5.8×10^{-3}	4.06×10^{-2}
1 month	$< 2.9 \times 10^{-7}$	1.1×10^{-3}	$< 3.0 \times 10^{-7}$	1.1×10^{-3}	5.8×10^{-3}	0.176
1 year	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12
10 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12
50 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12

Table H.3 Water and sediments GUC/Colne reach: $f_p = 0.05$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides of half life > 5 y

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	$< 1.2 \times 10^{-6}$	1.1×10^{-3}	$< 1.3 \times 10^{-6}$	1.1×10^{-3}	5.8×10^{-3}	5.8×10^{-3}
1 week	$< 1.2 \times 10^{-6}$	1.1×10^{-3}	$< 1.3 \times 10^{-6}$	1.1×10^{-3}	5.8×10^{-3}	4.06×10^{-2}
1 month	$< 1.2 \times 10^{-6}$	1.1×10^{-3}	$< 1.3 \times 10^{-6}$	1.1×10^{-3}	5.8×10^{-3}	0.176
1 year	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12
10 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12
50 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12

Table H.4 Water and sediments, Sutton Courtney reach: $f_p = 0.05$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides half life > 5 years

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	$< 2.9 \times 10^{-7}$	1.1×10^{-3}	$< 3.1 \times 10^{-7}$	1.1×10^{-3}	5.8×10^{-3}	5.8×10^{-3}
1 week	$< 2.9 \times 10^{-7}$	1.1×10^{-3}	$< 3.1 \times 10^{-7}$	1.1×10^{-3}	5.8×10^{-3}	4.06×10^{-2}
1 month	$< 2.9 \times 10^{-7}$	1.1×10^{-3}	$< 3.1 \times 10^{-7}$	1.1×10^{-3}	5.8×10^{-3}	0.176
1 year	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12
10 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12
50 years	0	1.1×10^{-3}	0	1.1×10^{-3}	0	2.12

Table H.5 Water and sediments, Pangbourne reach: $f_p = 0.95$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides half life > 5 y

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	$< 2.8 \times 10^{-7}$	5.5×10^{-5}	$< 5.7 \times 10^{-6}$	1.1×10^{-3}	1.1×10^{-1}	1.1×10^{-1}
1 week	$< 2.8 \times 10^{-7}$	5.5×10^{-5}	$< 5.7 \times 10^{-6}$	1.1×10^{-3}	1.1×10^{-1}	7.7×10^{-1}
1 month	$< 2.8 \times 10^{-7}$	5.5×10^{-5}	$< 5.7 \times 10^{-6}$	1.1×10^{-3}	1.1×10^{-1}	3.3
1 year	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2
10 years	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2
50 years	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2

Table H.6 Water and sediments GUC/Colne reach: $f_p = 0.95$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides of half life > 5 y

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	$< 1.2 \times 10^{-6}$	5.5×10^{-5}	$< 2.4 \times 10^{-5}$	1.1×10^{-3}	1.1×10^{-1}	1.1×10^{-1}
1 week	$< 1.2 \times 10^{-6}$	5.5×10^{-5}	$< 2.4 \times 10^{-5}$	1.1×10^{-3}	1.1×10^{-1}	7.7×10^{-1}
1 month	$< 1.2 \times 10^{-6}$	5.5×10^{-5}	$< 2.4 \times 10^{-5}$	1.1×10^{-3}	1.1×10^{-1}	3.3
1 year	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2
10 years	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2
50 years	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2

Table H.7 Water and sediments Sutton Courtney reach: $f_p = 0.95$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. All radionuclides of half life > 5 years

Time	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
1 day	$< 2.9 \times 10^{-7}$	5.5×10^{-5}	$< 5.8 \times 10^{-6}$	1.1×10^{-3}	1.1×10^{-1}	1.1×10^{-1}
1 week	$< 2.9 \times 10^{-7}$	5.5×10^{-5}	$< 5.8 \times 10^{-6}$	1.1×10^{-3}	1.1×10^{-1}	7.7×10^{-1}
1 month	$< 2.9 \times 10^{-7}$	5.5×10^{-5}	$< 5.8 \times 10^{-6}$	1.1×10^{-3}	1.1×10^{-1}	3.3
1 year	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2
10 years	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2
50 years	0	5.5×10^{-5}	0	1.1×10^{-3}	0	40.2

Table H.8 Short term K_d (l kg⁻¹) values and fraction of radionuclide sorbed to solid phase assuming a suspended solids concentration, s , of 13.0 mg l⁻¹

Element	Best estimate K_d [l kg ⁻¹]	Fraction in solid phase []	Recommended value for model (water, fish)	Recommended value for model (bed sediment)
³ H	1	0.00001	0	0
¹⁴ C	1×10^4	0.12	0	0.95
³² P	5.7×10^3	0.07	0	0.95
Cs	5×10^3	0.04	0	0.95
Sr	10^2	0.0013	0	0.05
Zn	5×10^2	0.0065	0	0.05
I	50	0.00065	0	0.05
Co	10^3	0.013	0	0.05
U	50	0.00065	0	0.05
Pu	10^3	0.013	0	0.05
Am	5×10^3	0.04	0	0.95

Table H.9 Conversion formulae for radionuclides of half life <5 years. After 1 year instantaneous activity concentration is assumed to be zero

