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Model for radionuclide transport in running waters

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Abstract

Two sites in Sweden are currently under investigation by SKB for their suitability as places for deep repository of radioactive waste, the Forsmark and Simpevarp/Laxemar area. As a part of the safety assessment, SKB has formulated a biosphere model with different sub-models for different parts of the ecosystem in order to be able to predict the dose to humans following a possible radionuclide discharge from a future deep repository.

In this report, a new model concept describing radionuclide transport in streams is presented. The main difference from the previous model for running water used by SKB, where only dilution of the inflow of radionuclides was considered, is that the new model includes parameterizations also of the exchange processes present along the stream. This is done in order to be able to investigate the effect of the retention on the transport and to be able to estimate the resulting concentrations in the different parts of the system. The concentrations determined with this new model could later be used for order of magnitude predictions of the dose to humans.

The presented model concept is divided in two parts, one hydraulic and one radionuclide transport model. The hydraulic model is used to determine the flow conditions in the stream channel and is based on the assumption of uniform flow and quasi-stationary conditions. The results from the hydraulic model are used in the radionuclide transport model where the concentration is determined in the different parts of the stream ecosystem. The exchange processes considered are exchange with the sediments due to diffusion, advective transport and sedimentation/resuspension and uptake of radionuclides in biota. Transport of both dissolved radionuclides and sorbed onto particulates is considered. Sorption kinetics in the stream water phase is implemented as the time scale of the residence time in the stream water probably is short in comparison to the time scale of the kinetic sorption. In the sediment, however, the approximation of equilibrium chemistry is assumed to be sufficient for order of magnitude predictions when a constant inflow of radionuclides is considered.

A first sensitivity analysis of the model is performed in which different model parameters have been varied. At the time for the model development, almost no detailed site-specific information about e.g. channel geometry or sediment characteristics were available. Simulations were therefore performed for a hypothetical case, where the ranges of possible parameter values was based on literature information, generalizations from other stream systems and some site-specific information such as large-scale information of the morphology at the present sites.

For order of magnitude predictions of the concentration or amount of radionuclides in the different parts of the stream ecosystem, a yearly mean value of the water flow was assumed to be sufficient. Therefore, the further sensitivity analyses were performed for constant flow conditions.

The sensitivity analyses indicated that the main retention along the stream is due to uptake within the sediment. Initially, the uptake will cause a retardation of the solute transport. The sediment capacity is however limited and after saturation, the outflow of radionuclides in the longitudinal direction will be completely determined by the inflow to the system. The time for reaching this equilibrium and the equilibrium concentration in the sediment varies however with different conditions and radionuclides, e.g. due to sorption characteristics, sedimentation velocity and advective velocity within the sediment. The degree of variation caused by different factors is, however, different.

In the simulations performed in this study, the time for reaching equilibrium ranges from less than a year to a couple of hundred years.

For predictions of the dose to humans, the accumulated amount in the sediment should also be considered and not only the concentration in the stream water. Therefore, transport calculations by a model with the appropriate exchange processes included are important.

To be able to make certain statements about the transport of radionuclides in the current streams, further investigations are recommended in the report where field measurements of site-specific parameters is an important component. An updated more complete sensitivity analysis of the model is also suggested to be performed after the inclusion of site-specific information.

Contents

1 Introduction

1.1 Background

At present SKB is investigating possible sites and construction for repository of spent nuclear waste. As a part of the safety assessment, SKB has formulated biosphere models for different ecosystems in order to calculate the dose to humans and biota from a possible radionuclide discharge to the biosphere.

The biosphere model has been divided in several objects, one for each different type of the ecosystems: agricultural land, peat bog, coast, lake, groundwater and running water /see e.g. Bergström et al. 1999, Karlsson et al. 2001/. The final goal is to couple the different objects in order to be able to describe the transport of radionuclide through the different ecosystems and to evaluate the risk for human and biota, following a possible discharge of radionuclides from a deep repository.

The biosphere model is to be used for predicting the effect of a possible radionuclide discharge from the two sites under investigation: the Forsmark area in Östhammar and the Simpevarp/Laxemar area in Oskarshamn as a step in the ongoing investigations of their suitability as sites for repository of spent nuclear waste.

This report deals with radionuclide transport in running waters where a new more detailed transport model than the one previously used by SKB is presented.

1.2 Objectives

The previous stream transport model used by SKB /Bergström et al. 1999/ is simplified, considering only dilution of the inflow of radionuclides with no regard of any other interactions. In order to be able to evaluate the dose to human following a possible discharge of radionuclides to the biosphere, a stream transport model with the main processes incorporated is needed.

The aim with the new model is to extend the previous framework to include parameterizations of the retention processes present along the stream, in order to be able to investigate the effect of retention on the radionuclide transport. By such an approach it is possible to derive concentrations of radionuclides in different parts of the stream ecosystem, which eventually can be used to calculate the dose to humans and thereby the effect of a possible radionuclide discharge from a deep repository. In the present study, no dose calculations are performed, rather a model concept for estimating the radionuclide concentration in the different parts of the stream ecosystem is proposed.

In the report the effect of varying the primary parameters is studied. Moreover, sensitivity analyses of the proposed model are performed in order to get information about possible ranges of concentrations in the different parts of the stream ecosystem i.e. stream water, sediment and biota.

2 Transport and retention in streams

2.1 Processes in stream water and sediment

Several hydraulic, chemical and biological processes in both the stream water and in the sediment will affect solutes transporting along a stream or river. A number of these processes will cause a retardation of the longitudinal transport, of different order of magnitude.

The longitudinal transport of solutes along a river is driven by the mean flow velocity in the stream. Further, dispersion i.e. the combined effect of longitudinal differential advection and transversal mixing, will cause a spreading of an introduced solute pulse.

Depending on flow conditions, a more or less pronounced transport of adsorbed solutes due to bed-load movements will occur, in addition to the longitudinal transport of dissolved and suspended adsorbed solutes driven by the mean flow velocity.

Retention of the solute transport will be caused by temporal storage of solutes in stagnant re-circulating zones in the stream water or in the sediment surrounding the stream channel.

Different processes control the exchange of solutes between the stream water and the sediment. For dissolved solutes a diffusive exchange will occur due to the concentration differences in the stream water and in the pore water in the sediment. Another exchange mechanism for dissolved solutes is advection into and out of the stream sediment. Laboratory studies report pressure driven advection as the main exchange mechanism for solutes over bed-forms /Thibodeaux and Boyle 1987, Huettel et al. 1996/. The advective exchange occurs due to a higher pressure on the flow-obstruction opposing the flow yielding a transport into the bed and a lower pressure on the downstream side yielding a transport out of the sediment, a process referred to as “pumpning”. Presented theories describing this exchange often assumes a flat bed with a sinousidal distribution of pressure head on the bed surface /Elliott and Brooks 1997ab, Packman et al. 2000ab/.

For reactive solutes, the degree of particle association also has to be taken into consideration, as it can cause short- or long-term immobilization of the solute. Different chemical processes i.e. adsorption, dissolution and precipitation are examples of such processes. For reactive solutes there is therefore also an exchange of adsorbed solutes due to sedimentation and resuspension of carrier particles, in addition to the exchange of dissolved fractions between the stream water and sediment.

Previous studies have shown that the effect of solute uptake in the sediment (or the hyporheic zone) can greatly affect solute transport in streams at certain time-scales /e.g. Jonsson 2003/. During a field experiment with a short injection of ^{51}Cr in a small stream, only ~24% of the injected mass had passed a station 30 km downstream the injection point within a week. The main part of the initial loss from the stream water was due to retention in the sediment. In this experiment the wash-out from the sediment was observed during several months /e.g. Jonsson 2003/. The time-scale of the retention in a stream system is naturally dependent of substance in consideration and the effect of the retention may therefore be more or less pronounced.

Expect for the exchange of solutes between the stream water and the sediment at “normal” flow conditions, temporal flooding of rivers can transfer transporting solutes to floodplains where storage in overbank sediment can cause additional retention in the stream system /e.g. Macklin and Klimek 1992, Walling et al. 1999/. Also in a long-time perspective, changes in geomorphology (e.g. meandering rivers) can influence the fate of the solutes, where deposition and erosion can cause long- or short-term immobilization of the transporting substance.

Retardation of the solute transport can also be caused by uptake in biota, where for some substances also transformation processes can affect the fate of the solute. Depending on the amount of biomass as well as the ability of the solute to accumulate in the plants, the retention in biota can be more or less pronounced.

The above mentioned processes will act on different spatial and temporal scale and all processes will not necessarily be present in every watercourse or effective for all substances.

2.2 Existing stream transport models used in the context of radioactive repository

The transport model for running waters, given a scenario of a discharge of radionuclides from a deep repository of radioactive waste, that until now have been used by SKB in the context of dose calculations is based on a simplification of the transport processes /Bergström et al. 1999/. The box model encompasses a single compartment, representing the amount of radionuclide in the stream water. Only dilution of the inflow of radionuclides is considered where as other exchange processes, e.g. the exchange with the sediment, are neglected. No thorough analysis is made of how leaving out the exchange processes affects the results.

Also in other countries, the biosphere models used in the safety assessment in the context of spent nuclear waste seldom includes a more detailed model for the stream transport model. If streams and rivers are considered, often only dilution is considered or a simplified exchange with the sediment is implemented.

In the stream model formulated by Nirex (United Kingdom), the interaction between the stream and the stream sediment is not included and the concentration in the stream water is calculated by means of the discharge in the stream and the inflow of radionuclides to the system /Stansby and Thorne 2000/. In Finland (POSIVA), the approach follows the model by /Karlsson et al. 2001/. The streams are however of low priority in their case due to the location of the site for the planned repository (island on Baltic coast). In Switzerland (NAGRA) the dose calculations following a possible radionuclide discharge are performed on the assumption that the aquifer is used for drinking water. The concentration in stream is then not considered for the dose calculations as the concentration in the aquifer will be higher as a further dilution is assumed when the radionuclides reach the surface water. It is assumed that the bed sediment interactions are dominated by solid material transport as the concentration of suspended matter in the rivers there is high. The river water-bed sediment pore water interaction is neglected due to the assumption that the material in the bed sediment is exchanged with the stream water once per year. This is an assumption that is considered to be reasonable for the fast-flowing alpine rivers present there, however, not valid for larger rivers. Also the exfiltration of aquifer water to the sediment is neglected, rather the transfer is directly from the aquifer to the stream water /Klos et al. 1996/.

In Sweden the landscape is different, and the assumption of a yearly exchange of the sediment as in Switzerland is probably not valid as the sediment transport is lower and therefore also the solute exchange with the sediment might be of importance.

In order to be able to evaluate the effect of retention in e.g. the sediment, a more detailed model than the one by /Bergström et al. 1999/, previously used by SKB, is needed. In this way there is a possibility to get an estimation of the accumulated amount of radionuclide and thereby also the radionuclide concentration in the different parts of the ecosystem for the dose-calculations. Only after such an evaluation, a conclusion can be made whether the retention processes can be neglected for the specific purpose or not.

Even if a more detailed stream transport model is not often used in the context of radioactive repository, a numerous number of stream transport models have been formulated within other research areas from which knowledge could be gained. Some of theses model types will be described in the next section.

2.3 Other stream transport models

In context other than strictly in connection to radionuclide deep repository, a numerous number of models have been proposed in the literature describing solute and sediment transport in streams and rivers.

There exist mainly three types of models for radionuclide transport in surface waters: numerical models, compartment models and analytical models /IAEA 2001/. The use of numerical models involves a formulation of the basic transport equations by using e.g. finite differences. To be able to apply compartments models, the assumption of an instantaneous mixing in the compartments has to be applied and to be able to solve the transport equations analytically, simplifying assumptions regarding e.g. the geometry and flow conditions has to be done.

In some of the models used for predictions of solute or radionuclide transport, the interaction with the sediment is included, while in others, this exchange is neglected with the argument that by excluding this exchange, the risk for underestimating the concentration in the stream water and thereby the dose to humans is minimized /e.g. IAEA 2001/. However, by neglecting the sediment interactions, the consideration of e.g. human exposure to accumulated amounts in the sediment is disregarded, which is not always to prefer.

A review of river transport models from different research fields has e.g. been performed by AECL (Atomic Energy of Canada Limited) proceeding their work of formulating a river transport model to be used in the in context of nuclear fuel waste disposal /Bird 1996/. Here the models for sediment of solute transport are classified either as empirical models, nutrient spiraling models, numerical models or random walk models. The resulting conclusion from this report is that a combination and reformulation of existing models is appropriate for finding a model concept applicable for radionuclide transport in streams with the final goal of using them for approximate dose calculations over long time-periods.

Another literature review of available aquatic models for lakes, rivers, coasts and seas was made within the EC project FASSET /Brown et al. 2003b/. The river models mentioned here are MIKE11, PRAIRIE, RIVTOX and CASTEAUR (see /Brown et al. 2003b/ and references therein for further information).

Many of the existing solute transport models are based on mass conservation in terms of the advection-dispersion equation, used early in stream water studies by e.g. /Fischer et al. 1979/. With the objective of finding model concepts describing the transport and retention of pollutants in streams, information from field experiments where a tracer is injected into a stream have often been used during the model development. During such experiments, it was observed that the advection-dispersion equation had to include also a parameterization of the temporal immobilization in storage zones to be able to represent the empirical observations /Thackston and Schnelle 1970, Bencala and Walters 1983/. Different model formulations, especially variations in the description of the exchange with the sediment have since then been formulated. The parameterization of the exchange between the stream water and sediment is either treated in a zero-dimensional box-type model /e.g. Bencala and Walters 1983/ or with the intention of trying to describe the exchange mechanism in a more physically correct way /e.g. Jackman et al. 1984, Wörman 2000/ where the concentration in the sediment is spatially distributed in the vertical direction.

A common approach is to describe the uptake in the sediment with an exchange diffusive coefficient as advective transport into the sediment has been found able to describe in a diffusive manner /Elliott 1990/. In this way, both the diffusive and the advective exchange can be parameterized with one expression. However, a lumped coefficient complicates the generalization to other stream and stream conditions. Another approach of describing the exchange with the sediment is to formulate the exchange as an advective processs due to pressure variations on the bed surface, which probably is the dominating exchange mechanism between the stream water and the sediment in most running waters. Model concepts with advective storage path in the sediment are often paramterized using residence time distributions within the sediment /e.g. Wörman et al. 2002a/.

When dealing with reactive solutes, also the adsorption onto suspended particulate matter and sediment particles are implemented in the models, either assuming equilibrium chemistry or kinetic sorption /e.g. Jonsson 2003/. A number of semi-empirical relationships have also been proposed for sediment bed load transport /e.g. Chanson 1999, van Rijn 1984/ which can be an important transport mechanism for contaminant transport in certain cases.

In the review by /Bird 1996/, mainly four of the existing river models reviewed were recommended to be considered as a basis for the AECL's development of a river model in context of nuclear fuel waste assessments. The presented models (SERATRA, HEC-6, OTIS and FCM) (see /Bird 1996/ and references therein for further information) are either models that do not simulate the fate of contaminants, but rather formulated for predictions of solely sediment transport or contains a set of sub-models for both sediment transport and contaminant transport.

The OTIS model (One Dimensional Transport with Inflow and Storage) formulated by the USGS is an example of a one-dimensional model based on the advection-dispersion equation with additional terms for the transient storage, lateral inflow, first-order decay and sorption /Runkel 1998/. This model is often used in conjunction with data from field experiment to be able to quantify the hydrological parameters included in the solute transport model.

A number of radioecological models have also been developed that includes a more detailed description of the river or stream system and the exchange processes along the river. This is e.g. the case with the model RIVTOX developed within the EC project RODOS (Realtime Online DecisiOn Support system), at IMMSP, Cybernetics Centre, Kiev /Zheleznyak et al. 2003/ that was formulated in order to develop a model-based decision support system after the Chernobyl accident. The model describes the transport of radionuclides in a network of river channels and includes more detailed descriptions of both the hydraulics and the

retention mechanisms for radionuclides in streams than most models used in the context of dose calculations from possible deep repository leakage of radionuclides. The model is a one-dimensional transport model where the pollutant in the river is transported with the water flow. The exchange mechanisms sedimentation and resuspension of suspended contaminated sediment and the diffusive exchange of radionuclides into the interstitial water in the sediment are treated in the model formulation by using equilibrium distribution coefficients.

Mike 11 is an example of an engineering software package by DHI Water and Environment for simulation of flow and transport of dissolved and particulate material in rivers, estuaries, channels and irrigation systems /DHI 2004/. The model for transport in rivers is also here based on the advection dispersion equation. Examples of other available sub-models are the water quality model for heavy metal predictions in stream water and sediment and the model describing deposition and erosion of cohesive sediment.

For prediction purposes, a model based on the underlying physical processes is prefered. For safety assessments of a deep repository, which involves long-term predictions into the future of the magnitude of resulting concentrations in different parts of the ecosystem, a balance between the complexity of the model and the possibility to run the model with meaningful parameter values for a long time is however necessary. Without the ability to relate the parameters to variables possible to get from site-specific information, a complex model is rather useless. Also, as mentioned before, the different processes act on different spatial and temporal scales, and can therefore be more or less important depending on model application.

The information from existing models and experimental investigations reported in the literature are used in the present study as a basis for the development of a compartment model suitable for order of magnitude predictions at the sites currently under investigation by SKB for their suitability as places for nuclear waste repository. The proposed model concept is presented in the following chapter.

3 Model approach for radionuclide transport in running waters

A new stream transport model for the prediction of the effect of a possible radionuclide discharge from a deep repository has been formulated. This model should serve as a base for the evaluation of the effect of the retention processes on radionuclide transport in streams. The model can be used to estimate the amount of radionuclide in different parts of the stream system. The model is formulated in terms of a compartment model where the formulations of the mass-transfer relationships are based on the underlying physics of the exchange mechanisms. In the following sections, the processes that are considered in the model and their mathematical implementation together with assumed simplifications are presented.

3.1 Conceptual model

A conceptual figure describing the transport mechanisms and retention processes for the radionuclides in the new transport model is found in Figure 3-1.

The present model includes a description of the transport for both adsorbed and dissolved phases of the radionuclide. The uptake between the sediment and the stream water is described as advective and diffusive for the dissolved phase of the radionuclide while the exchange of the adsorbed phases is due to sedimentation and resuspension. The transport of radionuclides along the stream is due to advection with the running water and due to bed-load transport. However, depending on the radionuclide and site specific conditions, the bed-load transport might be of minor importance.

The sediment is further represented as a surface and a deeper layer and the net sedimentation over a year-cycle is assumed to be zero. This implies that the sedimentation and resuspension of particulate matter i.e. carrier particles for the radionuclides, is of equal size.

Equilibrium chemistry is assumed to prevail in the sediment whereas sorption kinetics is implemented in the stream water phase. The reason for including the kinetics in the stream water phase is that the water residence time in the streams in consideration is found to be rather short in comparison to the time-scales of the kinetic sorption constant. Including a kinetic sorption description in the stream water in the model allows for a test of the influence of the kinetic sorption in the case when necessary data is available.

