Modelling the dispersion of radionuclides following short duration releases to rivers: Part 1. Water and sediment.

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ABSTRACT

This paper evaluates and generalises state-of-the-art approaches for modelling short duration liquid discharges of radionuclides (^3H , ^{14}C , ^{60}Co , ^{134}Cs , ^{137}Cs , ^{65}Zn , ^{89}Sr , ^{90}Sr , ^{125}I , ^{131}I , ^{241}Am , isotopes of Pu and U) to rivers. An advection-dispersion model was parameterised and used to predict the concentrations of radionuclides in the river environment, i.e. in river water, river bed sediment and fish (Part II of this paper covers uptake to fish). The coupled transport and bio-uptake model was used to predict the concentrations of radionuclides in the River Thames, UK, and one of its tributaries as a result of hypothetical short duration discharges. A simplified version of this model was developed and presented as “look-up” graphs. The influence of various environmental parameters on model output was evaluated by sensitivity analysis. Time-integrated water and sediment concentrations and maximum sediment concentrations may be predicted for all rivers on the basis of the river volumetric flow rate only. Maximum concentration in water is, however, also dependent on other river characteristics. For this latter case, generalised modelling approaches are tested for use in situations where detailed hydrological and dispersion data are not available.

Key words: radioactivity, river, sediment, dispersion, advection.

INTRODUCTION

Current dose assessment models are chiefly designed for continuous releases of radionuclides. They therefore commonly 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 potentially have a significant influence on concentrations and bioaccumulation of the radionuclide downstream of the discharge site. A number of models for radionuclide dispersion in rivers have recently been reviewed by Monte et al. (2005).

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 rapid dispersion down river of that 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 a short period of time (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).

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). 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 (e.g. Fischer, 1973). Transverse and longitudinal mixing parameters may be estimated on the basis of empirical relations between river characteristics and dispersion measured using rhodamine dye-tracer experiments (Jobson, 1997; Gharbi and Verette, 1998; Won Seo et al. 1998). These empirical relations can have significant uncertainties (Jobson, 1997), so for site-specific studies, *in situ* dye-tracer experiments are often carried out.

Modelling of 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. 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; Smith et al., 2000) and for nutrients in experimental channels (House et al., 1995). Both particle settling and direct diffusion processes are dependent on the distribution coefficient, K_d . In general, the rate of transfer of high K_d radionuclides to sediments is greater than those with low K_d .

The resuspension of bottom sediments is, like particle settling, determined by water shear stress on the bed sediments (e.g. Partheniades and ASCE, 1965). 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 and spatial resolution (to predict, for instance, daily resuspension rates). Following passage of the radionuclide 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 (Simmonds

et al., 1995). Sedimentation of suspended particulates is expected to decrease as river flow velocity increases.

The present study uses models of pollutant dispersal in rivers together with empirical measurements to assess short term discharges of a number of radionuclides (Table 1) to rivers. Examples are given of application of the model to hypothetical discharges to three 10 km river reaches in the Thames catchment, Southern England. The reaches chosen were on the Thames near Oxford (Sutton Courtney reach), near Pangbourne (Pangbourne-Reading reach) and on the River Colne and Grand Union Canal (GUC) near Rickmansworth, North London. In principle, the dispersion model is similar to previously developed models for radionuclide releases to rivers (Schnoor et al., 1992; Albrecht, 2003) but this study applies such models to a wide range of radionuclides and presents generalised and simplified models for application to new rivers. In addition, a model for the dynamics of radionuclide in fish is incorporated in the radionuclide transport model (see Part II, Smith, 2006).