	Filtered water concentrations		Total water concentrations		Bed sediment	
	Instantaneous (Bq/l)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq l ⁻¹)	Time integrated (Bq d l ⁻¹)	Instantaneous (Bq kg ⁻¹)	Time integrated (Bq d kg ⁻¹)
Formula	Multiply by	Unchanged	Multiply by	Unchanged	Multiply by	Multiply by
	$\exp(-\lambda\tau)$		$\exp(-\lambda\tau)$		$\exp(-\lambda\tau)$	$\frac{1 - \exp(-\lambda\tau)}{\lambda\tau}$ for
	for times 1 d < τ < 1 yr		for times 1 d < τ < 1 yr		for times > 1 d	times 1 d < τ < 1 yr

Table H.10 Max. activity concs. in fish all three study sites: $f_p = 0.0$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$

Radionuclide	Fish at 12 ⁰ C
	Maximum (Bq/kg)
P-32	0.26
Zn-65	0.13
C-14	8.0×10^{-2}
Am-241	2.6×10^{-2}
Cs-134,137	1.1×10^{-2}
Co-60,I-125,131,Pu,U	1.3×10^{-3}
Sr-89,90	8.0×10^{-4}

Table H.11 Instantaneous activity concentrations in fish, all three study sites: $f_p = 0.0$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. For H-3 values are the same as instantaneous activity concentrations in water

Time	Instantaneous activity concentrations in fish at 12^0C , Bq kg^{-1}											
	^{137}Cs	^{134}Cs	^{32}P	^{125}I	^{131}I	^{89}Sr	^{90}Sr	^{60}Co	^{14}C	Pu, U	^{65}Zn	^{241}Am
1 day	1.14×10^{-2}	1.14×10^{-2}	2.60×10^{-1}	1.03×10^{-3}	1.03×10^{-3}	7.48×10^{-4}	7.48×10^{-4}	7.81×10^{-4}	8.00×10^{-2}	1.30×10^{-3}	1.30×10^{-1}	2.60×10^{-2}
1 week	1.10×10^{-2}	1.10×10^{-2}	1.56×10^{-1}	8.06×10^{-4}	4.79×10^{-4}	6.32×10^{-4}	6.92×10^{-4}	7.66×10^{-4}	7.81×10^{-2}	1.10×10^{-3}	1.08×10^{-1}	2.19×10^{-2}
1 month	9.75×10^{-3}	9.50×10^{-3}	2.87×10^{-2}	3.50×10^{-4}	3.66×10^{-5}	3.60×10^{-4}	5.34×10^{-4}	7.18×10^{-4}	7.23×10^{-2}	6.26×10^{-4}	5.74×10^{-2}	1.25×10^{-2}
1 year	1.68×10^{-3}	1.23×10^{-3}	0	2.31×10^{-9}	0	1.13×10^{-7}	1.32×10^{-5}	2.85×10^{-4}	2.40×10^{-2}	2.04×10^{-7}	7.231×10^{-6}	4.07×10^{-6}
10 years	0	0	0	0	0	0	0	3.29×10^{-8}	4.69×10^{-7}	0	0	0
50 years	0	0	0	0	0	0	0	0	0	0	0	0

Table H.12 Time integrated activity concentrations in fish, all three study sites: $f_p = 0.0$, release duration 3 hours, 1 km downstream, $Q = 10 \text{ m}^3 \text{ s}^{-1}$. For H-3, values are the same as time integrated activity concentrations in water

Time	Time integrated activity concentrations in fish at 12^0C , Bq d kg^{-1}											
	^{137}Cs	^{134}Cs	^{32}P	^{125}I	^{131}I	^{89}Sr	^{90}Sr	^{60}Co	^{14}C	Pu, U	^{65}Zn	^{241}Am
1 day	1.14×10^{-2}	1.14×10^{-2}	0.25	1.02×10^{-3}	9.79×10^{-4}	7.39×10^{-4}	7.44×10^{-4}	7.80×10^{-4}	7.98×10^{-2}	1.28×10^{-3}	0.13	2.57×10^{-2}
1 week	7.86×10^{-2}	7.84×10^{-2}	1.43	6.41×10^{-3}	5.05×10^{-3}	4.82×10^{-3}	5.04×10^{-3}	5.41×10^{-3}	0.55	8.36×10^{-3}	0.83	0.17
1 month	0.32	0.32	3.19	1.92×10^{-2}	9.07×10^{-3}	1.61×10^{-2}	1.93×10^{-2}	2.28×10^{-2}	2.31	2.80×10^{-2}	2.70	0.56
1 year	1.86	1.67	3.58	2.90×10^{-2}	9.41×10^{-3}	3.10×10^{-2}	6.64×10^{-2}	0.18	16.97	5.41×10^{-2}	4.84	1.08
10 years	2.17	1.87	3.58	2.90×10^{-2}	9.41×10^{-3}	3.10×10^{-2}	6.76×10^{-2}	0.28	24.23	5.41×10^{-2}	4.84	1.08
50 years	2.17	1.87	3.58	2.90×10^{-2}	9.41×10^{-3}	3.10×10^{-2}	6.76×10^{-2}	0.28	24.23	5.41×10^{-2}	4.84	1.08