In reality effects of sorption kinetics also in the sediment might influence the transport /see e.g. Jonsson 2003/. However, an equilibrium description can be used as an approximation, to get an idea of how the radionuclide will be transported in the stream when a constant inflow of radionuclides to the system is considered. The kinetics in the sorption process is probably not of vital importance, for the present purpose, in comparison to the kinetics introduced by the exchange processes between the streaming water and the sediment. Also, at present, no data is available for a calibration of a detailed kinetic sorption model, and inclusion of such a description is therefore not justified. However it is likely that an equilibrium chemistry description of the retention process in the bed sediment is enough for getting an approximative description of the equilibrium state of the radionuclide transport from a future possible repository discharge of radionuclides. However, if the

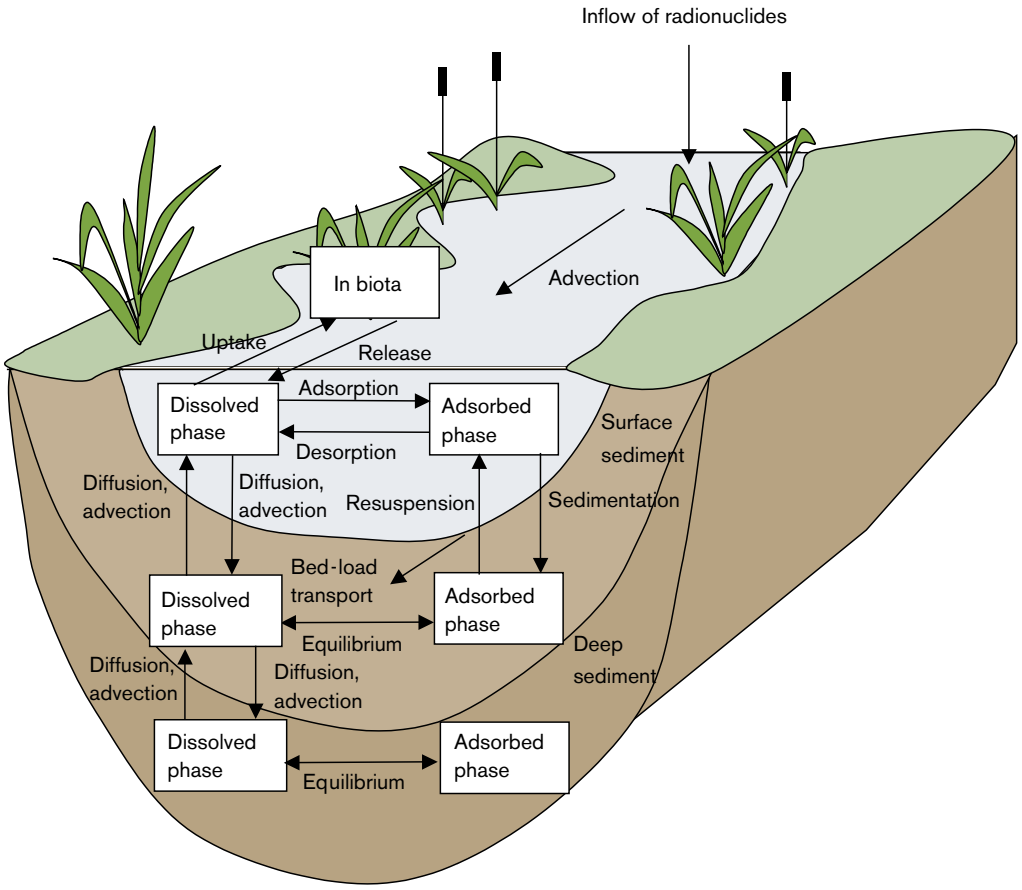

Figure 3-1. *Conceptual figure describing the phases and processes considered in the transport model for radionuclide transport in streams. Boxes represent mass conservation relationships and arrows mass transfer relationships.*

inflow of radionuclides to the system ceases and the wash-out of radionuclides from the sediment is to be studied, a kinetic sorption description also in the sediment might be needed.

In the present model, the transfer of dissolved radionuclides to biota such as macrophytes and attached communities or biofilms such as epiphyton or epilithon is also considered with mass transfer relationships by means of equilibrium conditions corresponding to bioconcentration factors (BCF). The biomass is assumed to be constant along the river and no change in biomass over time is assumed. The amount of biomass is estimated from reported observations of standing crops in rivers.

The model is divided in two parts, a hydraulic model where the flow conditions are calculated and a radionuclide model, where the concentration of radionuclides in the different parts of the ecosystem is calculated using the flow conditions determined by means of the hydraulic model.

In this study, the geometry of the channel is assumed to be valid also for future conditions, i.e. no geomorphological changes are considered in the present model. Also, only the transport in the actual stream channel is considered, but flooding is assumed to occur if the water level exceeds the height of the banks. However, the consequences of flooding and storage in overbank sediment are outside the scope of this work and must be considered using another model. Further, dispersion is neglected in the model formulation as it is assumed to be of minor importance when considering long-term constant releases as is the

case with the leakage from nuclear repository. In the sensitivity analyses performed and presented in this report, an inflow of radionuclides is assumed to the stream water, but an inflow of radionuclides to the system could instead be applied to the sediment if that is to prefer.

3.2 Box model representation

The conceptual model presented in the previous section is formulated as a compartment or box model. The assumption necessary for applying a box-model to running waters is that complete mixing is assumed in the compartments. This is of course not the case in reality and the assumption can lead to that the concentration in some parts of the ecosystem is underestimated. However, the model is to be used for estimating the equilibrium concentrations on long time-scales and therefore, the assumption of using a box model representation is assumed to be justified for the purpose. However, the box model can be divided into several compartment systems coupled along the stream if a finer computational grid is wanted. For example in /Elert and Lindgren 1993/ the effect of an increasing number of compartments on the calculated concentration in a radionuclide transport model for soil is investigated. A detailed evaluation of the effect of the number of boxes has not been performed within this study. However, in this case, the discretisation of the stream is expected to have minor effects on the results. The residence time in the stream water is short compared to the exchange rate with the sediment and low gradients in water concentration are expected along the stream.

A sketch of the formulated box model is found in Figure 3-2. The five compartments represent dissolved, adsorbed or total amount of radionuclide in the phases, stream water, surface sediment, deep sediment and biota.

The arrows between the boxes represent mass transfer relationships between the different phases. The exchange processes represented with the arrows in the box model sketch is as follow:

- k_{ads} adsorption onto suspended particulate matter,
- k_{des} desorption from suspended particulate matter,
- $k_{12\text{-}diss}$ advection, diffusion,
- $k_{21\text{-}diss}$ advection, diffusion,
- $k_{12\text{-}ads}$ sedimentation,
- $k_{21\text{-}ads}$ resuspension,
- k_{14} uptake in biota from stream water,
- k_{41} release from biota to stream water,
- k_{23} advection, diffusion,
- k_{32} advection, diffusion,
- k_{sed} outflow due to sediment transport,
- k_{out} outflow due to advection in the stream water,
- k_{flood} outflow due to flooding of the stream channel.

As the concentration of suspended material in the stream water is assumed constant with time, i.e. the sedimentation and resuspension of particulate matter is of equal size, the exchange from the surface to deep sediment is not due to sedimentation and resuspension but rather solely due to advection and diffusion.

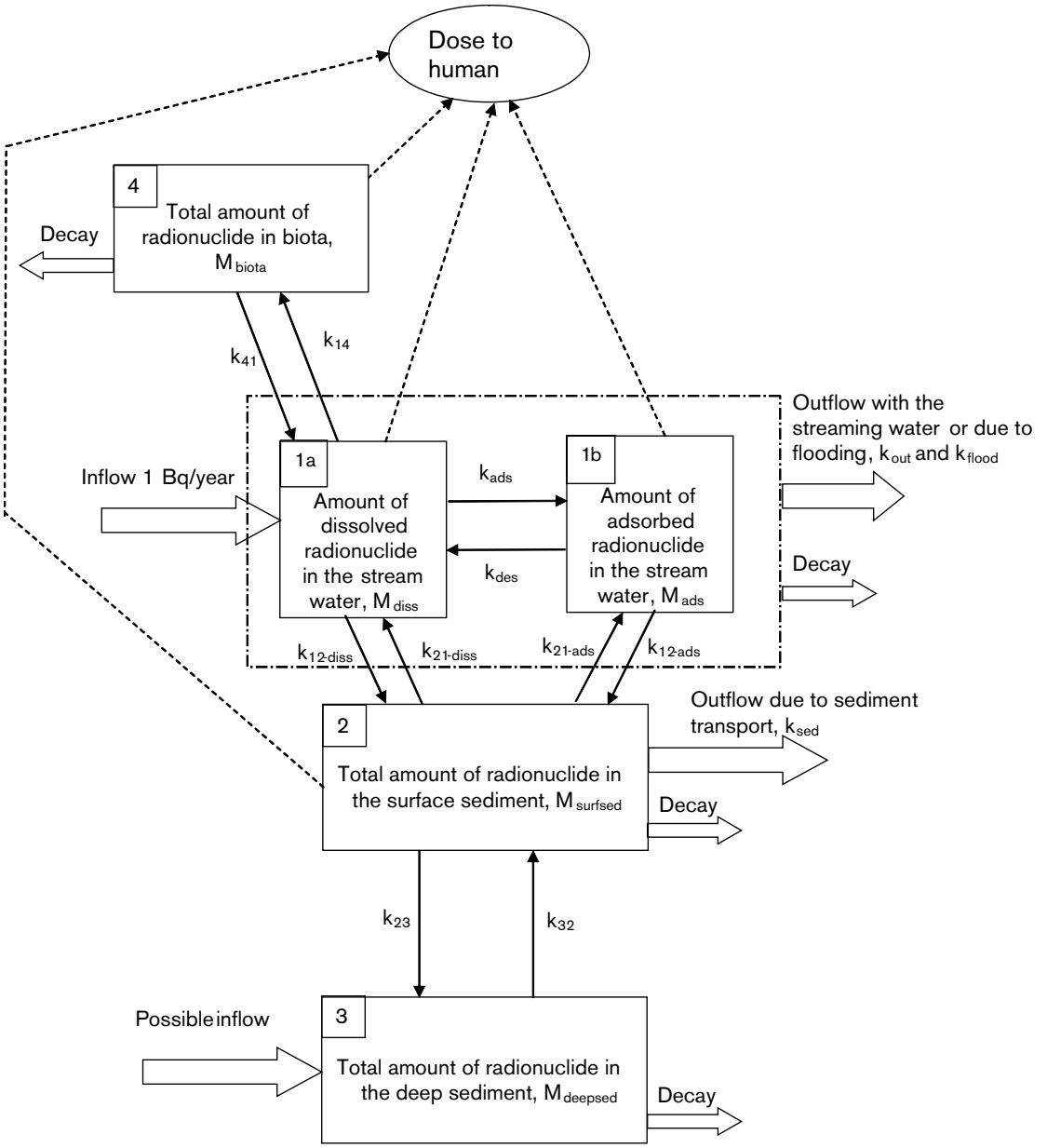

Figure 3-2. *Box model representation of the conceptual model presented in Figure 3-1. Boxes represent mass conservation relationships and arrows between boxes mass transfer relationships.*

The dotted lines in Figure 3-2 showing the relation to the dose to human from the boxes is just included to point the connection with the results from this transport model to the future dose calculations. However, in this report no dose-calculations are made.

Also decay of the radionuclides has to be considered. However, for the sensitivity analyses performed within this study, long-lived radionuclides are considered and a constant inflow or radionuclides to the river system is assumed. In this case no decay of the radioisotopes is implemented.

The inflow of radionuclides to the model could be applied either to the stream water or the sediment. In the model simulations presented in this report, a constant rate of 1 Bq/year is applied to the stream water.

3.3 Mathematical formulation of transport processes

The transport model used for estimating radionuclide concentrations in the different parts of the stream ecosystem is divided into two model parts, one hydraulic model where the flow conditions are calculated and one where the radionuclide transport is calculated using the results from the hydraulic model. In the next two sections, the mathematical formulations of these two models are presented.

3.3.1 Hydraulic calculations

A prerequisite for the hydraulic calculations is that uniform flow prevails which is the case when the geometry along the channel does not change. Further the flow is assumed to be quasi-steady i.e. the water depth is calculated for each time-step without taking inertia effects into consideration. Also that the energy grade line is equal to the water surface slope, which is an assumption that is justified when the velocity head is small.

The flow in the stream, Q (m^3/s) is calculated by means of values on specific runoff, q_s ($l/s,km^2$), and the area of the watershed, A_{ws} (km^2) as a function of time.

The shape of the stream cross section is assumed to be triangular defined by the angle, α (Figure 3-3).

Applying the assumption mentioned above, the water depth in the stream channel can be determined by calculating the normal depth, y_n, by means of Manning's equation as a function of time. The Manning equation relates the water flow to the geometry of the channel and the roughness coefficient as:

$$Q = V_{adv} A_{cross} = \frac{R_h^{\frac{2}{3}} S_b^{\frac{1}{2}}}{n} A_{cross} \qquad (1)$$

where Q is the water flow (m^3/s), V_{adv} the mean flow water velocity (m/s), A_{cross} the cross-sectional area (m^2), R_h the hydraulic radius (m) ($R_h=A_{cross}/P$ where P is the wetted perimeter (m)), S_b the slope of the channel [–] and n is the Manning roughness coefficient ($m^{-1/3}s$).

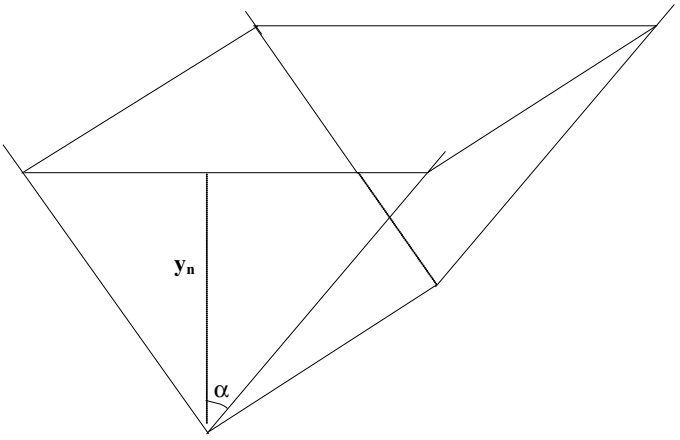

Figure 3-3. *Assumed shape of the stream channel.*

When a triangular cross section is assumed the normal depth, y_n (m), can be calculated according to:

$$y_n = \left(\frac{Q\, n\, 4^{1/3}}{(\sin\alpha)^{2/3} S_b^{1/2} \tan\alpha} \right)^{3/8} \quad (2)$$

By means of the normal depth the wetted perimeter, cross-sectional area and hydraulic radius can be calculated for each time-step as:

$$P = \frac{2 y_n}{\cos\alpha} \quad (3)$$

$$A_{cross} = y_n^2 \tan\alpha \quad (4)$$

$$R_h = \frac{A_{cross}}{P} \quad (5)$$

The advective flow in the stream, V_{adv} ($=Q/A_{cross}=q_s A_{ws}/(y_n^2 \tan\alpha)$), is then determined.

The technique used for the hydraulic calculations is as mentioned applicable during uniform flow conditions. No consideration is made to calculate dammed conditions if change in slope or geometry downstream causes e.g. dammed conditions upstream. In such a case, a more sophisticated iterative method for calculating the flow upstream must be applied. However, the applied method should be sufficient for the purpose of this model where no detailed information about geometry along the river is available.

The hydraulic calculations for extreme flow conditions can be treated as an outflow from the stream transport model when a certain water depth, y_{max}, is reached in the channel (Figure 3-4). The transport in the defined cross section in this case is calculated according to the above mentioned method, while the flow of radionuclides to surrounding land is regarded as a loss term from this part of the ecosystem. The loss could e.g. be transferred to the mire or agricultural model as an inflow. No re-transport of the flooded radionuclides to the stream system will be considered in this model.

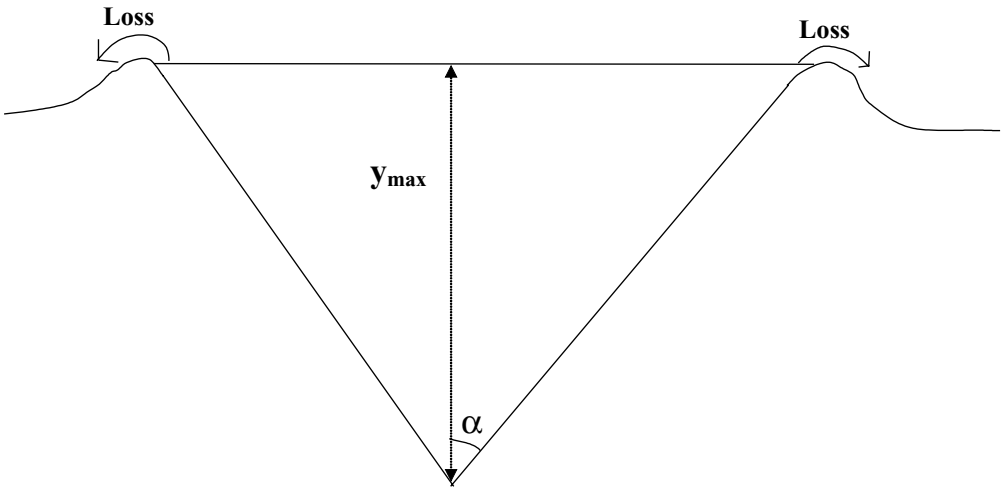

Figure 3-4. *Flooding of the main cross-section yields a loss from the stream water to surrounding land. No re-transport of the radionuclides in the flooded water to the stream channel is considered.*

The maximum water flow (Q_{max}) corresponding to the maximum water depth in the main channel (y_{max}) is determined by means of Equation (1). Also the total volume of water that is flowing out of the system due to flooding is determined. This volume is then distributed evenly over the year ($q_{flooded\ water}$) and later used to calculate an extra outflow of radionuclides from the stream channel, in addition to the outflow due to longitudinal advection in the main channel.

3.3.2 Radionuclide transport model

3.3.2.1 Mass balance equations

The presented box model is formulated as one mass balance equation for each box described in Figure 3-2 i.e. the amount of dissolved radionuclide in the stream water (M_{diss}), the amount of adsorbed radionuclide in the stream water (M_{ads}), the total amount of radionuclide in the surface sediment ($M_{surfsed}$), the total amount of radionuclide in the deeper sediment ($M_{deepsed}$) and finally, the total amount of radionuclide in biota (M_{biota}) as:

Mass conservation for radionuclides in dissolved phase in stream water

$$\frac{dM_{diss}}{dt} = Inflow - k_{ads}M_{diss} + k_{des}M_{ads} - k_{12-diss}M_{diss} + k_{21-diss}M_{surfsed} - k_{14}M_{diss} + k_{41}M_{biota} - k_{out}M_{diss} - k_{flood}M_{diss} - \Theta \ln(2)/T_{1/2}M_{diss} \quad (6)$$

Mass conservation for radionuclides in adsorbed phase in stream water

$$\frac{dM_{ads}}{dt} = k_{ads}M_{diss} - k_{des}M_{ads} - k_{12-ads}M_{ads} + k_{21-ads}M_{surfsed} - k_{out}M_{ads} - k_{flood}M_{ads} - \Theta \ln(2)/T_{1/2}M_{ads} \quad (7)$$

Mass conservation for total amount of radionuclides in surface sediment

$$\frac{dM_{surfsed}}{dt} = k_{12-diss}M_{diss} - k_{21-diss}M_{surfsed} - k_{23}M_{surfsed} + k_{32}M_{deepsed} - k_{sed}M_{surfsed} + k_{12-ads}M_{ads} - k_{21-ads}M_{surfsed} - \Theta \ln(2)/T_{1/2}M_{surfsed} \quad (8)$$

Mass conservation for total amount of radionuclides in deeper sediment

$$\frac{dM_{deepsed}}{dt} = k_{23}M_{surfsed} - k_{32}M_{deepsed} - \Theta \ln(2)/T_{1/2}M_{deepsed} \quad (9)$$

Mass conservation for total amount of radionuclides in biota

$$\frac{dM_{biota}}{dt} = k_{14}M_{diss} - k_{41}M_{biota} - \Theta \ln(2)/T_{1/2}M_{biota} \quad (10)$$

where Inflow is the flux of radionuclides into the model domain (Bq/s), $T_{1/2}$ the radioactive half-time (s) and Θ a parameter regulating if radioactive decay should be implemented or not (for long-lived radionuclides, no radioactive decay is considered and Θ=0; for short-lived radionuclides, radioactive decay is considered and Θ=1).