MODELLING AND METHODS

Advection-dispersion model (ADE)

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	Suspended solids concentration	mg l^{-1}
ρ_s	Bed sediment density	kg m^{-3}
d_s	Sediment mixing depth	cm
f_p	Particulate-sorbed fraction	dimensionless
K_d	Distribution coefficient	l kg^{-1}

The longitudinal transport of a radionuclide with concentration, C (Bq m^{-3}), along a river can be characterized by the one-dimensional advection–dispersion equation (e.g. Fischer, 1973):

$$\frac{\partial C}{\partial t} = D \frac{\partial^2 C}{\partial x^2} - v \frac{\partial C}{\partial x} - \lambda C \quad (1)$$

where v is the mean flow velocity, D is the longitudinal dispersion coefficient, λ is the decay constant of the radionuclide, t is time since discharge of the radionuclide and x is the distance down river. The output of the advection-dispersion model (ADE model) does not account for transverse mixing of the plume: i.e. predictions are for the average concentration across the river cross-section.

The ADE model was implemented in FORTRAN using a backward-difference explicit numerical simulation of the advection-dispersion equation (e.g. Crank, 1975). 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 by sediment deposition is evaluated using the transfer rate, k_1 (d^{-1}):

$$k_1 = \frac{f_p v_p}{d} = \left(\frac{s K_d}{1 + s K_d} \right) \frac{v_p}{d} \quad (2)$$

where f_p ([dimensionless]) is the fraction of the radionuclide sorbed to suspended particulates, v_p (m d^{-1}) is the particulate settling velocity and d (m) is the river mean depth. The particulate sorbed fraction can be calculated from the distribution coefficient, K_d , and suspended solids concentration, s as indicated (ensuring that units are consistent: these parameters are usually quoted in l kg^{-1} and mg l^{-1} respectively).

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:

$$C_s = \frac{A_s}{d_s \rho_s} \quad (3)$$

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.

Simplifying the ADE model

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

Parameters of the simplified ADE 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	$\text{m}^3 \text{s}^{-1}$
k_1	Rate of transfer of radionuclides to bed sediments	s^{-1}
λ	Radionuclide decay constant	s^{-1}
T_i	Duration of release	s

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 ($\text{m}^3 \text{s}^{-1}$):

$$C_w^{\max}(x) = \frac{C_i \times 10^{-3}}{QT_i} (1 - f_p) \quad (4)$$

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 point may be estimated from the following relationships:

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

and

$$C_w^{\max}(x) \approx (1 - f_p)C_T^{\max}(x) \quad (5)$$

where erf is the error function. The error function term represents a solution to the ADE for an extended source of limited extent (Crank, 1975). The exponential term accounts for radioactive decay and transfers to sediments as the plume moves downstream.

Estimating integrated water concentrations

The integrated (infinite time integral, effectively equal to the integral over any time > 3 d since the plume travel time down the 10 km reach is < 3 d) 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 ($\text{m}^3 \text{s}^{-1}$):

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

and

$$\Sigma C_w(x) \approx \Sigma C_T(x)(1 - f_p) \quad (6)$$

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. In Equation (6), 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 reaches considered (see below), this assumption is appropriate as a conservative estimate of radioactivity in the water and for transfers to fish.

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

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

and

$$\Sigma C_w(x) \approx \Sigma C_T(x)(1 - f_p) \quad (7)$$

The latter equations (Equations 7) 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, or within the period of radioactive decay. The former condition may apply in slow flowing rivers which are dredged (i.e. little natural 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, we will conservatively use Equations (6), i.e. assuming no uptake to sediments.

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:

$$C_s^{\max}(x) = \frac{10^3 \times k_1 \times 24 \times 3600 \times d \times \sum C_T(x)}{\rho_s \times d_s} \quad (8)$$

where the factor 10^3 converts the units of $\sum C_T(x)$ from Bq d l⁻¹ to Bq d m⁻³ and the factor 24×3600 converts the units of k_1 from s⁻¹ to d⁻¹. In the above Equation (8), $\sum C_T(x)$ is calculated using Equation (7), i.e. for the case of transfer to sediments, and no subsequent movement of bottom sediments.

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

$$\sum C_s(x) \approx \frac{C_s^{\max}(x)}{\lambda \times 24 \times 3600} [1 - \exp(-\lambda \times 24 \times 3600 \times \tau)] \quad (9)$$

where τ is the integration time in days.