The mathematical relationships for the transfer functions will be presented in the following section.

3.3.2.2 Mass transfer functions

In this section the rate coefficients between the different compartments (i.e. mass phases) are given, as well as a short explanation of their derivation. The rate coefficients in a box-model could be described by the quotient of the flux between boxes and the amount of mass in the box from which mass is leaving.

Mass transfer between dissolved and adsorbed phase in stream water, k_{ads} and k_{des}

A rate-limited expression is applied to describe the sorption and desorption of radionuclides onto/from the particulate matter in the stream water. The transfer coefficients are defined as:

$$k_{ads} = \frac{\ln(2)}{T_k} \tag{11}$$

$$k_{des} = \frac{\ln(2)}{T_k c_p K_d} \tag{12}$$

where T_k is the half-time to reach sorption equilibrium (s), c_p the concentration of suspended particulate matter in the stream water (kg/m^3) and K_d the distribution coefficient in the stream water (m^3/kg) defined as:

$$K_d = \frac{c_a}{c_d c_p} \tag{13}$$

where c_a is the concentration of radionuclides onto particulate matter in the stream water (Bq/m^3) and c_d is the concentration of dissolved radionuclides in the stream water (Bq/m^3). This formulation is different from the formulation previously implemented in the lake model by /Karlsson et al. 2001/. In their formulation, the expression for the adsorption rate coefficient included the distribution coefficient and the concentration of suspended particulate matter in the stream water. However, the formulation used in the present study describes the actual sorption behaviour in a more appropriate way. This due to the fact that it is more likely that the desorption rate coefficient differs for different radionuclides with different distribution coefficients, than does the adsorption rate coefficient.

Mass transfer of dissolved radionuclide from stream water to surface sediment, $k_{12\text{-}diss}$

The considered processes that will transfer dissolved radionuclides from the stream water to the surface sediment are diffusion and advection.

The diffusive flux over the sediment/stream water interphase (A_{sed}) is stated as:

$$\frac{D c_d A_{sed} \eta}{\Delta z_1 / 2} \tag{14}$$

where D is the diffusion coefficient (m^2/s), η the sediment porosity [–] and Δz_1 the depth of the surface sediment (m). The area of the sediment/stream water interphase, A_{sed}, is assumed constant by time and is determined as $\Delta x \overline{P}$ where $\overline{P}$ is the yearly mean value of the wetted perimeter.

The advective flux over the sediment/stream water interphase (A_{sed}) is stated as:

$$V_Z c_d \frac{A_{sed}}{2} \tag{15}$$

where V_Z is the advective velocity into the sediment due to pressure variations on the sediment surface (m/s) (see also Appendix A). The reason for using half of the interphase area between the stream water and the surface sediment is that half of the area is assumed to be subjected to inflow into the sediment and the other half is assumed to be subjected to flow out of the sediment.

The amount of dissolved radionuclide in the stream water is stated as:

$$c_d A_{cross} \Delta x \tag{16}$$

where Δx is the length of the stream-reach (m).

Using the above expression the rate coefficient due to advection and diffusion into the sediment can be stated as:

$$k_{12-diss} = \frac{2D\eta}{\Delta z_1 R_h} + \frac{V_z}{2R_h} \tag{17}$$

Mass transfer of dissolved radionuclide from surface sediment to stream water, $k_{21\text{-}diss}$

The transport of dissolved solutes out from the sediment to the stream water is due to the same processes as for the inflow in the sediment i.e. advection and diffusion and the flux terms are therefore almost identical.

The diffusive flux over the sediment/stream water interphase (A_{sed}) is stated as:

$$\frac{Dc_{d,sed} A_{sed} \eta}{\Delta z_1 / 2} \tag{18}$$

where $c_{d,sed}$ is the dissolved radionuclide concentration in the surface sediment (Bq/m^3).

The advective flux over the sediment/stream water interphase (A_{sed}) is stated as:

$$V_Z c_{d,\,sed} \frac{A_{sed}}{2} \tag{19}$$

The amount of radionuclide in the surface sediment is stated as:

$$c_{d\,,\,sed}\,(1+K_B \rho_{sed}) \forall_{sed,1} \tag{20}$$

where $\forall_{sed,1}$ is the volume of the surface sediment ($=A_{sed}\Delta z_1$) (m^3), ρ_{sed} the sediment density (kg/m^3) and K_B the distribution coefficient in the sediment defined as:

$$K_B = \frac{c_{a,sed}}{c_{d,sed}\rho_{sed}} \tag{21}$$

where $c_{d,sed}$ is the dissolved radionuclide concentration (Bq/m^3) and $c_{a,sed}$ is the adsorbed radionuclide concentration (Bq/m^3).

The rate coefficient due to advection and diffusion of dissolved radionuclides from the sediment can then be stated as:

$$k_{21-diss} = \frac{2D\eta P\Delta x}{\Delta z_1 (1 + K_B \rho_{sed}) \forall_{sed,1}} + \frac{V_z P \Delta x}{2 (1 + K_B \rho_{sed}) \forall_{sed,1}} \quad (22)$$

Mass transfer of adsorbed radionuclide from stream water to surface sediment, $k_{12\text{-}ads}$

The processes responsible for transport of adsorbed radionuclides from the stream water to the sediment is due to sedimentation of particulate material onto which radionuclides are adsorbed.

The flux of radionuclide from the stream water to the surface sediment is stated as:

$$c_d K_d c_p V_{partsed} A_{sed} \quad (23)$$

where $V_{partsed}$ is the sedimentation velocity (m/s).

The amount of radionuclide in the stream water is stated as:

$$c_d (1 + K_d c_p) A_{cross} \Delta x \quad (24)$$

The rate coefficient describing the transfer of adsorbed radionuclides to the sediment can then be stated as:

$$k_{12\_ads} = \frac{K_d c_p V_{partsed}}{(1 + K_d c_p) R_h} \quad (25)$$

Mass transfer of adsorbed radionuclide from surface sediment to stream water, $k_{21\text{-}ads}$

The concentration of suspended material is assumed constant in the stream water, which implies that the sedimentation and resuspension of particulate matter should be of equal size, i.e. the flux of particulate matter to the bed should be as large as the flux of resuspended particulate matter.

In this case, the sedimentation and resuspension flux of particulate matter can be stated as:

$$Sedimentation = Resuspension = c_p V_{partsed} A_{sed} \quad (26)$$

The flux of adsorbed radionuclide from the surface sediment to the stream water is stated as:

$$c_{d,\,sed} K_B \; Resuspension \quad (27)$$

The total mass of radionuclide in surface sediment:

$$c_{d,\,sed} (1 + K_B \rho_{sed}) \forall_{sed,1} \quad (28)$$

The rate coefficient describing the transfer of adsorbed radionuclides from the surface sediment to the stream water is then:

$$k_{21\_ads} = \frac{Resuspension\, K_B}{(1 + K_B \rho_{sed}) \forall_{sed,1}} \tag{29}$$

where the expression for Resuspension is found in Equation (26) when the assumption that resuspension and sedimentation balances.

Mass transfer of radionuclide from surface sediment to deep sediment, k_{23}

The exchange of radionuclides between the surface and deep sediment is due to advection and diffusion. As the concentration of suspended particulate matter in the stream water is assumed to be constant, the resuspension and sedimentation of particulate matter (not radionuclide) in the surface sediment balances. Therefore no exchange due to sedimentation or resuspension in the deeper sediment is considered. However, if the assumption of a constant suspended concentration is not done, an additional exchange term due to sedimentation and resuspension has to be considered also in this sediment layer.

The diffusive flux over the surface and deep sediment interphase is:

$$\frac{D c_{d,sed} A_{deep} \eta}{\Delta z_1 / 2} \tag{30}$$

where A_{deep} is the area of the deeper sediment equal to the area of the surface sediment A_{sed} (m^2).

The advective flux is stated as:

$$V_Z c_{d,sed} \frac{A_{deep}}{2} \tag{31}$$

The mass of radionuclides in the surface sediment as:

$$c_{d,sed}(1 + K_B \rho_{sed}) \forall_{sed,1} \tag{32}$$

The rate coefficient can then be stated as:

$$k_{23} = \frac{2 D \eta A_{deep}}{\Delta z_1 (1 + K_B \rho_{sed}) \forall_{sed,1}} + \frac{V_z A_{deep}}{2\,(1 + K_B \rho_{sed}) \forall_{sed,1}} \tag{33}$$

Mass transfer of radionuclide from deep sediment to surface sediment, k_{32}

As for the transport of radionuclides into the deep sediment, the transport from the deep sediment to the surface sediment is due to advective and diffusive transport.

The diffusive flux over the interphase surface/deep sediment is:

$$\frac{D c_{d,sed2} A_{deep} \eta}{\Delta z_2 / 2} \tag{34}$$

where $c_{d,sed2}$ is the radionuclide concentration in the deeper sediment (Bq/m^3) and Δz_2 is the depth of the deep sediment (m).

The advective flux is stated as:

$$V_Z c_{d,sed2} \frac{A_{deep}}{2} \tag{35}$$

The mass of radionuclide in the deep sediment is stated as:

$$c_{d,sed2}(1+K_B\rho_{sed})\forall_{sed,2} \tag{36}$$

where $\forall_{sed,2}$ is the volume of the deeper sediment (=$A_{deep}\Delta z_2$) (m^3).

The transfer could then be stated with the following rate coefficient:

$$k_{32} = \frac{2D\eta A_{deep}}{\Delta z_2 (1+K_B\rho_{sed})\forall_{sed,2}} + \frac{V_z A_{deep}}{2(1+K_B\rho_{sed})\forall_{sed,2}} \tag{37}$$

Mass transfer of radionuclide between stream water and biota, k_{14} and k_{41}

The transfer of radionuclides between the stream water and biota is assumed to occur for dissolved radionuclides. The exchange is formulated with time varying rate coefficients by means of equilibrium conditions corresponding to the bioconcentration factor, BCF ((Bq/kg fw)/(Bq/l water)), that can be found in the literature for different radionuclides and biota. A constant standing crop of total biomass along the stream is assumed, $M_{biomass,\,tot}$ (kg fw), with no change of biomass over the year.

The rate coefficients is then stated as:

$$k_{14} = BCF \frac{M_{biomass,\,tot}}{\forall_{water}} k_{41} \tag{38}$$

$$k_{41} = k_{biota} \tag{39}$$

where $\forall_{water}$ is the volume of water (dm^3). For simulating fast equilibrium uptake, k_{41}, is set to a large value (=1 $year^{-1}$).

Outflow rates of radionuclide from the stream water, k_{out} and k_{flood}

The outflow of both dissolved and adsorbed radionuclides from the stream water is due to longitudinal outflow with the running water in the downstream direction and due to flooding of the main stream channel in the lateral direction.

The flux due to outflow in the longitudinal direction can be stated as:

$$V_{adv} c_{tot} A_{cross} \tag{40}$$

where c_{tot} is the total concentration of radionuclides in the stream water (Bq/m^3).

The flux of the outflow due to flooding in the lateral direction is stated as:

$$q_{flooded\,water} c_{tot} \tag{41}$$

where $q_{flooded\,water}$ is the flow of flooded water out from the stream channel expressed as a constant mean value over the year (m^3/s). The flow of flooded water is determined by calculating the difference of the flow determined by the specific runoff and the area of the

watershed and the maximum flow that can flow in the main channel before flooding occurs. The maximum flow in the main channel is calculated by Equation (2) with the defined maximum depth, y_{max}, of the main channel. The differences in the two calculated flows are added up over the year and is evenly distributed over the year as a constant outflow in the lateral direction, $q_{flooded\ water}$.

The mass of radionuclide in the stream water box is stated as:

$$c_{tot} A_{cross} \Delta x \tag{42}$$

For the case when flooding occurs, the total cross sectional area over which the concentration is defined is equal to a virtual area that should exist if no maximum height were defined for the main stream channel.

The outflow rates due to advection in the longitudinal direction and due to flooding in the lateral direction can than be stated as:

$$k_{out} = \frac{V_{adv}}{\Delta x} \tag{43}$$

where $V_{adv}=Q/A_{cross,}$ $Q=q_sA_{ws}$ and A_{cross} is determined according to Equation (4).

$$k_{flood} = \frac{q_{flooded\ water}}{\Delta x\ A_{cross}} \tag{44}$$

Outflow rate of radionuclide from the surface sediment, k_{sed}

On the uppermost surface of the sediment, sediment particles will move due to rolling and saltation as an effect of the flowing water. This movement causes a transport from the surface sediment out from the considered system also of radionuclides that are adsorbed onto the sediment particles. Several semi-empirical formulas have been presented in the literature in an effort to be able to describe sediment transport in streaming waters /see e.g. Chanson 1999/. The bed-load transport in this model is based on a relationship by /van Rijn 1984/ wich relates the bed load transport to geometrical dimensions, sediment characteristics and critical mean flow velocity based on Shield's criterion. In the present model, it is assumed that the particulate transport on the bed-surface is mainly constituted of minerals, whereas the particulate transport in the stream water is mainly of organic material.

According to expressions given by /van Rijn 1984/ the critical mean flow velocity, above which sediment motion will occur, can for particles in the range of 100–2,000 μm be determined according to /van Rijn 1984/:

$$\bar{u}_{cr} = 0.19(D_{50})^{0.1} \log\left(\frac{12R_b}{3D_{90}}\right) \quad for\ 100 \le D_{50} \le 500\mu m \tag{45}$$

$$\bar{u}_{cr} = 8.5(D_{50})^{0.6} \log\left(\frac{12R_b}{3D_{90}}\right) \quad for\ 500 \le D_{50} \le 2000\mu m \tag{46}$$

where $\bar{u}_{cr}$ is the critical flow velocity based on Shield's criterion (m/s), R_b the hydraulic radius (m), D_{50} and D_{90} the diameter of the sediment particles for the 50% and 90% percentiles (m).

According to /van Rijn 1984/, the sediment transport can then be stated as:

$$q_b = \bar{u}\, d\, 0.005 \left(\frac{\left(\bar{u} - \bar{u}_{cr}\right)}{\left[(s-1) g D_{50}\right]^{0.5}} \right)^{2.4} \left(\frac{D_{50}}{d} \right)^{1.2} \tag{47}$$

where q_b is the bed-load transport ($m^3/(m,s)$), s specific density (density of sediment/density of water) [–], d the water depth (m) and ū is the mean flow velocity in the stream channel (m/s).

The flux of radionuclides out of the surface sediment by bed load transport can then be stated as:

$$q_b w c_{a,sed} = q_b w \frac{c_{tot,sed}}{\left(1 + \frac{1}{\rho_{sed} K_B}\right)} \tag{48}$$

where w is the width of the stream (the maximum width of the triangular channel is used in the calculations) (m) and $c_{tot,sed}$ is the total radionuclide concentration in the surface sediment (Bq/m^3). As the bed-load transport is determined for mineral particles, an under-estimation of the radionuclide transport will be done when the adsorbed radionuclide concentration in the sediment (with a mean density of both the organic and mineral material) is multiplied with the bed-load transport of particulate material. On the other hand, the bed-load transport of radionuclides is probably overestimated in the sense that it is probable that the radionuclide concentration is higher on organic material than on minerals, a matter that is not considered within the proposed model.

The mass of radionuclide in the surface sediment is:

$$c_{d,sed}(1 + K_B \rho_{sed}) \forall_{sed,1} = c_{tot,sed} \forall_{sed,1} \tag{49}$$

Rate coefficient describing transport of adsorbed mass out of surface sediment is then:

$$k_{sed} = \frac{q_b w}{\forall_{sed,1} \left(1 + \frac{1}{\rho_{sed} K_B}\right)} \tag{50}$$

The results from the above mentioned equations for the bed-load transport should only be considered as an estimation of the order of magnitude of the sediment transport. The equations are based on laboratory experiments and only valid in certain parameter intervals /see van Rijn 1984/. However, in the absence of measurements, these equations are used as an estimation. If it is found after application of site-specific data that the hydraulic conditions in the system is outside the defined valid range of the bed-load equations by /van Rijn 1984/, further considerations should be done whether to implement alternative model equations or if the presented equation still can be used for an order of magnitude estimation.

A summary of the model transfer functions is found in Table 3-1.

Table 3-1. Summary of model transfer functions.

Rate	Transfer function
k_{ads}	$\frac{\ln(2)}{T_k}$
k_{des}	$\frac{\ln(2)}{T_k c_p K_d}$
$k_{12-diss}$	$\frac{2D\eta}{\Delta z_1 R_h} + \frac{V_z}{2R_h}$
$k_{21-diss}$	$\frac{2D\eta P\Delta x}{\Delta z_1 (1 + K_B \rho_{sed}) \forall_{sed,1}} + \frac{V_z P \Delta x}{2 (1 + K_B \rho_{sed}) \forall_{sed,1}}$
k_{12_ads}	$\frac{K_d c_p V_{partsed}}{(1 + K_d c_p) R_h}$
k_{21_ads}	$\frac{Resuspension\, K_B}{(1 + K_B \rho_{sed}) \forall_{sed,1}}$
k_{23}	$\frac{2D\eta A_{deep}}{\Delta z_1 (1 + K_B \rho_{sed}) \forall_{sed,1}} + \frac{V_z A_{deep}}{2 (1 + K_B \rho_{sed}) \forall_{sed,1}}$
k_{32}	$\frac{2D\eta A_{deep}}{\Delta z_2 (1 + K_B \rho_{sed}) \forall_{sed,2}} + \frac{V_z A_{deep}}{2 (1 + K_B \rho_{sed}) \forall_{sed,2}}$
k_{14}	$BCF \frac{M_{biomass,tot}}{\forall_{water}} k_{41}$
k_{41}	k_{biota}
k_{out}	$\frac{V_{adv}}{\Delta x}$
k_{flood}	$\frac{q_{flooded\ water}}{\Delta x\ A_{cross}}$
k_{sed}	$\frac{q_b w}{\forall_{sed,1} \left(1 + \frac{1}{\rho_{sed} K_B}\right)}$

4 Model parameters

The proposed model contains three different types of parameters: radionuclide specific, site specific and constant values. For some of the parameters the classification of a parameter is not self-evident and more than one category could be applied to the same parameter. In Table 4-1 the model parameters and their assigned classification are listed. The parameters assigned constant literature values are also in reality a type of site-specific parameters. However, application of site-specific information on these parameters is assumed to be of less importance than for those categorised as site-specific. Parameters included in the model equations such as e.g. the geometrical parameters: hydraulic radius (R_h), cross-sectional area (A_{cross}) and volume of the sediment compartments ($\forall_{sed,i}$) are derived from several of the parameters in the table and are therefore not tabulated in Table 4-1.

This report does not contain a complete determination of the parameter values needed to run the model as not all site-specific information that is needed was available at the time for the model development. However, during the performed sensitivity analyses (presented in Chapter 5), an attempt was made to use parameter values with an order of magnitude that would be probable. In the future when new information becomes available from e.g. field investigations, the parameter values in the model and thereby the calculations should be updated. The order of magnitudes of the different parameters are discussed in the following sections and the values used during the sensitivity analyses are given in Table 5-1.

Table 4-1. Parameters in the model.

Description	Parameter	Type of parameter
Distribution coefficient in the stream water	K_d	Radionuclide specific
Distribution coefficient in the sediment	K_B	Radionuclide specific
Radioactive half-lives	$T_{1/2}$	Radionuclide specific
Half-time to reach sorption equilibrium	T_k	Radionuclide specific
Bioconcentration factor	BCF	Radionuclide specific
Specific run-off	q_s	Site specific
Watershed area	A_{ws}	Site specific
Cross-sectional angle	α	Site specific
Slope of the channel	S_b	Site specific
Length of the channel	Δx	Site specific
Maximum depth in main channel	y_{max}	Site specific
Sediment density	ρ_{sed}	Site specific
Advective transport velocity in bed sediment	V_z	Site specific
Particle size distribution in the sediment	D_{50}, D_{90}	Site specific
Suspended particulate matter in stream water	c_p	Site specific
Sediment porosity	η	Site specific
Diffusion coefficient	D	Constant literature value
Depth of surface and deep sediment	Δz_i	Constant literature value
Plant biomass	$M_{biomass}$	Constant literature values
Manning friction coefficient	n	Constant literature value
Sedimentation velocity	$V_{partsed}$	Constant literature value

4.1 Radionuclide specific parameters

The formulated model concept contains five different radionuclide specific parameters, the distribution coefficients in the stream water and in the bed sediment, the half-lives of the radionuclides, the half-time to reach sorption equilibrium in the stream water and the bioconcentration factors in different types of plants.

The radionuclide specific parameters included in the model must naturally be changed when different radionuclides are considered. The availability of data on the radionuclide specific parameters differs between the different parameters. For example, the half-lives of the radionuclides are uniquely given and tabulated in the literature, whereas data of the reaction time to reach sorption equilibrium or the bioconcentration factors are more sparse and not as uniquely given.

For order of magnitudes of the distribution coefficients during the sensitivity analyses, the compilation of distribution coefficients for suspended matter in lakes by /Bergström et al. 1999/ is used as a basis. The best estimate of this coefficient for the radionuclides listed is within the range 0.001–100 m^3/kg. For iodine, the best estimate is given to 0.3 m^3/kg, which will be used during some sensitivity analyses in combination with a bioconcentration factor for iodine (see below). The distribution coefficient is of course dependent on environmental conditions such as pH and redox potential and can therefore vary from site to site. Also the distribution coefficient in the sediment is not necessarily the same as the distribution coefficient in the stream water. In the following sensitivity analyses the distribution coefficient in the sediment has been assumed to be 10% of the distribution coefficient in the stream water in most of the performed simulations, as the distribution coefficient in the sediment often is smaller. This approach has been applied also in other studies as field measurements have suggested that the distribution coefficient in the sediment is smaller /IAEA 2001/. The reason for the smaller coefficient in the bed sediment is partly due to the higher relative abundance of particle in the sediment than in the surface water and the fact that the particles in the sediment are relatively larger. However, in the present study also the case when the distribution coefficients are set equal is investigated.

Values on the reaction time to reach sorption equilibrium for different radionuclides reported in the literature are either determined by means of laboratory batch tests or by calibrating a proposed sorption model to field data. /Karlsson et al. 2001/ lists the range 10^{-5}–10^{-1} year on the reaction half-time where the longer time is based on information of the slow caesium sorption to clay.

Bioconcentration factors in freshwater or marine environment can be found for different radionuclides and different plants in the literature see e.g. compilation made within the FASSET project /Brown et al. 2003a/ or by IAEA /IAEA 2004/. The concentration factors for macrophytes in freshwater systems listed in the FASSET report is within the range 70–14,600 (Bq/kg fw)/(Bq/l) for the radionuclides ^{36}Cl, ^{90}Sr, I, ^{137}Cs, ^{219}Po, ^{226}Ra, ^{230}Th, U, ^{230}Pu and ^{240}Pu i.e. rather extensive variations exist between different nuclides and different determinations. For iodine the BCF factor 200 (Bq/kg fw)/(Bq/l) is given, which will be used during some part of the sensitivity analyses in combination with the distribution coefficient for iodine. This to be able to test the model for a significant combination of BCF and K_d values.

4.2 Site specific parameters

To the site specific parameters we include information about the watershed, specific runoff, geometry of the stream channel, sediment characteristics and the advective velocity in the sediment.

The model is intended to be used for predictions of radionuclide transport in the water-courses that are situated within the two sites, Simpevarp/Laxemar and Forsmark, currently under investigation for their suitability as a place for radioactive repository. At this time, no detailed information of the stream morphology e.g. cross-sections and slopes are available. Therefore during the sensitivity analyses, a general shape of the cross section and the large scale slope is assumed constant along the stream reach and is partly based on large-scale GIS-information from the area. The large-scale slope of the stream channels in the areas is approximately a few per mille. The watercourses in consideration have watersheds of maximum a few square kilometres and a length of approximately a couple of km. In the absence of site-specific information of the cross-sectional geometry, a triangular shape is assumed and the effect of the magnitude of the angel, α, on the hydraulic calculations are investigated in the range 15–80°. For all cases, the maximum depth of the main channel, y_{max}, is assumed to 1 m.

For the sensitivity analyses, the monthly mean values of the specific discharge at Vattholma (station 50110) for the period 1917–2000 (Figure 4-1), is used together with an assumption of a watershed of 4 km^2 for the generic stream that is used as to represent the streams in the present areas. An assumption is also that the flow conditions in the future will be as in the past.

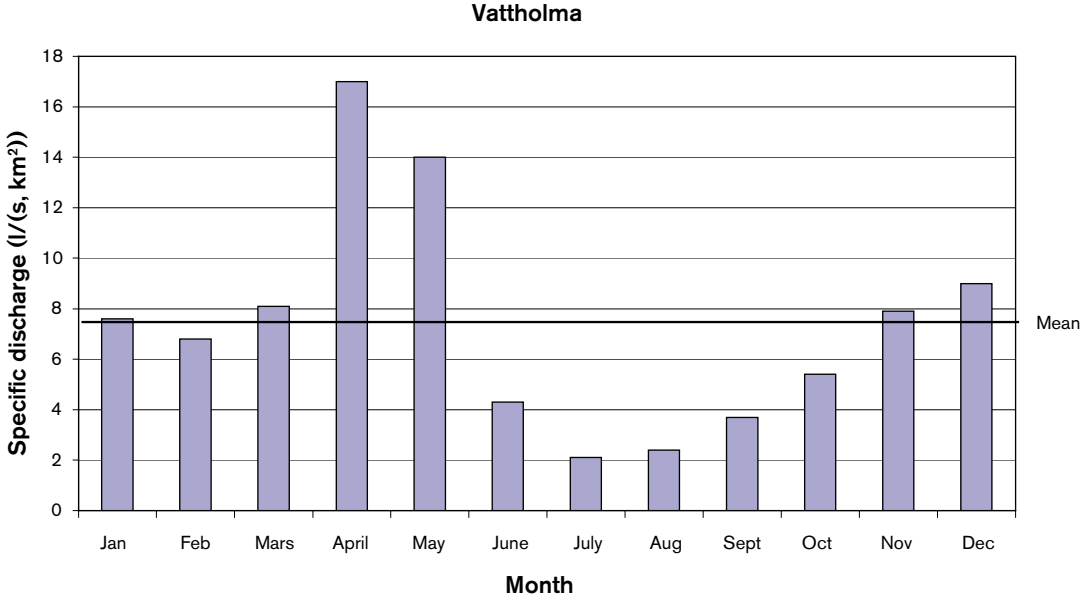

Figure 4-1. *Monthly mean values on specific discharge at Vattholma for the period 1917–2000 (station 50110) used during the model test. /From Larsson-McCann et al. 2002/.*

Also sediment characteristic such as sediment density, porosity, grain-size distribution (D_{50}, D_{90}) as well as the advective velocity in the sediment are parameters that are site specific. The sediment density is assumed to be 1,100 kg/m^3 in the sensitivity analysis, a value that should be realistic considering reported sediment densities in other stream systems in Sweden where the organic content is rather high. The bed load transport is assumed to constitute mainly of mineral particles whereas the resuspension into the stream water phase is mainly assumed to be of organic origin. For the bed-load calculations the specific density, s, is therefore assumed equal to 2.6 [–] based on the density for the mineral particles. This is because it is assumed that the bed-load transport is constituted of mineral particles with a higher density than the average density of the sediment. Lighter sediment particles e.g. organic material is instead assumed to be transported in suspended phase. In the absence of information of the sediment grain-size distribution, needed for the calculations of the bed load transport, the values 700 μm and 1,900 μm is assumed on the coefficients D_{50} and $D_{90.}$ The sediment porosity is assumed to be 0.8.

A realistic value of the advective velocity in the sediment is more complicated to obtain for a specific watercourse. As a basis for the estimation of the order of magnitude on the velocities, theories by /Wörman et al. 2002ab/ is used for the approximation. A generalization of their results to the system in the present study yields an approximation of a probable value on V_Z within the range 7×10^{-6}–8×10^{-4} m/s (see Appendix A). However, this value should be considered with great caution as the calculations are based on a number of assumptions and generalization from another watercourse. To obtain more certain and detailed information on the magnitude of this parameter, a more thorough investigation including some field measurements should be performed, but is outside the scope of the present study. In the sensitivity analyses within this study, the estimated range of the advective velocity served as a starting point for the variation of this parameter.

The suspended particulate matter in the stream water is also a parameter that is included in the model and that can be quite easily determined by field measurements. A probable order of magnitude of the particulate concentration in the considered streams is here assumed to be around 20 mg/l, which is found in literature for another small stream (Sävaån) in Sweden /Jonsson 2003/.

4.3 Parameters based on literature information (constants)

The parameters that are set to a constant value are those where literature information of the parameter value is judged to be sufficient for the calculations. The reason for this is that the variation between different systems is assumed to be of less importance than the variation of a site-specific parameter, even though the parameter is not a constant in reality. Also some parameters are difficult to measure in field and here literature information is the second best alternative.

An example of such a parameter included in the model is the Manning roughness coefficient, tabulated in most hydraulic literature /e.g. Fox and McDonald 1994/. For natural canals and rivers with many stones and weeds, a Manning roughness coefficient of 0.035 m$^{-1/3}$s is tabulated in the literature while for a uniform earth channel 0.017 m$^{-1/3}$s is given /Fox and McDonald 1994/. A probable range for the roughness coefficient in the rivers in the Forsmark or Simpevarp/Laxemar area is therefore in the range 0.02–0.04 m$^{-1/3}$s.

For the sedimentation velocity, also a constant value is assumed. The settling velocity is dependent on particle size and density and can be determined for ideal conditions by Stokes law. In the coast model by /Karlsson et al. 2001/ a best estimate of 365 m/year is used, which will correspond to the settling velocity of silt. The minimum and maximum values are given to 73 and 7,300 m/year, respectively. In the lake model by /Karlsson et al. 2001/ the best estimate of the velocity is lower, 183 m/year, due to the fact that a larger fraction of the particles in the lake is organic matter which tends to sink with a slower velocity than mineral particles. Here are the minimum and maximum values given to 36.5 and 3,600 m/year, respectively. The effect of different order of magnitudes of the sedimentation velocity, based on the order of magnitudes for the parameter given in the lake and coastal model, will be performed within the present study.

The diffusion coefficient in the sediment is also assumed to be a constant, where the order of magnitude is taken from literature. The exchange described by the diffusion coefficient in the present model is solely due to concentration differences in the pore water and stream water and is not a lumped exchange coefficient representing also exchange due to flow-induced exchange, as is common approach in solute stream transport models. A reasonable order of magnitude for the diffusion coefficient, used during the sensitivity analyses, is 1×10^{-10}–5×10^{-10} m^2/s.

The plant biomass is of course a parameter that in reality will be site-specific. For the order of magnitude estimation in this study, the amount is however based on information of standing crops in other watercourses at site-specific information was not available at the time for the sensitivity analyses of the model. An amount of 5 kg wet substance per m^2 is used in the sensitivity analyses, which is based on observations of biomass for the Rideau River in Canada (mesotropic lowland river in the littoral zone down to 2 m in September 20th to 27th, 2000, pers. comm. /Frances R Pick/) which was found to be in the range 40–4,300 g fw/m^2 under the assumption of a dry weight content of the fresh weight of 20%. For the estimation of the total biomass along the river, a width of the stream of 1 m is assumed.

The depth of the surface and deep sediment are set to each 0.5 m in the simulations. However, the penetration depth for the radionuclides is of course dependent of characteristics of the sediment. The depth in the sediment that will be influenced by exchange with the streaming water can be determined e.g. by tracer experiments, where an injection of a tracer into the stream water is followed by observations of the penetration in the sediment. The assigned depth is also a matter of definition of the model volume, i.e. where the interface between the groundwater model and the stream model is assumed to be located.

5 Model evaluation with sensitivity analyses

In this section, the results from a first set of model calculations are reported. Sensitivity analyses of parameters regulating both the hydraulic and the radionuclide transport are performed and presented, from which it is possible to gain a first idea of how the retention processes will affect the longitudinal transport.

The interpretation of the results from the sensitivity analyses is to some extent dependent on the scenario in consideration, e.g. if a short pulse or a continuous inflow of radionuclides to the system is considered. For a short pulse, the time for recovery of polluted sediment can e.g. be of interest, while for a constant inflow, the equilibrium concentration and time to saturation of the sediment will be of more interest. In the present model evaluation, a continuous constant inflow of radionuclides to the system is considered.

5.1 Results from hydraulic calculations

The normal depth in the stream channel have been calculated for the mean values of the flow calculated by means of monthly mean values of the specific run-off at Vattholma and an assumption of a 4 km^2 watershed. In Figure 5-1, the angel defining the cross section of the main stream channel has been varied to investigate the resulting variation of the normal depth for a constant flow. A decrease from 80 to 15 degrees of the angel, α, causes the normal depth to increase by a factor of approximately 4.

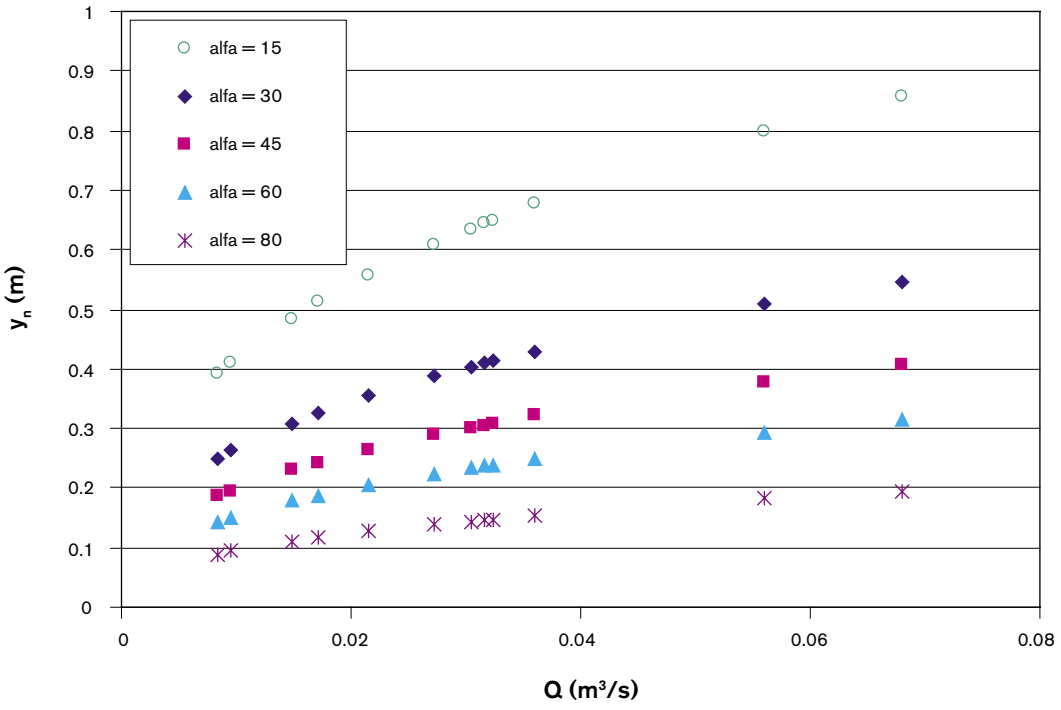

Figure 5-1. *Sensitivity analyses of the influence of the shape of the cross-section, α, on the normal depth in the stream channel for different flow conditions (S_b=0.002, A_{ws}=4 km^2, y_{max}=1 m, specific run-off according to Figure 4-1).*

Without any site-specific information about channel geometry, the determination of the normal depth will become rather uncertain. As a basis for a qualified guess of an acceptable approximation of the angle in the absence of detailed information of channel geometry, at least some site-specific information about whether the walls of the cross section are steep of flat is desirable. If site-specific information implies that another cross-sectional type, other than the triangular shape, better describes the actual stream channel, this shape could be implemented in the model. In the present study, however, all calculations are performed for a triangular cross-section, as no site-specific information is available.

A sensitivity analyses was also performed to investigate the influence of the magnitude of the Manning roughness coefficient. For the assumed watershed of 4 km^2, the water depth will at its maximum differ by approximately 10 cm (corresponding to an increase in water depth of less than 30%) when the coefficient is varied within the range of 0.02–0.04 $m^{-1/3}s$, which is assumed to be the upper and lower limit for the stream channel (Section 4.3) (Figure 5-2). The exact value of the roughness coefficient is therefore not considered to be crucial for the prediction of the radionuclide transport and a constant value of 0.03 $m^{-1/3}s$ was used in the further model tests. Also this parameter is not easily determined by field measurements.

In Figure 5-3 the channel slope has been varied from 0.5 per mille to 1 percent to illustrate the influence of the channel slope on the calculated water depths. The slope of 0.002 is based on information of large-scale slopes for a typical stream in the Forsmark area.

A conclusion from the sensitivity analyses of the hydraulic part of the model is that the uncertainty introduced in the model simulations due to the lack of site-specific information of the channel geometry is probably larger than the uncertainty in the exact value of the Manning roughness coefficient. The sensitivity analyses shows that a variation of a single hydraulic parameter, within the intervals tested during the sensitivity analyses, provides in all cases a change in the normal depth that that is less than a factor 10.