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 ($r^2 > 99\%$, $n = 55$) between the simplified analytical model and the numerical model.

Generalising the model: US Geological Survey model

Where rhodamine dye tracer data is not available for determining model parameters, these must be estimated on the basis of available river characteristics (Jobson, 1997; Won Seo et al., 1998) The model of Jobson (1997) for predicting plume dispersion uses the following input parameters:

- D_a – the catchment area above the river reach, m²;
- 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, m³ s⁻¹;
- Q_a – the mean annual volumetric flow rate at the reach, m³ 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:

$$v' = \frac{vD_a}{Q}, \quad D_a' = \frac{D_a^{1.25} \sqrt{g}}{Q_a}, \quad Q_a' = \frac{Q}{Q_a} \quad (10)$$

the average water velocity may be estimated using:

$$v = 0.094 + 0.0143(D_a')^{0.919} (Q_a')^{-0.469} S^{0.159} \frac{Q}{D_a} \quad (11)$$

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.

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:

$$v_l = \frac{v}{0.89} \quad (12)$$

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

$$C_w^{\text{max}} = \frac{M_i \times 857}{Q \times 10^9} \left(\frac{x}{3600 \times v} \right)^{-0.76} \left(\frac{Q}{Q_a} \right)^{-0.079} \quad (13)$$

where M_i is the amount of radionuclide released (Bq), Q is measured in m³ s⁻¹ and $C_w^{\text{max}}(x)$ 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.

Generalising the model – Won Seo et al. (1998) model

The dispersion coefficient of a plume may be estimated (Won Seo et al., 1998) using:

$$D = 5.915 \times d \times U \times \left(\frac{W}{d} \right)^{0.62} \left(\frac{v}{U_*} \right)^{1.428} \quad (14)$$

where W is the river width, d is the mean depth, v is the mean flow velocity and U_* is the shear velocity which may be estimated using:

$$U_* = \sqrt{g \cdot d \cdot S} \quad (15)$$

where g (m s^{-2}) is the acceleration due to gravity and S (m m^{-1}) is the slope.

Fraction of radionuclide adsorbed to suspended solids and settling velocity

The concentration of suspended sediments in the River Thames is extremely variable, being in the range $1.3 - 50.1 \text{ mg l}^{-1}$ 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 2 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 four categories of radionuclide sorption: $f_p = 0.0, 0.05, 0.5$ and 0.95 . In Table 2, we give recommended best estimates of f_p for use in the model: these are given separately for estimation of activity concentrations in water and fish (where lower f_p gives higher predicted activity concentrations) and for activity concentrations in sediment (where higher f_p gives higher predicted activity concentrations).

Bed sediment parameters

To estimate the settling velocity of suspended sediments, we use a typical value for freshwater sediments of $v_p = 1 \text{ m d}^{-1}$ (Smith et al., 1999, from measurements in lakes). 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, unpubl. results). The sediment mixing depth, d_s , is taken as 0.02 m .

Experimental methods

For the three study reaches, rhodamine WT dye was used to estimate the advection and dispersion of a short term release of radionuclides from the top of each of the reaches: GUC/Colne, Sutton Courtney and Pangbourne-Reading. For the Sutton Courtney reach, previous data obtained by the Environment Agency (EA) for advection and dispersion in this part of the river was used. At the other sites, 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. Measurements were taken in the centre of the river channel at a depth of approximately 20 cm .

RESULTS

Determination of advection and dispersion rates

A solution to the ADE was fitted to the dye tracer data in order to obtain advection and dispersion rates (Figure 1). Parameter values obtained from these model fits are given in Table 3.

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 including the Sutton Courtney reach (Figure 2):

$$D = 0.0148Q^2 + 0.33Q \quad R^2 = 0.93 \quad (16)$$

$$v = 0.028Q^{0.7} \quad R^2 = 0.93 \quad (17)$$

Before the dye-tracer experiments were carried out, expected values of v and D were predicted (see Table 3). The flow velocity was predicted on the basis of the measured volumetric flow rate at a gauging station, and average cross section area (XSA) of the river ($v = Q/XSA$). The dispersion coefficient was predicted on the basis of the empirical relationship given in Won Seo et al. (1998).