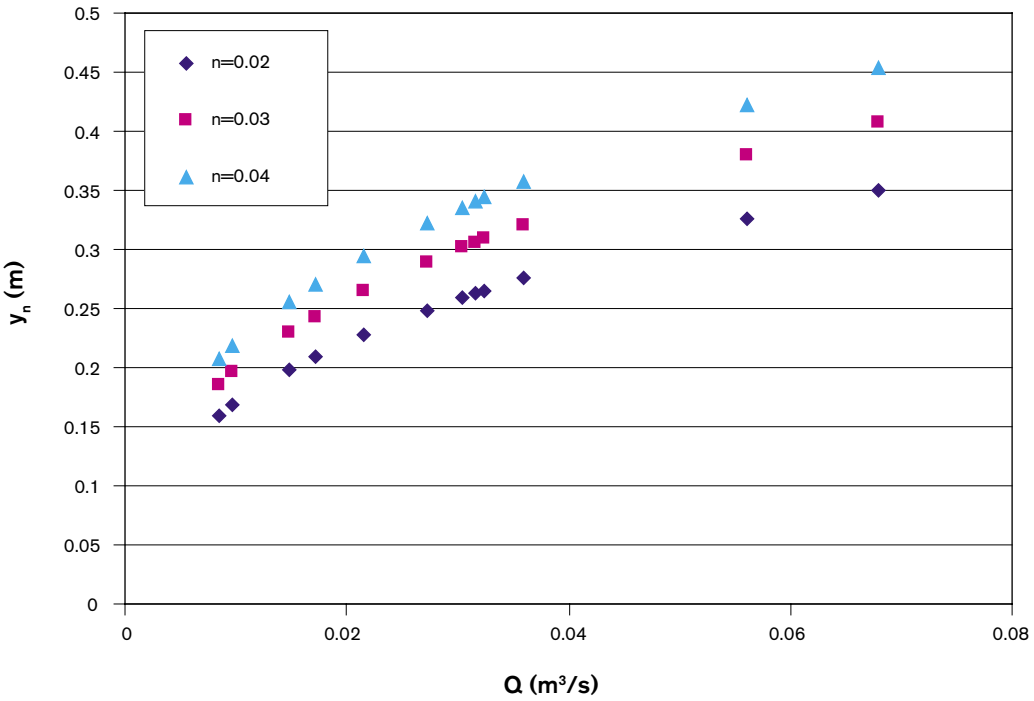

Figure 5-2. *Sensitivity analyses of the influence of Manning roughness coefficient, n, on the calculated normal depth in the stream channel for different flow conditions (α=45°, S_b=0.002, A_{ws}=4 km^2, y_{max}=1 m, specific run-off according to Figure 4-1).*

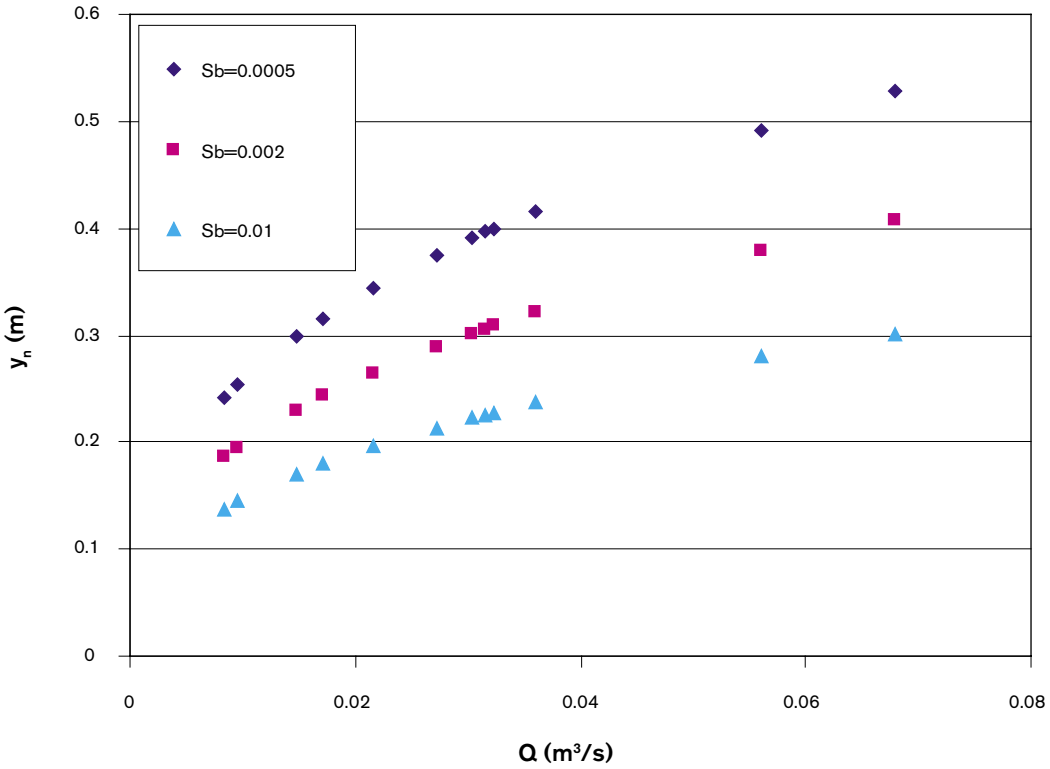

Figure 5-3. *Sensitivity analyses of the influence of the slope, S_b, on the calculated normal depth in the stream channel for different flow conditions (α=45°, n=0.03 $m^{-1/3}s$, A_{ws}=4 km^2, y_{max}=1 m, specific run-off according to Figure 4-1).*

The slope of the channel could be based on large-scale slopes in the area determined from elevation maps in the absence of field measurements. For information about cross sectional shape, useful site-specific information requires most often field measurements.

5.2 Results from radionuclide transport calculations

5.2.1 Effect of time varying parameters and different averaging alternatives

An investigation was performed to evaluate the effect on the simulation results of time-varying parameters and used averaging method during the calculations. The different investigated alternatives involve the use of the following, during the simulations:

1) Monthly mean values of the rate coefficients.
2) Yearly mean values of the rate coefficients.
3) Yearly mean value of the water flow (before the determination of the rate coefficients).

Due to the non-linear relationship between some parameters in the Manning equation, averaging operations in different stages of the calculations will yield different results. The matter to investigate is whether these differences are as pronounced as to justify the inclusion of a time-varying flow over the year for the order of magnitude calculations within this study.

The parameters used during the test of time-varying conditions are given in Table 5-1. In the performed tests, identical values are used for the parameters appearing in both the surface and deep sediment e.g. sediment density and porosity.

The results from the calculations could be found in Figure 5-4 to Figure 5-9, where the amount or concentration of radionuclides in different compartments are plotted for the three alternatives described above. From Figure 5-4 and Figure 5-5, it is possible to draw the conclusion that the necessity of including time-varying conditions is not crucial for the determination of the amount of radionuclides in the stream water. However, if the results instead are presented as concentration, as in Figure 5-6 and Figure 5-7, the difference will be slightly larger. This is due to the fact that the determination of the volume of water differs in the three cases. For the sediment, the results are given in Figure 5-8 and Figure 5-9, where also rather moderate differences for the three cases are found. Considering these results, the uncertainty introduced by using constant flow conditions instead of time-variable conditions over the year does not seem to be crucial for the order of magnitude predictions at this stage. Further model test when the distribution coefficients were set equal (K_d=K_B=0.3 m^3/kg) or higher (K_d=10 m^3/kg, K_B=1 m^3/kg) gave similar results concerning the effect of time-varying flow. Some of the resulting concentration variations for the case with higher distribution coefficients (K_d=10 m^3/kg, K_B=1 m^3/kg) are found in Figure 5-10 to Figure 5-12. From these figures it is also clear that the time to reach equilibrium conditions is longer than for the case with the lower values on K_d and K_B. Higher distribution coefficients yield naturally also higher radionuclide concentration in the sediment.

Considering the relative small difference between the different averaging methods, constant flow conditions are applied in the remainder of the performed model evaluation. Though, inclusion of time-varying flow can become important if more exact predictions are needed. However, before such a prediction is possible, the uncertainty introduced by e.g. lack of site-specific information must be dealt with.

Table 5-1. Parameter values applied during the sensitivity analyses of time-varying parameters.

Parameter	Value	Parameter	Value
K_d	0.3 (m^3/kg)	V_z	1×10^{-5} m/s
K_B	0.03 (m^3/kg)	D_{50}, D_{90}	700; 1,900 µm
$T_{1/2}$	Decay not considered (Θ=0)	s	2.6 [–]
T_k	1×10^{-3} year	c_p	0.02 kg/m^3
BCF	200 (Bq/kg fw)/(Bq/l)	D	5×10^{-10} m^2/s
A_{ws}	4 km^2	η	0.80 [–]
α	45°	Δz_i	0.5 m
S_b	0.002 [–]	$M_{biomass}$	5 kg fw/m^2
Δx	2,000 m	n	0.03 ($m^{-1/3}$s)
y_{max}	1 m	$V_{partsed}$	400 m/year
ρ_{sed}	1,100 kg/m^3	Inflow	1 Bq/year

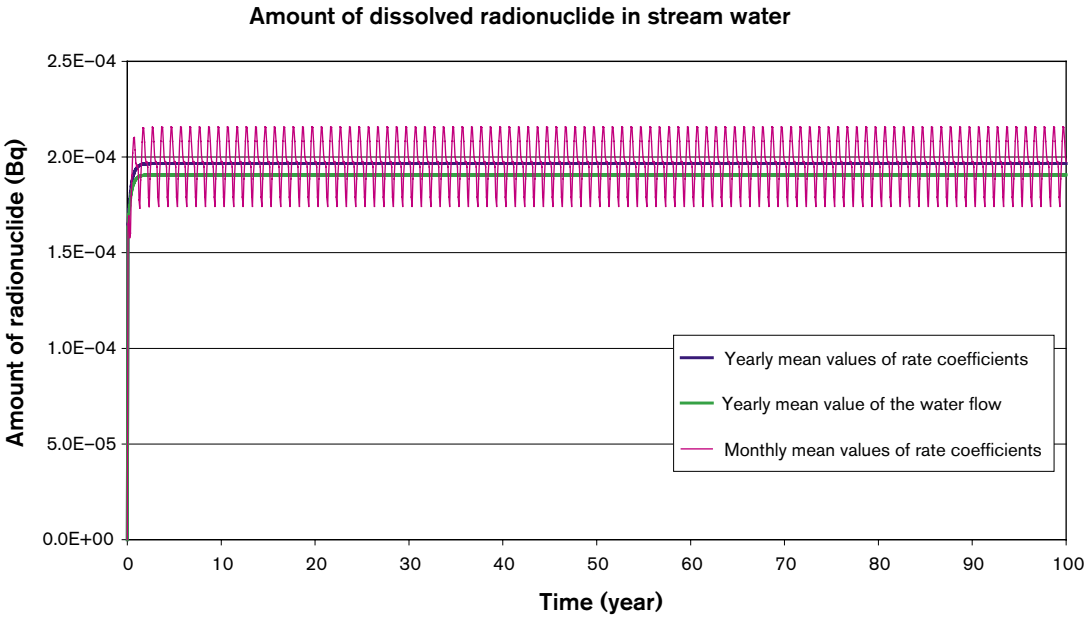

Figure 5-4. *The influence of time-varying conditions on the amount of dissolved radionuclides in the stream water. (Used parameter values in Table 5-1).*

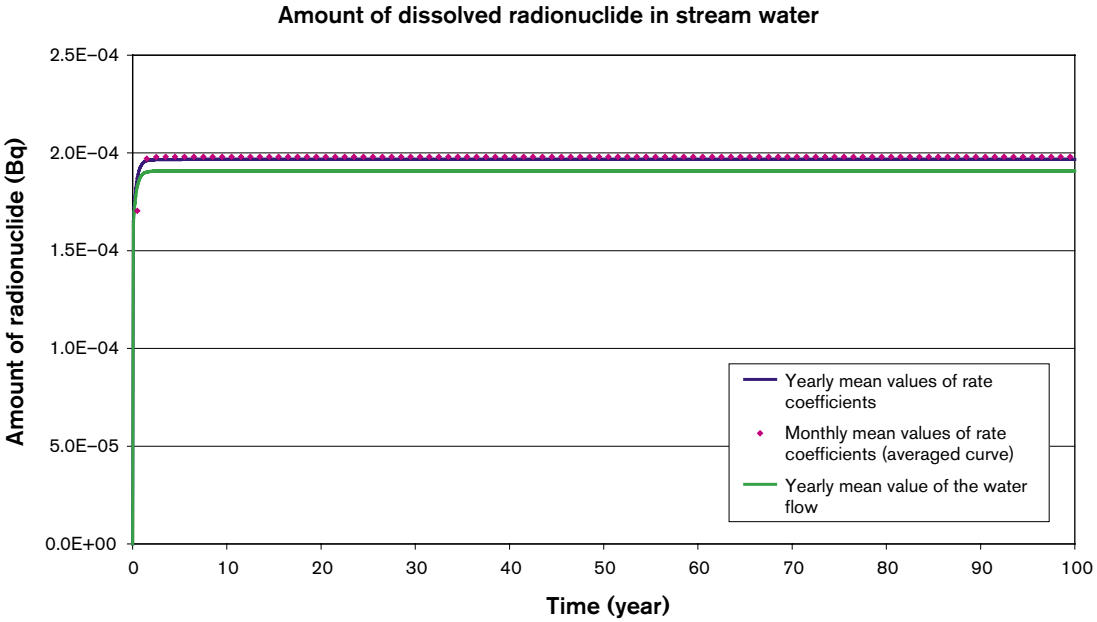

Figure 5-5. *The influence of time-varying conditions on the amount of dissolved radionuclides in the stream water. (The resulting curve using the monthly mean values have been averaged). (Used parameter values in Table 5-1).*

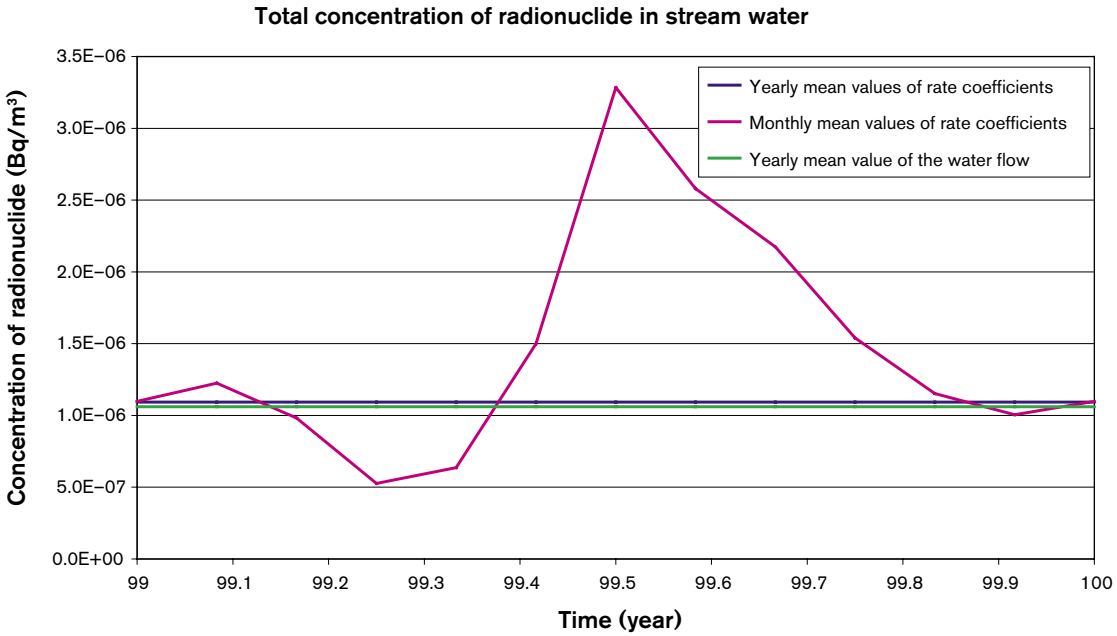

Figure 5-6. *The influence of time-varying conditions on the total concentration of radionuclides in the stream water (results shown for a year-cycle). (Used parameter values in Table 5-1).*

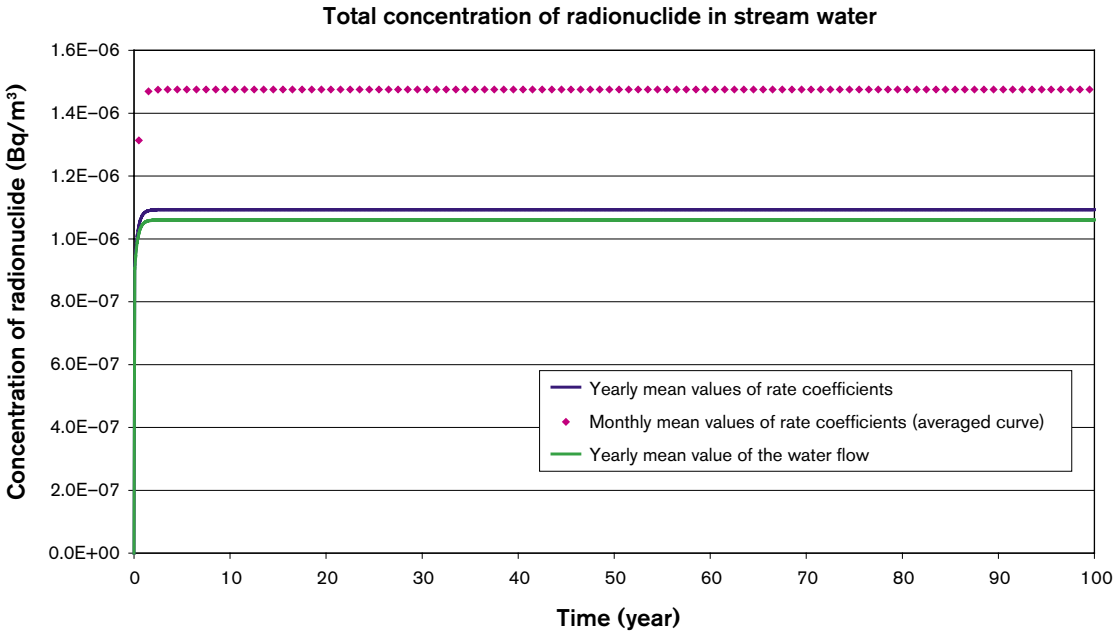

Figure 5-7. *The influence of time-varying conditions on the total concentration of radionuclides in the stream water. (Used parameter values in Table 5-1).*

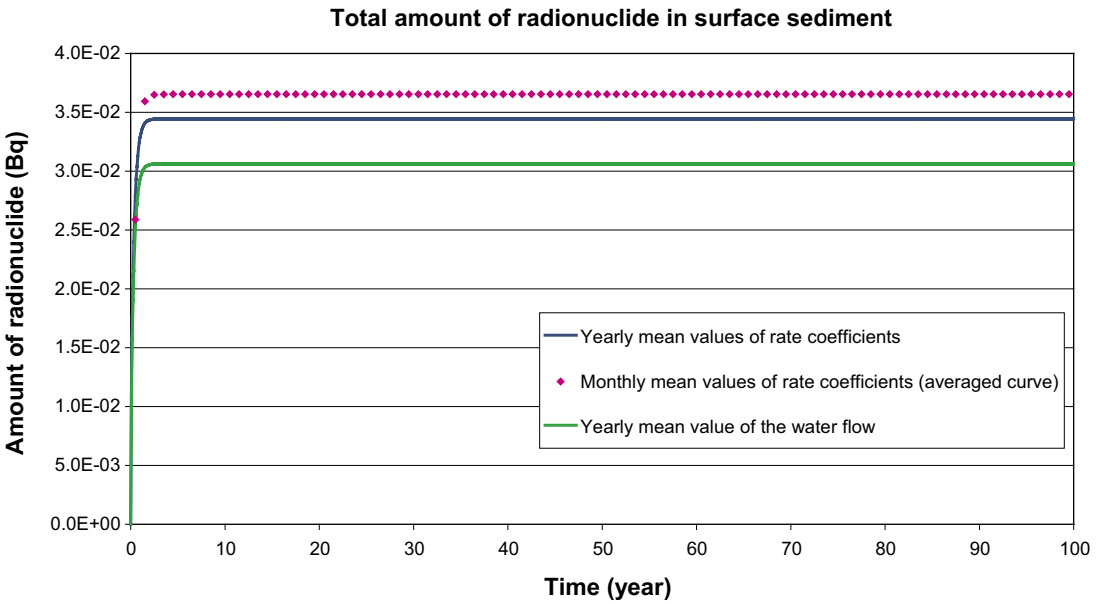

Figure 5-8. *The influence of time-varying conditions on the total amount of radionuclides in the sediment. (Used parameter values in Table 5-1).*