Scenarios

The model was run for low, medium and high river volumetric flow rates in each of the three reaches, Sutton Courtney, Pangbourne-Reading and GUC/Colne. The Grand Union Canal (GUC) and River Colne are linked, with water transfer alternating between the canal and river. The river and hydrological characteristics for each of the three reaches under these scenarios are summarised in Tables 4-6.

Model output

Model output is given as a series of graphs of activity concentration and time integrated activity concentration resulting from a hypothetical 1 MBq discharge at the top of each reach for 5 release durations and five locations downstream. The release durations modelled were 5 min, 1 h, 3 h, 12 h and 24 h. The five locations for which predictions were made were 100 m, 300 m, 1000 m, 3000 m and 10000 m downstream of the discharge point. 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.

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 (Equation 4). Equation 4 is only valid for distances down river, x' , where $x' < v \times T_i$. For the three sites studied here, Equation 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, Equation 4 gives the maximum water concentration at any distance up to 24 metres downstream of the discharge. For a 24 hour discharge, Equation 4 gives the maximum water concentration at any distance up to 6912 metres downstream of the discharge. These minimum distances are for the scenario with the lowest water velocity and will be greater for medium or high flow conditions. For distances greater than $x' = v \times T_i$, Equation 4 will over-estimate maximum activity concentrations in water and the relationships presented in Equatons (5) should be used.

Maximum water concentrations further downstream of the discharge point

Further downstream of the discharge point, maximum water concentrations may be estimated using the simplified model above (Equations 5). Since maximum water concentration varies with duration of release, 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. Figures 3 & 4 show model sensitivity to volumetric flow rate, release duration and fraction of the radionuclide sorbed to particulates, f_p , for the Pangbourne-Reading reach.

Figure 3 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 (by assuming no decay) is only approximately 10% at the furthest site (Figure 3). It can be seen from Figure 3 that flow rate and distance from the discharge have a major influence on maximum concentrations at a given site. The duration of the radioactive release also has a major influence on maximum concentrations downstream. Figure 3 (b) shows maximum water activity concentrations for various discharge times under medium flow conditions.

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 4, particle sorption has relatively little influence on total activity concentrations in water (solid lines), but at high values of f_p 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 in both phases under the assumption of no particle sorption, then estimating the dissolved phase activity concentration using $C_w(\text{dissolved}) \approx C_w(\text{total}) \cdot (1 - f_p)$.

Maximum activity concentrations in water over the three reaches studied here are presented in Figure 5 assuming the same volumetric flow rate at all three sites. Generalised estimates of downstream activity concentrations for other rivers are discussed below.

Time integrated water concentrations

Figure 6 gives the estimated integrated water concentration, (total, $\Sigma C_T(x)$ and dissolved phase, $\Sigma C_w(x)$), 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 6 applies to all integration times greater than 3 days. The time integrated water concentration may be used to determine the integrated dose:

$$\text{Estimated dose} = \sum C_w \cdot e_w \cdot I_w \cdot H_{e(ing)} \quad (18)$$

where e_w is the efficiency of removal of the radionuclide during water treatment, I_w is the drinking water ingestion rate and $H_{e(ing)}$ is the effective dose per unit intake by ingestion of the particular radionuclide.

Maximum and time-integrated sediment concentration

Figure 7 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. Time-integrated sediment concentrations for different radionuclides are presented in Figure 8 for two different values of the particulate sorbed fraction, f_p .

DISCUSSION

Assumptions and errors

Rather than explicitly determining model errors, we have taken a bounding value approach to model parameter values by taking the worst-case scenario for each dose pathway. An important potential error in the model concerns the fraction of radioactivity absorbed to suspended sediments and removed to bed sediments. Since values of distribution coefficients vary significantly (IAEA, 1994) conservative assumptions concerning the role of sediments have been made.