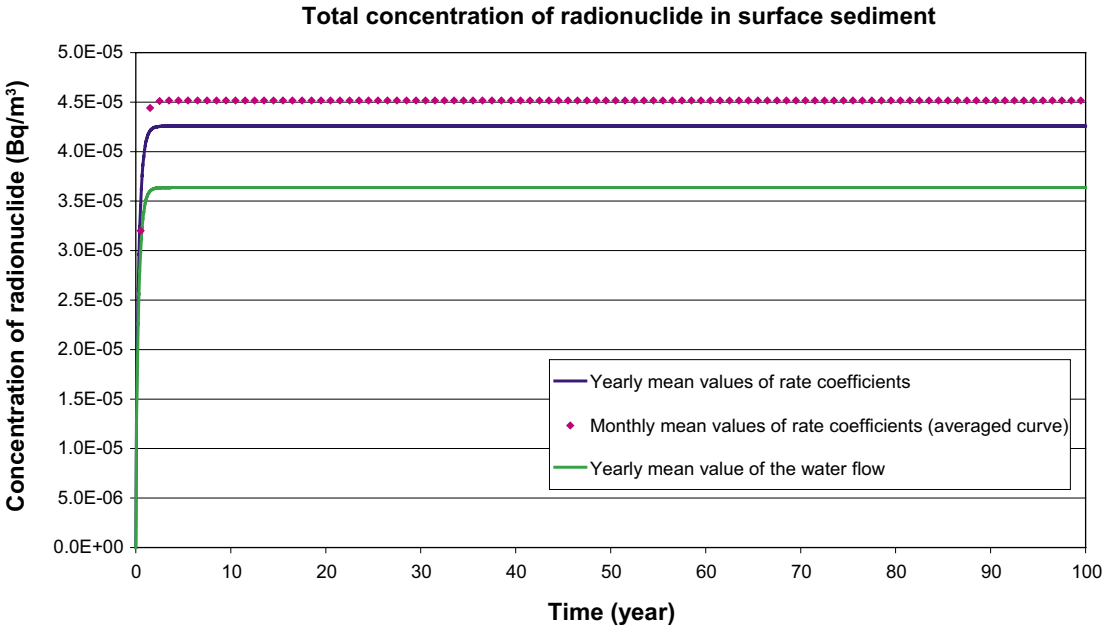

Figure 5-9. *The influence of time-varying conditions on the total concentration of radionuclides in the sediment. (Used parameter values in Table 5-1).*

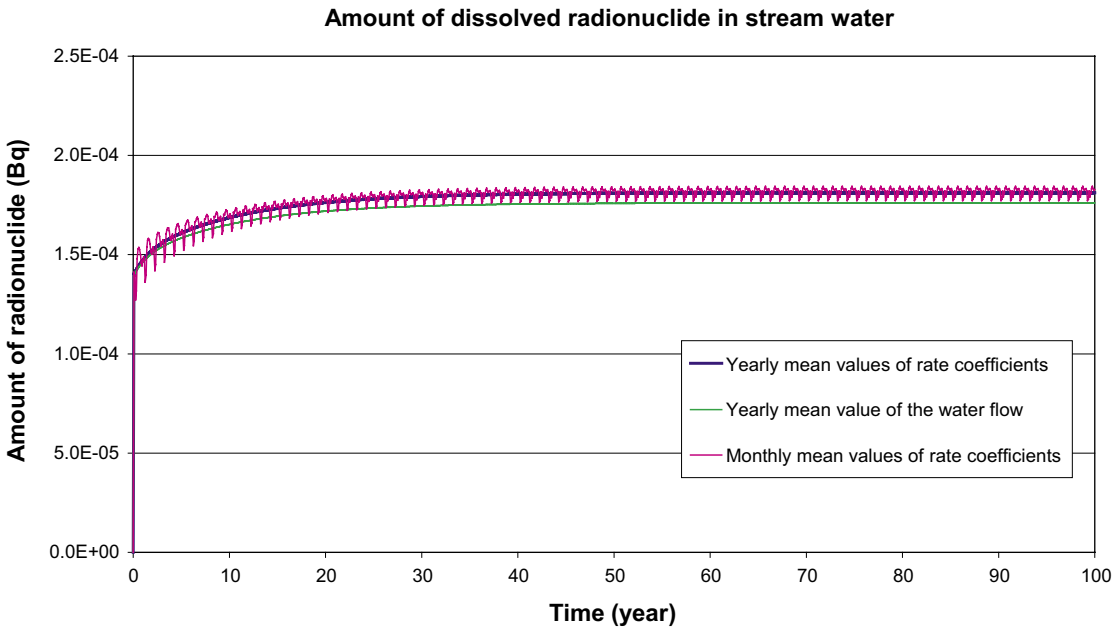

Figure 5-10. *The influence of time-varying conditions on the amount of dissolved radionuclides in the stream water. (Used parameter values in Table 5-1, except for the distribution coefficients. Here K_d=10 m³/kg, K_B=1 m³/kg).*

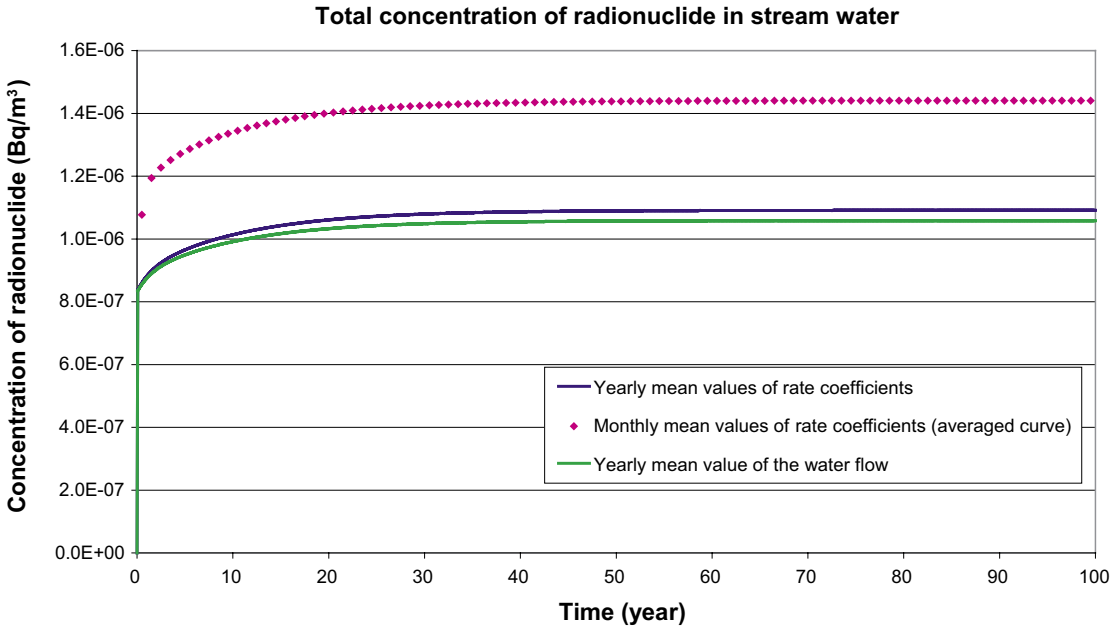

Figure 5-11. *The influence of time-varying conditions on the total concentration of radionuclides in the stream water. (Used parameter values in Table 5-1, except for the distribution coefficients. Here K_d=10 m³/kg, K_B=1 m³/kg).*

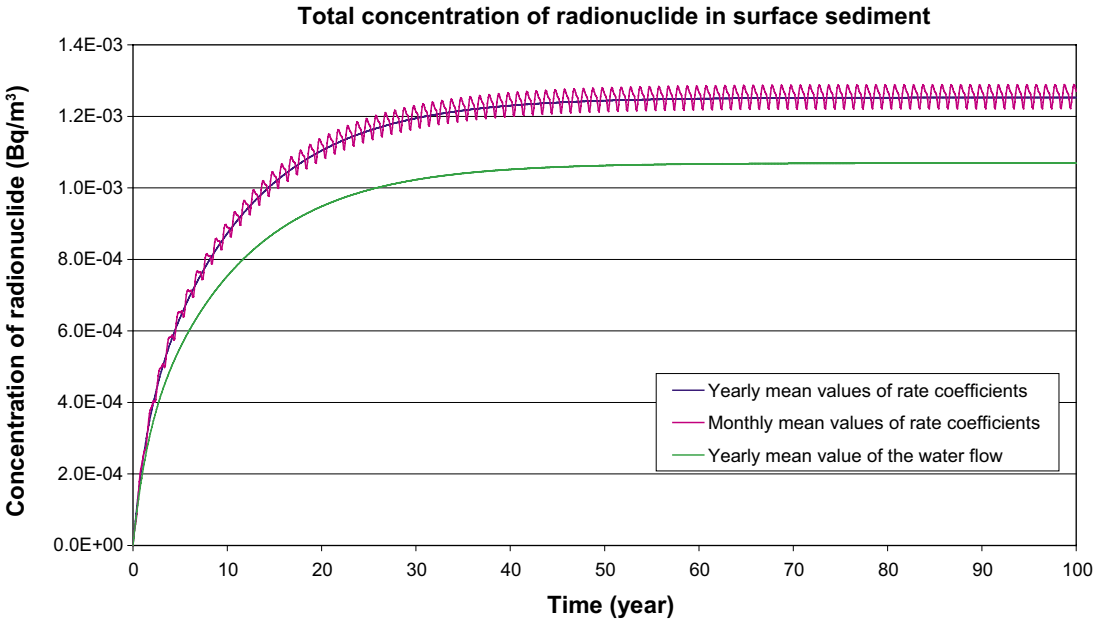

Figure 5-12. *The influence of time-varying conditions on the total concentration of radionuclides in the sediment. (Used parameter values in Table 5-1, except for the distribution coefficients. Here K_d=10 m³/kg, K_B=1 m³/kg).*

5.2.2 Sensitivity analyses of transport parameters

In this section, results from the model tests where mainly the effect on the transport due to a variation in sorption parameters, advective velocity in the sediment and sedimentation velocity have been investigated. The model test performed within this study is not to be considered as a complete sensitivity analysis, as the results might be influenced of the applied values of the parameters kept constant during the test. To get the total model uncertainty, an uncertainty analysis must be performed, where also more site specific information should be used. This is however outside the scope of the present study, and the results here should only be considered as a first sensitivity analysis of the postulated model.

By performing such an analysis, the effect of the exchange processes could be expressed as approximate ranges of the resulting radionuclide concentration in the different parts of the system. Indications could also be gained of which parts of the model that need further consideration due to a high sensitivity in the results.

If nothing else is given, the values in Table 5-1 are used for the parameters kept constant during the sensitivity analyses.

Variation of advective velocity in sediment

A sensitivity analysis of the advective velocity in the sediment is performed by varying the velocity in a range based on the estimated range given in Section 4.2. The effect of a variation of this exchange coefficient is evaluated for two sets of distribution coefficients (respectively, K_d=0.3 m³/kg, K_B=0.03 m³/kg and K_d=10 m³/kg, K_B=1 m³/kg) which will represent radionuclides with different ability to adsorb onto particulate matter. Also for the stronger sorbing solute, the effect of a change in advective velocity is investigated for two different values of the sorption reaction time (T_k), representing radionuclides with either a fast or a slow sorption mechanism.

The variation in the advective velocity, V_z, reveals that for a higher velocity, equilibrium conditions in the system will be reached more rapidly, which can be seen for all the simulated cases, exemplified by plotting the radionuclide concentration in the sediment (Figure 5-13 to Figure 5-15). For the lower distribution coefficients (K_d=0.3 m³/kg, K_B=0.03 m³/kg), the difference in time to reach equilibrium concentrations for the investigated range of the advective velocity is at its maximum in the order of a few years. The corresponding range for the more reactive radionuclide (K_d=10 m³/kg, K_B=1 m³/kg) is in the order of approximately hundred years.

For the lower distribution coefficients (K_d=0.3 m³/kg, K_B=0.03 m³/kg) in combination with a sorption reaction half-time of 1×10^{-3} year (~9 h) the equilibrium level is independent of the advective velocity (Figure 5-13). For the higher distribution coefficients (K_d=10 m³/kg, K_B=1 m³/kg), the variation in the advective velocity in the sediment will influence also the resulting equilibrium level. For a sorption reaction half-time of 1×10^{-3} year (~9 h) the concentration (or amount) in the sediment is lower for a lower advective velocity, whereas the equilibrium concentration in the sediment will be higher for a lower advective velocity when the effect of sorption kinetics is less pronounced, $T_k=1\times10^{-5}$ year (~5 min), with other conditions unchanged (Figure 5-14, Figure 5-15 and Table 5-2). The variation in concentration between the different tested values of the advective velocity is however rather small. For the investigated range in V_Z, given an inflow of 1 Bq/year and for $T_k=1\times10^{-5}$ year, the total retained mass in the surface and deep sediment (totally 1 m depth) varies practically insignificantly around 1.8 Bq. The time for reaching these equilibrium levels varies, however, in the order from ~10 years to ~100 years. Though, the variation in equilibrium level and the time for reaching equilibrium might become larger for another parameter set-up e.g. for other sorption characteristics.

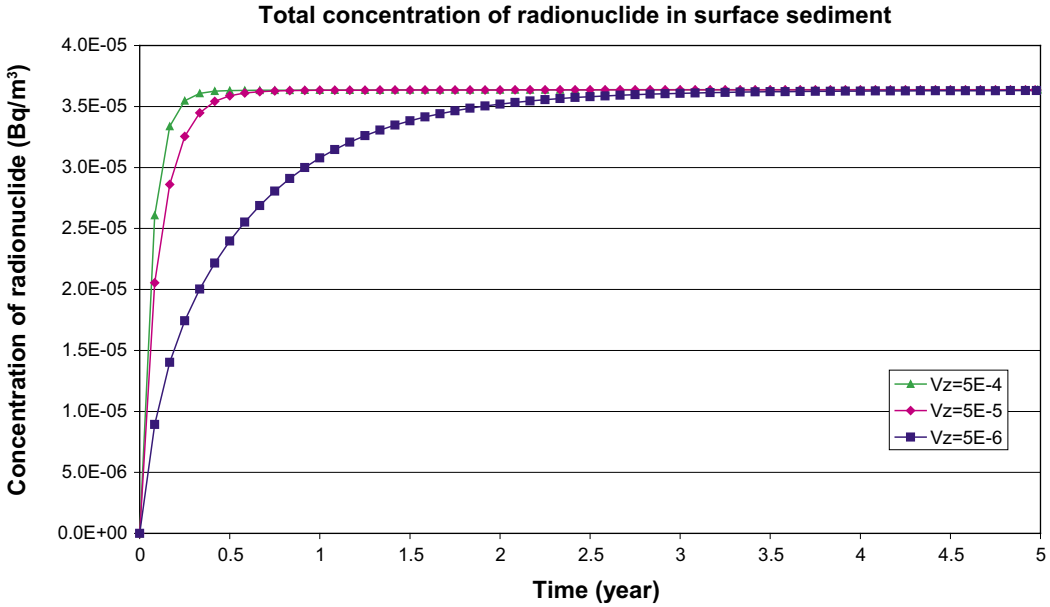

Figure 5-13. *Total concentration of radionuclides in surface sediment for different values of the advective velocity in the sediment for the distribution coefficients K_d=0.3 m³/kg, K_B=0.03 m³/kg and the half-time to reach sorption equilibrium T_k=0.001 year ($V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).*

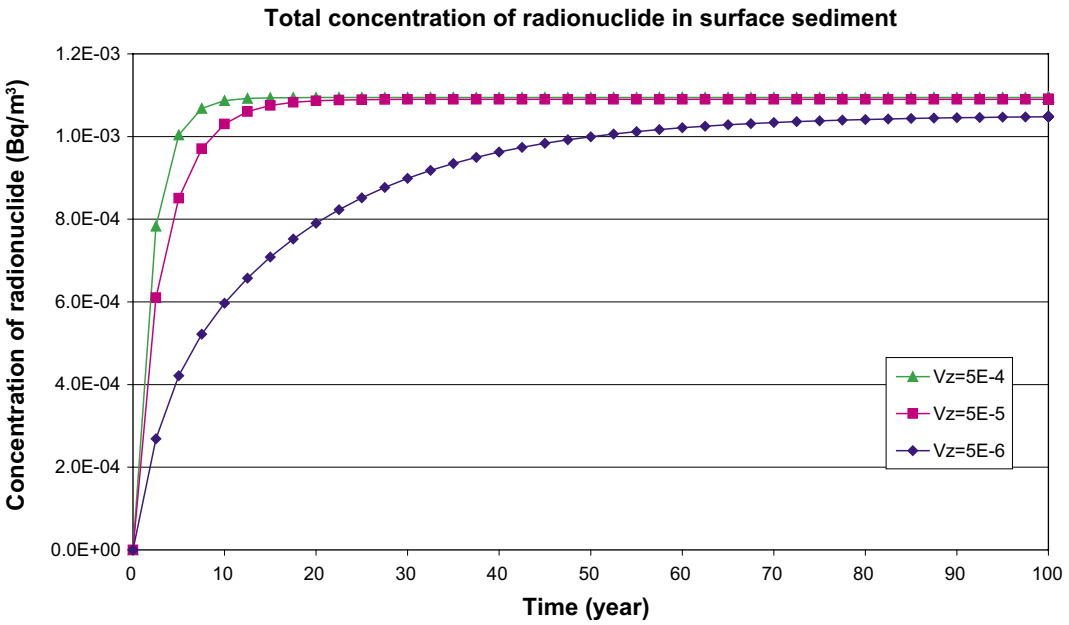

Figure 5-14. *Total concentration of radionuclides in surface sediment for different values of the advective velocity in the sediment for the distribution coefficients K_d=10 m³/kg, K_B=1 m³/kg and the half-time to reach sorption equilibrium T_k=0.001 year ($V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).*

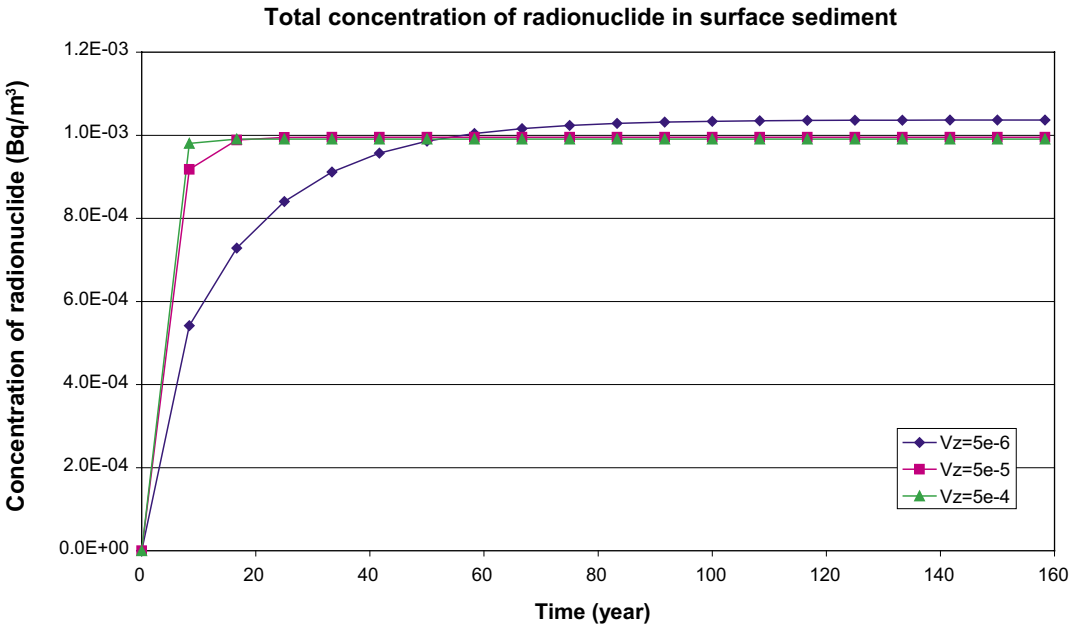

Figure 5-15. *Total concentration of radionuclicdes in surface sediment for different values of the advective velocity in the sediment for the distribution coefficients K_d=10 m³/kg, K_B=1 m³/kg and the half-time to reach sorption equilibrium, T_k=1×10⁻⁵ year ($V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).*

Table 5-2. Equilibrium concentration in surface sediment as a function on V_z for different pronounced effect of sorption kinetics, T_k (K_d=10 m³/kg, K_B=1 m³/kg, other parameter values as in Table 5-1).