1. It is assumed that the radionuclide is discharged in the dissolved phase at a uniform rate during the discharge period. This may not always be true: for example, in some cases discharged radioisotopes may be bound to particulates or organic molecules. This could potentially affect both particle sorption characteristics and uptake through the aquatic food chain.
2. Estimates of water and sediment concentrations are presented as averages across the river cross-section, though it is noted that close to the discharge point, complete transverse mixing of a release at one side of the river may not have occurred.
3. In the graphs of maximum and time integrated water concentration, to simplify presentation of results, 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, Pangbourne reach, low flow conditions) considered in this study, using Eqs. (5) and (6), this leads to an over-estimate of the maximum activity concentration in water of 12%.
4. When calculating activity concentrations in water 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.

5. For simplicity of presentation, the time integrated concentration in sediments (Figure 8) for all radionuclides of half-life ($T_{1/2}$) > 2 years is calculated assuming $T_{1/2} = \infty$. Using Equation (9), for the integration times considered here, this assumption 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%. This applies to all scenarios since, in Equation (9), as $T_{1/2} \rightarrow \infty$ ($\lambda \rightarrow 0$), then $(1 - \exp(-\lambda\tau))/\lambda \rightarrow \tau$. For all other long-lived radionuclides the error is negligible.

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 during which bed sediments can be transported long distances downstream, or out of the river into the estuary. On the other hand, some 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 were made which give conservative estimates of the target variables as follows:

- *Activity concentrations in water.* When these are calculated, it was assumed that there is no transfer of radionuclides to bed sediments. Under this assumption ($f_p = 0$), 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 is 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.

Generalising the model

The model developed and the graphical output presented is in principle applicable to all rivers. 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 of the radionuclide. 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.
2. In an accident situation, an estimate 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, 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 River 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.

In the absence of site specific measurements of river flow velocity and plume dispersion, the 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.

Model validation

Empirical measurements of the fate of short-term direct releases of radionuclides to rivers are limited, so direct “blind” testing of the model is difficult. The rhodamine dye measurements, however, provide a strong empirical basis for the parameters of advection and dispersion of the release at the three sites studied. These data have also provided a limited validation of the generalised models (Jobson, 1997; Won Seo et al., 1998).

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 Figure 9. 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. As shown in Table 3, the velocity, predicted on the basis of the measured volumetric flow rate at a gauging station, and average cross section area (XSA) of the river ($v = Q/XSA$) was reasonably accurate.

The dispersion coefficient, D , was also predicted on the basis of the empirical model of Won Seo et al. (1998). For the GUC/Colne reach, the prediction was relatively accurate (Table 3), but for the Pangbourne-Reading reach, the model (Won Seo et al., 1998) tended to over-estimate dispersion, particularly at short distances from the release. This may have been because the river channel was very straight (“canalised”) in this first part of the reach. In addition, it is known that at short distances from the release, there is a delay in the longitudinal spreading of the pulse, partly as a result of incomplete transverse mixing of the tracer. This leads to an increase in the observed value of D with distance downstream of the release (Liu et al., 1980; Reichert and Wanner, 1991).

As discussed above, the fraction of the radionuclide absorbing to the particulate phase is a key source of model uncertainty, hence the conservative assumptions made about the value of this parameter. A limited validation of these assumptions is given by the study of Albrecht (2003) of the controlled release of radiocobalt by the Beznau reactor to the River Aare in Switzerland. In this study (Albrecht, 2003), it was found “that sorption and/or sedimentation did not play a significant role in Co transport”. Thus, for radiocobalt, our assumption of $f_p = 0$ for estimating activity concentrations in the water phase, and a low value ($f_p = 0.05$) for estimating transfers to sediments appears realistic.