V_z (m/s)	Equilibrium concentration in surface sediment (T_k=1×10⁻³ year) (Bq/m³)	Equilibrium concentration in surface sediment (T_k=1×10⁻⁵ year) (Bq/m³)
5.00×10⁻⁶	1.05×10⁻³	1.04×10⁻³
5.00×10⁻⁵	1.09×10⁻³	9.95×10⁻⁴
5.00×10⁻⁴	1.09×10⁻³	9.91×10⁻⁴

Table 5-3. Comparison of order of magnitude of the exchange flux at equilibrium between the stream water and sediment for different magnitude of the advective velocity in the sediment. (Simulated case given in Figure 5-15).

V_z (m/s)	Fluxes into the sediment			Fluxes from the sediment		
	Advective flux (Bq/s)	Sedimentation flux (Bq/s)	Diffusive flux (Bq/s)	Advective flux (Bq/s)	Resuspension flux (Bq/s)	Diffusive flux (Bq/s)
5.00×10⁻⁶	3.8×10⁻⁹	6.3×10⁻¹⁰	2.4×10⁻¹²	4.0×10⁻⁹	4.0×10⁻¹⁰	2.5×10⁻¹²
5.00×10⁻⁵	3.8×10⁻⁸	6.3×10⁻¹⁰	2.4×10⁻¹²	3.8×10⁻⁸	3.9×10⁻¹⁰	2.4×10⁻¹²
5.00×10⁻⁴	3.8×10⁻⁷	6.3×10⁻¹⁰	2.4×10⁻¹²	3.8×10⁻⁷	3.8×10⁻¹⁰	2.4×10⁻¹²

In Table 5-3, the equilibrium values of the flux of radionuclides into and out of the sediment for the case given in Figure 5-15 are listed. Here it is possible to see that the exchange flux with the sediment due to advection is larger than due to sedimentation and diffusion for the tested magnitudes of the advective velocity. For a smaller advective velocity, the difference in exchange fluxes will however decrease.

Variation of distribution coefficients

The distribution coefficient in the stream water and sediment, defining the sorption properties of the radionuclide, influences the retention in the sediment as already partly have been shown. In Figure 5-16 to Figure 5-18, the effect of a varying distribution coefficient is visualised by showing the resulting concentrations in stream water and sediment. From Figure 5-16 and Figure 5-17 it is clear that for a higher distribution coefficient, corresponding to a more reactive radionuclide, the retention in the sediment will be more pronounced and consequently higher concentrations in the sediment will appear. An increase in K_d from 1 to 100 m³/kg, when K_B=0.1×K_d, provides a concentration in the sediment that is approximately 80 times higher. As an example, Sr, Cl, and Zr are listed as elements with a K_d-value of 1 m³/kg (best estimate) whereas Th, Pa and Pu, are listed as elements with a Kd-value of 100 m³/kg (best estimate) /Bergström et al. 1999/.

It is also possible to conclude that the time for reaching equilibrium conditions (applying the same value on T_k) are longer for an element with higher particle affinity. This is also visible if instead the concentration in the stream water is considered (Figure 5-18) where after 5 years, equilibrium conditions prevails if K_d=1 m³/kg whereas only 81% of the equilibrium concentration is reached if K_d=100 m³/kg (Table 5-4).

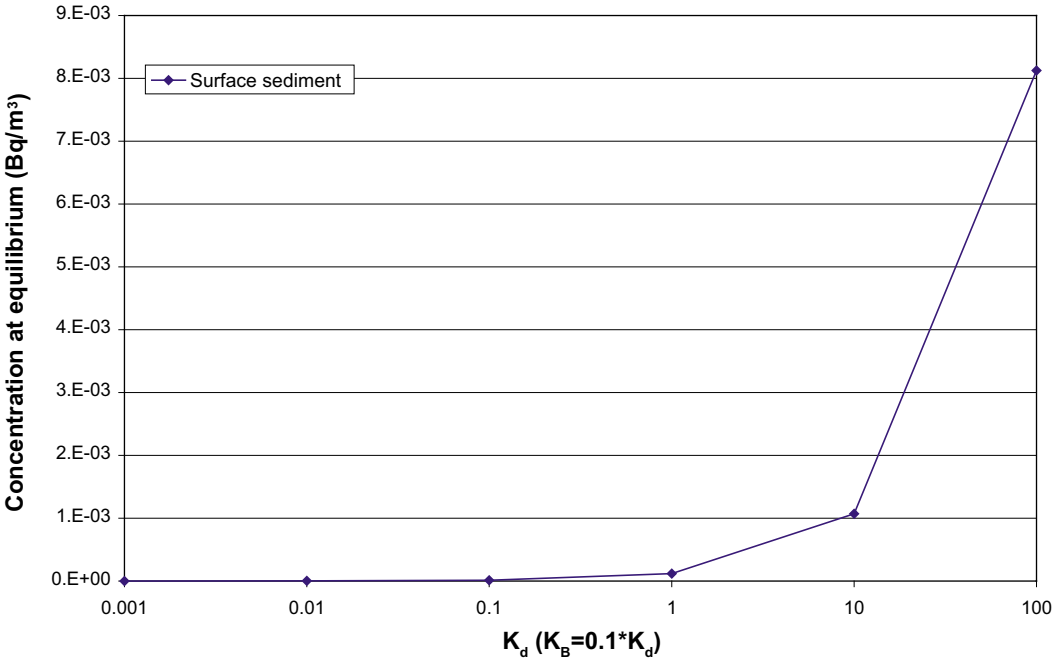

Figure 5-16. *Equilibrium concentration of radionuclides in surface sediment as a function of the distribution coefficient (V_Z=1×10^{-5} m/s, T_k=0.001 year, $V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).*

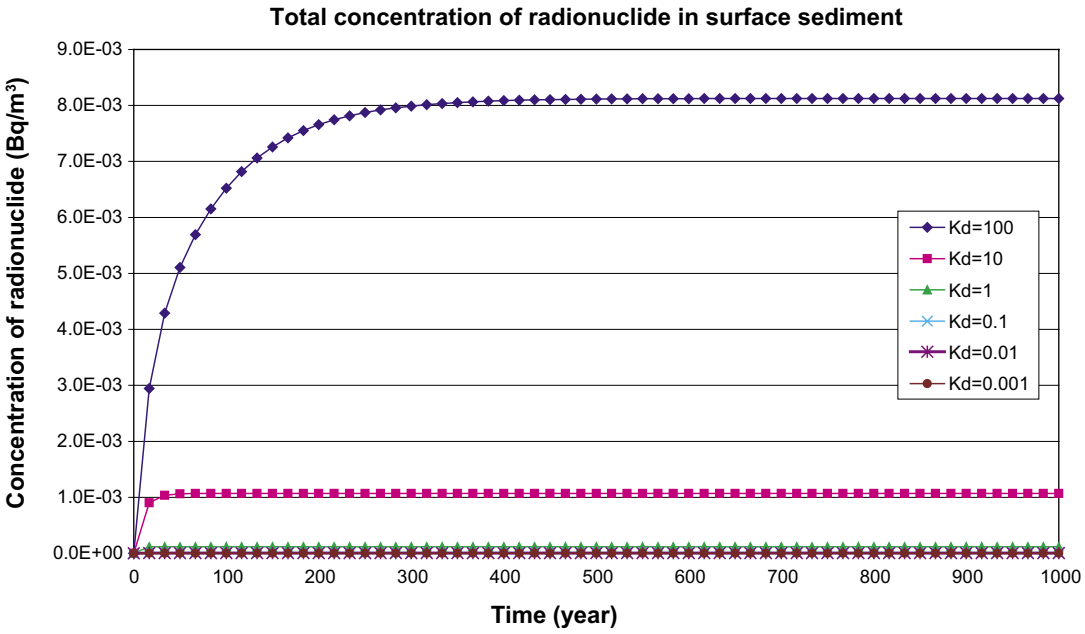

Figure 5-17. *Total concentration of radionuclides in surface sediment as a function of the distribution coefficient (V_Z=1×10^{-5} m/s, T_k=0.001 year, $V_{partsed}$=400 m/year, K_B=0.1×K_d, BCF=200 (Bq/kg fw)/(Bq/l)).*

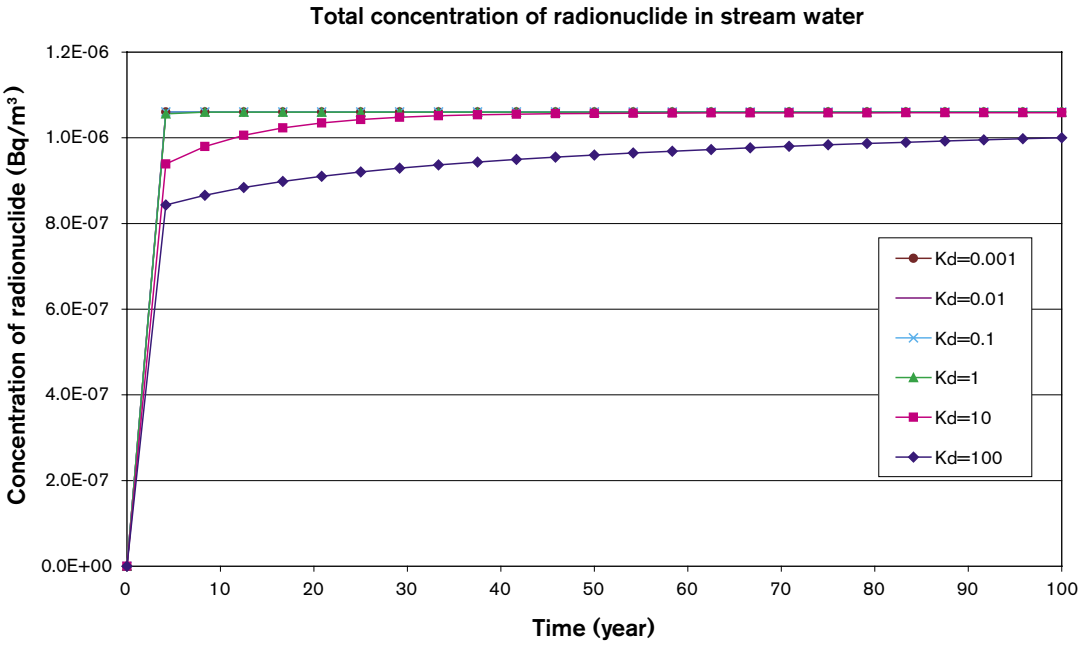

Figure 5-18. *Total concentration of radionuclides in stream water as a function of the distribution coefficient (V_Z=1×10^{-5} m/s, T_k=0.001 year, $V_{partsed}$=400 m/year, K_B=0.1×K_d, BCF=200 (Bq/kg fw)/(Bq/l)).*

Table 5-4. Proportion of the equilibrium concentration of radionuclides in the stream water reached after 5 years for different values on the distribution coefficient (V_Z=1×10^{-5} m/s, T_k=0.001 year, $V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l), K_B=0.1×K_d).

K_d (m³/kg)	Proportion of equilibrium concentration reached after 5 years
0.001	100%
0.01	100%
0.1	100%
1	100%
10	90%
100	81%

The predicted total amount of radionuclides in the sediment at equilibrium for elements with different particle affinity (given an inflow of 1 Bq/year to the stream water) is listed in Table 5-5. For the given prerequisite in the calculation example, the total amount of a strongly sorbing element (K_d=K_B=100 m³/kg) could be as high as ~31 Bq whereas for the case of a more conservative element (K_d/K_B=0.001/0.0001 m³/kg) the total retained mass is only ~0.002 Bq.

Table 5-5. Predicted total amount of radionuclides in sediment (surface + deep) at equilibrium for different distribution coefficients (V_Z=1×10^{-5} m/s, T_k=0.001 year, $V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).

K_d/K_B (m^3/kg)	Total amount of radionuclides in sediment at equilibrium (Bq)
0.001/0.0001	0.002
0.01/0.001	0.004
0.1/0.01	0.02
1/0.1	0.20
10/1	1.8
100/10	14
0.001/0.001	0.004
100/100	31

Variation of sorption kinetics

The kinetic behaviour of the sorption onto particulate matter in the stream water, expressed by the variable T_k, can also influence the partitioning between the different compartments. The magnitude of this coefficient should be considered in relation to the magnitude of the residence time in the stream water. If the half-time to reach sorption equilibrium is long in comparison to the residence time in the stream water, the effect of sorption will be lower than for a rapid sorption mechanism.

In Figure 5-19, the fraction of adsorbed and dissolved radionuclides in the stream water is given for different reaction times T_k when K_d=10 m^3/kg and K_B=1 m^3/kg. For a reaction time of 1×10^{-5} year, the adsorbed concentration at equilibrium is approximately 20% of the dissolved concentration, while for a reaction time of 1×10^{-2} year the adsorbed concentration is only ~2% of the dissolved concentration. A further increase of T_k will however not have any considerably influence of the partition in this case. Also for values on T_k smaller than 1×10^{-4} year, the change in the partition is rather small.

Consequently, for a slower sorption mechanism (large T_k), the dissolved concentration in the stream water becomes higher than for a rapid sorption mechanism which in its turn also will influence the relative importance of the dissolved and particulate exchange with the sediment.

The effect of a varying pronounced effect of sorption kinetics, with other variables constant during the simulations, can also be seen as a slightly higher concentration in the surface sediment at equilibrium for a slower sorption mechanism, although the difference is small (Figure 5-20). As was concluded above, for values larger than ~1×10^{-2} year (for T_k), any further increase will practically not influence the equilibrium concentration at all. The same thing applies for values smaller than ~1×10^{-4} year, where any further decrease in T_k will not provide any further significant decrease of the equilibrium concentration in the sediment.

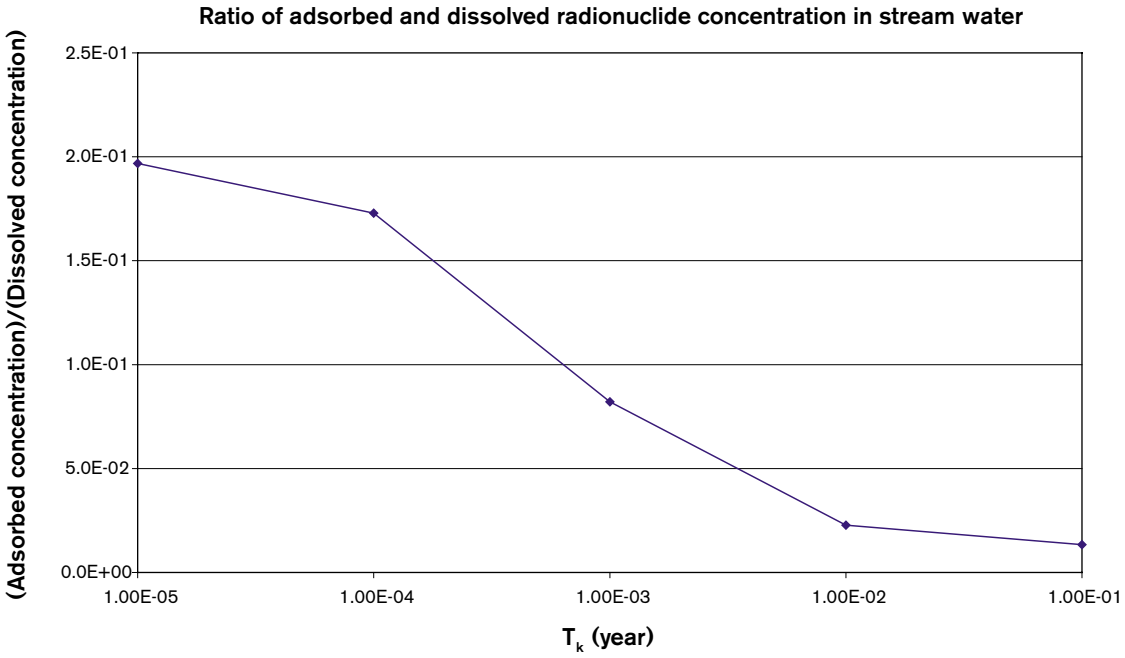

Figure 5-19. *The ratio between adsorbed and dissolved radionuclide concentration in the stream water at equilibrium conditions as a function of half-time to reach sorption equilibrium (V_Z=1×10^{-5} m/s, K_d=10 m^3/kg, K_B=1 m^3/kg, $V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).*

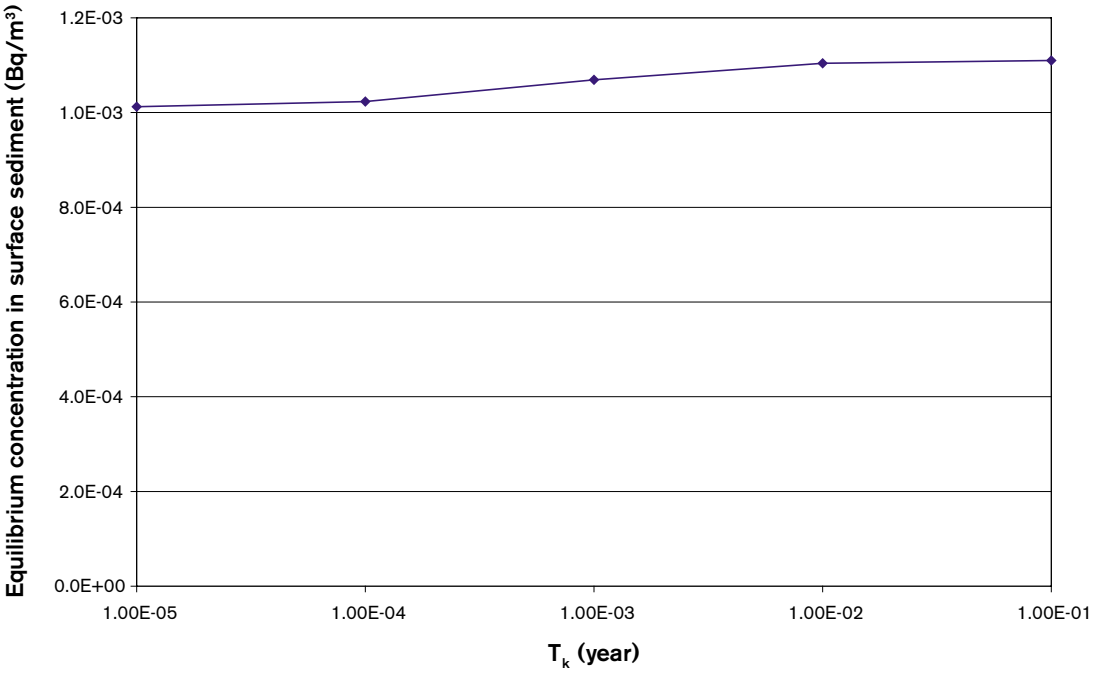

Figure 5-20. *Equilibrium concentration of radionuclides in surface sediment as a function of time for reaching sorption equilibrium (V_Z=1×10^{-5} m/s, K_d=10 m^3/kg, K_B=1 m^3/kg, $V_{partsed}$=400 m/year, BCF=200 (Bq/kg fw)/(Bq/l)).*

Variation of sedimentation velocity

The results of the sensitivity analyses of the sedimentation velocity, $V_{partsed}$, reveals that the equilibrium concentration in the sediment will become higher for a higher sedimentation velocity in the case when K_d=10 m³/kg, K_B=1 m³/kg and T_k=1×10⁻⁵ year (Figure 5-21). The effect of an increase in the sedimentation velocity on the equilibrium concentration in the sediment is largest for the lower of the two tested advective velocities in the sediment (V_Z=5×10⁻⁶ m/s and V_Z=5×10⁻⁵ m/s). This is because the exchange due to sedimentation will be of a higher relative importance when the advective velocity decreases. However, for a slow sorbing substance (T_k=1×10⁻³ year), the variation in sedimentation velocity is relative unimportant for the equilibrium concentration, for the higher value of the two tested advective velocities (Figure 5-21). The effect is rather small also for the lower value of the advective velocity, although an increase in sedimentation velocity will provide a small decrease in the surface sediment concentration. This is due to the resulting small fraction in adsorbed phase in the stream water when the time-scale of sorption is considerable in comparison to the residence time in the system. The dominating exchange will then be due to advection.