Comparison with Schaeffer model

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:

$$C_w(x) = C_w(0) \exp\left(-\left(\lambda/v + k'\right)x\right) \quad (19)$$

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 (see Equations 2 & 7), however the model parameterisation differs significantly. By comparing Equation (7) to Equation (19) we can relate the sedimentation parameter used in the present study to the Schaeffer coefficient:

$$k' = \frac{k_1}{v} = \frac{f_p v_p}{dv} \quad (20)$$

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 (20) we can compare the removal rate constant, k_1 in the model to recommended values of the Schaeffer coefficient. For the Pangbourne-Reading reach, we compare these values in Table 7. The parameter values for our model are of the same order as those in the Schaeffer model (Simmonds et al., 1995). The particulate settling model used here, however, estimates quite large variation in particle settling as volumetric flow rate changes, whereas the Schaeffer coefficient is independent of river flow velocity.

Key variables

The river volumetric flow rate is clearly the most important environmental variable determining activity concentrations in water and sediments following a radionuclide release. In general, the maximum and integrated activity concentrations in water 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, $C_w^{\max}(x)$, particularly near to the point of discharge, where $C_w^{\max}(x)$ is in inverse proportion to the release duration. Duration of release does not influence the maximum activity concentrations in sediments, nor does it influence the estimates of time integrated activity concentrations in water 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. 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 2). Therefore, the model estimates of activity concentrations in dissolved water and fish always conservatively assume that all 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 2).

CONCLUSIONS

A model was developed to assess hypothetical short term radionuclide releases to three reaches in the River Thames catchment. The model was used to estimate the following target variables:

- Maximum activity concentration in water (total and dissolved phase);
- Time integrated activity concentration in water (total and dissolved phase);
- Maximum activity concentration in bed sediments;
- Time integrated activity concentration in bed sediments.

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. Calculations were made for sites at the following distances downstream of the discharge points: 100 m, 300 m, 1000 m, 3000 m, 10000 m. Where possible, the calculations were presented as graphs of radionuclide concentrations versus volumetric flow rate and distance for a nominal 1MBq radionuclide release. The graphs may be applied to other scenarios by multiplying the output for a 1 MBq release by the actual release amount in MBq.

It was shown that time-integrated water and sediment concentration and maximum sediment concentration may be predicted for all rivers on the basis of the river volumetric flow rate only. Maximum concentration in water is, however, also dependent on other river characteristics. For this case, generalised modelling approaches were tested for use in situations where detailed hydrological and dispersion data are not available.

The models for time-of-travel and maximum pollutant concentration developed by the USGS (Jobson, 1997) are robust, easy to use and have a very strong empirical basis. It is believed therefore that this generalised model is fit-for-purpose, but requires further testing to assess its applicability to other rivers, and to determine the limits of its applicability. 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.

Testing generic methods of estimating river flow velocities (particularly under low flow conditions) and assessing uncertainties in travel time estimates would also be valuable. 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) could be further investigated.

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 2) suggests 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, however, 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.

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

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

Table 2. Estimated 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)
^3H	0	0.00001	0	0.05
^{14}C	1×10^4	0.12	0	0.5
^{32}P	5.7×10^3	0.07	0	0.5
Cs	5×10^3	0.04	0	0.5
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	0.5

* From a review by F. Denison (unpubl. res.). The review emphasised short term values: in the longer term, K_d values may be higher.

Table 3. Advection and dispersion coefficients derived from dye tracer experiments on the three study reaches. Predicted values of these parameters (see text) are given in italics. The Sutton Courtney reach (values in bold font) goes from Culham Lock to Clifton Lock.

Reach	Distance km	Flow, $Q \text{ m}^3 \text{ s}^{-1}$	Mean velocity, $v \text{ m s}^{-1}$	Dispersion, $D \text{ m}^2 \text{ s}^{-1}$
Upper Thames – Sutton				
Courtney reach				
Sandford Lock*	4.4	3.27	0.062	0.67
Sandford Lock*	4.4	19.8	0.17	13.2
Sandford Lock*	4.4	42.5	0.39	40.9
Culham Intake*	3.05	3.27	0.055	0.84
Culham Intake*	3.05	19.8	0.24	7.3
Culham Intake*	3.05	42.5	0.38	46.4
Abingdon Lock*	4.89	3.27	0.086	0.088
Abingdon Lock*	4.89	19.8	0.23	16.5
Abingdon Lock*	4.89	42.5	0.34	28.5
Culham Lock*	5.3	5.45	0.070	1.73
Culham Lock*	5.3	19.8	0.35	14.0
Culham Lock*	5.3	44.1	0.36	48.8
GUC/Colne reach				
1. Coppermill Lane	0.9	8.0	0.25	5.9
2. Denham Green	4.0	8.0	0.38	13.2
3a. Willowbank	9.0	8.0	0.35	13.2
3b. Willowbank	9.0	8.0	0.30	10.7
<i>Predicted before experiment</i>			<i>0.4</i>	<i>14.9</i>
CLNM4*	6.2	3.06	0.15	4.7
Pangbourne-Reading				
1. Boathouse	1.1	25.5	0.24	0.38
2. Mapledurham	2.2	25.5	0.21	0.96
3. Tilehurst	5.3	25.5	0.20	13.2
<i>Predicted before experiment</i>			<i>0.21</i>	<i>18.0</i>
1. Boathouse	1.1	10.5	0.11	0.97
<i>Predicted before expt.</i>			<i>0.09</i>	<i>5.1</i>

*Data supplied by Environment Agency (EA)

Table 4. Summary of river characteristics in the Pangbourne-Reading 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 Environment Agency (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 Environment Agency (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 $d = Q/v.W$

Table 6. Summary of river characteristics in the Grand Union Canal/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, $d = Q/v.W$

Table 7. Comparison of the settling velocity model parameter values for the Thames (Reading-Pangbourne) with the Schaeffer coefficient model parameter values for the river Rhône (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}	

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

Figure 1. Fit of the advection-dispersion (ADE) model (solid lines) to field data from the River Thames: Pangbourne-Reading reach (medium flow). Parameters determined from these data are given in Table 10.

Figure 2. Relationships between (a) advection and (b) dispersion of a conservative dye tracer and river volumetric flow rate for four reaches of the upper River Thames including the Sutton Courtney reach (from data in Table 3).

Figure 3. Maximum modelled water concentrations downstream of a 1 MBq release at the top of the Pangbourne-Reading reach. Results are given for (a) different volumetric flow rates; (b) different durations of the radioactive release. The estimates apply to any radionuclide since in this scenario radioactive decay makes no significant difference to maximum concentrations (the estimate for rapidly-decaying ^{131}I in graph (a) is close to that assuming no decay, even for low flow).

Figure 4. Maximum water concentration as a function of particle sorbed fraction, f_p , for a 1MBq release to the Pangbourne-Reading reach. For $f_p = 0$ -0.05, there is little difference in the total and dissolved-phase activity concentrations or in the change in concentration with distance. At a distance of 10 km, for $f_p = 0.95$, the total activity concentration (i.e. both phases) is reduced by a factor of approx. 2, but the dissolved phase activity concentration is reduced by a factor of approximately 40 compared to the $f_p = 0.05$ scenario.

Figure 5. Maximum concentration in water following releases of different durations and assuming a volumetric flow rate of $10 \text{ m}^3 \text{ s}^{-1}$ at all three study reaches.

Figure 6. 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

Figure 7. 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.

Figure 8. Time integrated (one year) sediment concentration (a) $f_p = 0.05$; (b) $f_p = 0.95$. These represent a very conservative estimate as it is assumed that there is no remobilisation of sediment during the one year period.

River Colne - USGS model "blind" test

Figure 9. Test of USGS model (Jobson, 1997) against dye tracer data from the GUC/Colne reach. The solid line shows the USGS model prediction of both flow velocity (travel time) and dispersion. The dashed line shows the USGS model prediction of dispersion given an independent estimate of flow velocity (travel time) from volumetric flow rate and river cross section ($v = Q/XSA$). The legend gives site distances from the discharge point.