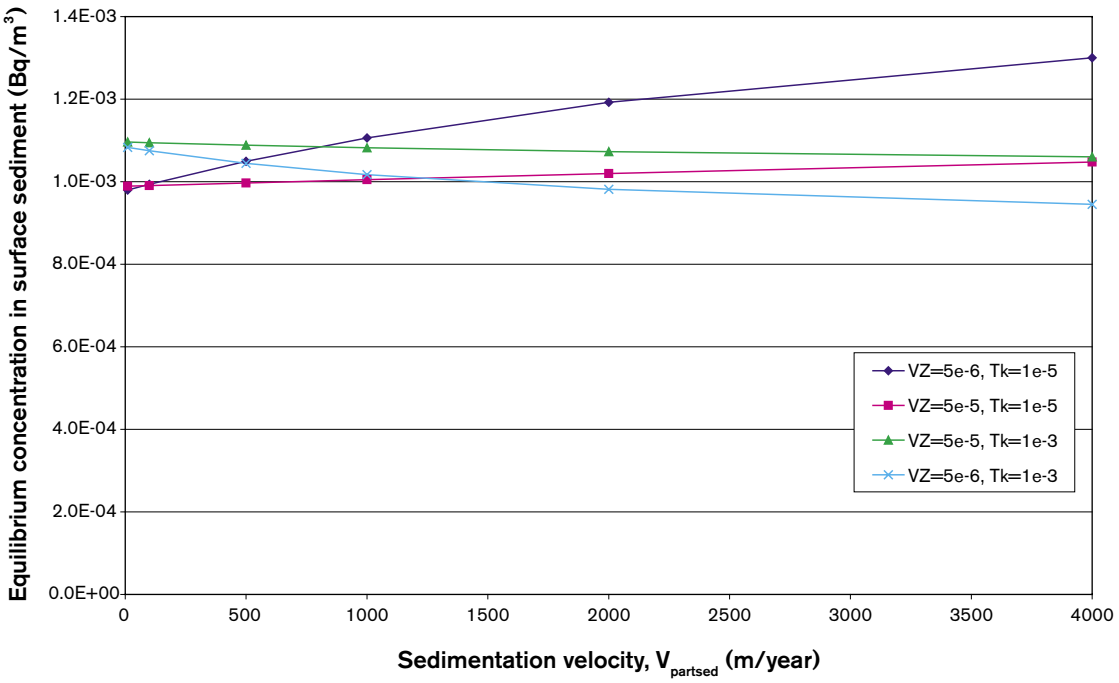

Figure 5-21. *Effect of sedimentation velocity on the equilibrium radionuclide concentration in the surface sediment when T_k=1×10⁻⁵ year and T_k=1×10⁻³ year (K_d=10 m³/kg, K_B=1 m³/kg, BCF=200 (Bq/kg fw)/(Bq/l)).*

Comparison of retention in sediment and biota

In Table 5-6 the amount of radionuclide in the sediment and in the biomass at equilibrium is compared for two different sets of distribution coefficients (K_d=10 m^3/kg, K_B=1 m^3/kg and K_d=0.3 m^3/kg, K_B=0.03 m^3/kg) and a bioconcentration factor equal to 200 (Bq/kg fw)/(Bq/l). It is clear that for a more reactive radionuclide, the total amount in the sediment will become higher. However, even for the lower K_d-value, the amount of radionuclide in the biomass constitutes only less than 4% of the mass in the sediment, yielding that the retention in the sediment will be larger than due to uptake in biota. However, the uptake in biota can become of a higher relative importance if a less reactive substance with retained order of magnitude on the biomass concentration factor is considered. The combination of a K_d value equal to 0.3 m^3/kg and a BCF equal to 200 (Bq/kg fw)/(Bq/l) are values applicable for iodine /Bergström et al. 1999, Brown et al. 2003a/.

Table 5-6. Comparison of amount of radionuclide in sediment and in biomass at equilibrium for different values on the advective velocity in the sediment (BCF=200 (Bq/kg fw)/(Bq/l), $V_{partsed}$=400 m/year, T_k=0.001 year).

V_z (m/s)	Amount in sediment (Bq)	Amount in biomass (Bq)	Amount in sediment (Bq)	Amount in biomass (Bq)
	K_d=10 m^3/kg, K_B=1 m^3/kg		K_d=0.3 m^3/kg, K_B=0.03 m^3/kg	
5×10^{-6}	1.76	0.0020	0.06	0.0021
5×10^{-5}	1.84	0.0020	0.06	0.0021
5×10^{-4}	1.84	0.0020	0.06	0.0021

Variation of bed-load transport

In Figure 5-22, the predicted transport of sediment particles on the bed surface using the equations by /van Rijn 1984/ is visualized for the flow conditions given in Figure 4-1. This is not to be mixed up with the transport of radionuclides, but naturally, as the sediment bed-load transport increases, also will the radionuclide transport. The sediment transport will increase with the discharge, as the erosion of particles will become more pronounced for larger flows. For a variation of the flow as large as given by the specific run-off at Vattholma over a year, the difference in flux of particulate matter differs several order of magnitudes. The bed-load calculations therefore tend to be rather sensitive for the determination of the flow conditions in the stream channel. Further, the equations by van Rijn are given with limits of applicability where e.g. the water depth should be given within the range 1–20 m and the velocity within 0.5–2.5 m/s. For the theoretical stream channel used in this study, the water depth and the velocity will fall slightly outside this range and the model equations are then not strictly valid. However, here they are judged to be useful for estimating an order of magnitude of the sediment transport.

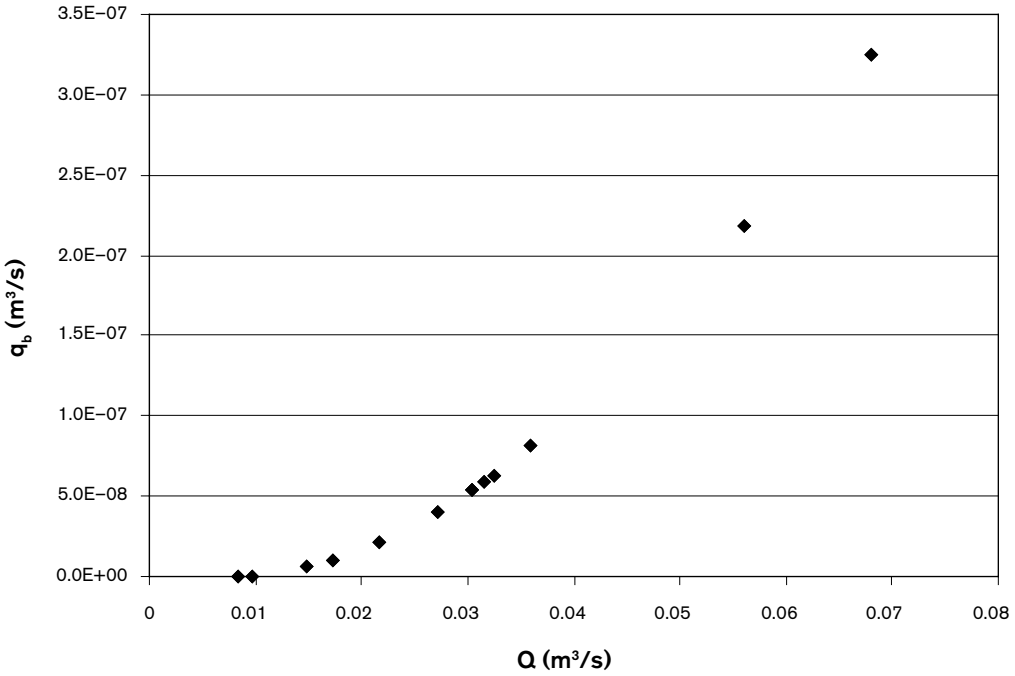

Figure 5-22. *Sediment transport (not radionuclides) predicted using the relationship by /van Rijn 1984/ as a function of flow condition (used parameter values are defined in Table 5-1).*

A further refinement of the part of the model describing the sediment transport might therefore be considered in a future model formulation after the inclusion of more site-specific information of the flow conditions in the actual stream in consideration. Until then, the equations by van Rijn are used as an approximation.

Outflow from stream water

The uptake in the sediment will initially cause a retardation of the solute transport. The sediment capacity is however limited and for the case of a constant inflow of dissolved radionuclides to the stream water, the sediment will after some time be saturated. When this saturation is reached, the outflow from the stream-reach is determined by the inflow, here 1 Bq/year. The time for reaching this state varies however with the exchange parameters. The effect on the time for reaching this saturation due to a change in sedimentation velocity is shown in Figure 5-23. The time for reaching 90% of the equilibrium value is reached within 1 year for the case of a sedimentation velocity of 10 m/year. The corresponding value for a sedimentation velocity of 4,000 m/year is ~6 years. The variation in time for reaching equilibrium between the tested cases is, as can be seen in the figure, rather small. This is because the dominating exchange is due to advection and not due to sedimentation. Therefore, variations in other parameters might be of higher importance for the time to saturation.

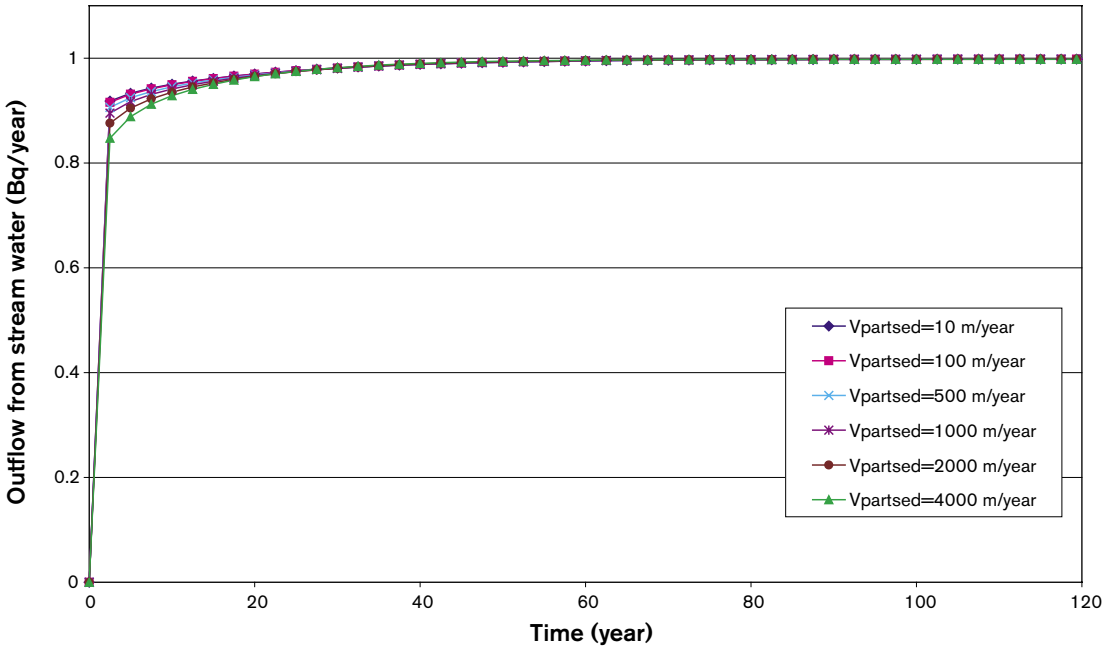

Figure 5-23. *Outflow of radionuclides with the streaming water for different sedimentation velocities (K_d=10 m^3/kg, K_B=1 m^3/kg, T_k=1×10^{-5} year, V_Z=5×10^{-6} m/s).*

6 Further investigations

This report includes a first sensitivity analysis of the proposed stream transport model. The sensitivity analyses have been performed for a hypothetical stream, as site-specific information from the watercourses in Forsmark and Simpevarp/Laxemar was quite sparse at the time for the model development. The order of magnitude of the parameters during the sensitivity analyses has therefore been based on available literature information and partly on generalizations from other watercourses. For a few parameters though, the magnitude has been based on large-scale information from the current site i.e. information on runoff, slope and catchment area.

To be able to fulfill a more complete sensitivity analysis, more site-specific data would be useful, in order to reduce the uncertainty of some of the parameters. Also, the expressions for some of the transfer coefficients could need a further consideration and possible refinements. Primarily the theories used for the estimation of the advective velocity in the sediment and the formulation of the bed-load transport could be refined, which can be important for the predicted accumulated amount in the sediment.

Data that was not available at the time for the sensitivity analysis, but required for an improved evaluation of the hydraulic part of the model, is site-specific information of cross-sectional geometry along the stream channel. The specific run-off from available measuring stations (exemplified in Figure 4-1) and approximation of the channel slope by large-scale slopes in the area is assumed to be sufficient for the calculations. Also, in the literature, tabulated values of the Manning roughness coefficient for natural channels are assumed to be applicable.

For the solute transport model, more site-specific information is needed. During the sensitivity analyses, the predicted results were found to vary with the advective velocity arising due to pressure variations on the bed surface. The effect was differently pronounced depending on assumed sorption characteristics and sorption kinetics. The effect was mainly seen on the time for reaching equilibrium, where the lower advective velocity resulted in a longer time for reaching equilibrium. In some cases, also the resulting equilibrium concentration in the sediment differed between the different advective velocities, however, for the investigated range of the parameters, the effect on the equilibrium concentration was rather small. The present estimation of the variable range for the advective velocity is based on theories by /Wörman et al. 2002ab/, and by generalizing their result to the present watercourse. In the generalizations, assumptions have been made, for example, of the hydraulic conductivity in the sediment. This is a parameter that easily is measured or estimated in field and where inclusion of data will provide a more certain estimate of the advective velocity and thereby the effect of the advective exchange. However, further investigations to find out if refinements of the model equations or the generalizations are needed, should also be done. Possibly, also field experiments with injected tracers can give valuable information on the magnitude of the exchange and the approximate extension of the sediments influenced by the interaction with the stream water.

With site specific information on sediment characteristics such as density, porosity and grain-size distribution, a more certain estimation of the bed-load transport can be made. However, also the chosen method for the bed-load calculations might need a revision when more site-specific information become available. From the sensitivity analysis of the transport of particulate material, it was found that the predicted sediment transport

was quite sensitive to variations in flow condition and consequently this will then also be important for the bed-load transport of radionuclides attached to the sediment particles. Also, a possibility to apply the equations for an order of magnitude-estimation outside the ranges for which the equations were derived should be considered further. This if the flow conditions in the streams considered fall outside the ranges, as was indicated in the sensitivity analyses. A possible way to verify this is also to perform measurements of the bed-load transport in field, even though finding a field method in which the results will not be disturbed by the measuring equipment can be difficult.

Data on the concentration of particulate matter in the stream, preferably measured during a year cycle, can be used to estimate the particulate concentration and the applicability of the approximation of a zero net-deposition over the year. This is an assumption that can be important for the predicted accumulated amount in the sediment.

Even though the parameters considered most sensitive for the results have been varied in the performed sensitivity analyses, a complete sensitivity analysis after the application of some site-specific data could include variations also in other model parameters. This should be done in order to get an estimation of the total model uncertainty. Also, simulations should be performed for different radionuclides, where values of the radionuclide specific parameters should be used throughout the whole model.

After application of more site-specific information, the resulting effect of the application of a finer computational grid in the longitudinal direction should be investigated. This is however dependent on the required level of details.

After the performance of the proposed complementary investigations, the resulting concentration in the different parts of the stream ecosystem could be used for calculating a predicted dose to humans, following a possible discharge of radionuclides from a deep repository.

7 Conclusions

A new model concept for radionuclide transport in streams, to be used for predicting the transport following a possible radionuclide discharge from a deep repository in Sweden, has been proposed and presented in this report. The main difference from the previous model that until now have been used by SKB in the biosphere model is the inclusion of retention processes along the stream, mainly by uptake in sediment and biota. The exchange processes considered are exchange with the sediment due to diffusion, advective transport and sedimentation/resuspension of particulate material. Also uptake in biota is considered.

A first sensitivity analysis of the proposed model has been performed where parameters of both radionuclide and site-specific characters have been varied. For the parameters for which no site-specific information was available, values from the literature or generic information from other stream systems have been used.

It was concluded that for order of magnitude predictions of the amount or concentration of radionuclides in the different parts of the stream ecosystem, application of a yearly mean value of the flow was sufficient. The use of monthly mean values of the water flow or yearly mean values of the rate coefficients in the compartment model did not provide significantly different results compared to values varying on a monthly basis. If more detailed information is required, the choice of averaging method might, however, become more important.

The sensitivity analyses showed that for a constant discharge of radionuclides (1 Bq/year) to the stream water, an increase in radionuclide amount in the sediment was obtained until a level of saturation in the sediment was reached. When the sediment is saturated, the outflow from the stream system is directly given by the inflow of radionuclides and the prevailing flow conditions. However, the time until saturation and the amount accumulated at equilibrium was found to vary several orders of magnitude depending on the exchange parameters as well as sorption characteristics. The time for reaching equilibrium varied from the order of a few hours to around one hundred years.

The value of the advective velocity in the sediment was found to influence, mainly the time evolution of the retention in the sediment. However, also the resulting equilibrium concentration in the sediment differed for some of the tested values on the advective velocity, although not considerably in the simulated cases. The magnitude of the advective velocity was estimated by generalizing information from another stream. This needs further consideration in a future study. For a system with minor effects of sorption kinetics in the stream water, i.e. when the sorption reaction half-time (T_k) is small, an increase in the advective velocity will result in a slightly lower equilibrium concentration in the sediment. When the sorption kinetics is more pronounced, the opposite, with a slightly higher equilibrium concentration in the sediment for a larger advective velocity, is gained.

At this stage of the model development it is difficult to make firm conclusions about the total amount of radionuclide retained in the sediment. The model indicates large differences in retained amount of radionuclide in the sediment for different substances with different sorption capacity. In one simulation performed in the present study, the total mass retained in the upper first meter of a 2 km long stream differed several order of magnitudes between a more conservative (K_d/K_B=0.001/0.0001 m^3/kg) and a more reactive ($K_d=K_B$=100 m^3/kg) radionuclide. The accumulated amount was also found to vary with the advective velocity.

For the investigated ranges of the parameters, the accumulated mass in the sediments was, however, found to be less sensitive to variations in the advective transport velocity than to differences in sorption capacity.

The equilibrium concentration in the sediment was found to increase with increasing sedimentation velocities, for the case where the effect of sorption kinetics is negligible. For the case with more pronounced effects of sorption kinetics, the equilibrium concentration in the sediment was instead decreasing as the settling velocity increased. This due to the resulting small fraction in adsorbed phase in the stream water when the time-scale of sorption is considerable in comparison to the residence time in the system, leading to settling of particles with no or a low adsorbed concentration. For the lower limit of the advective velocity in the sediment ($V_Z=5\times10^{-6}$ m/s) and for the case with less pronounced effect of sorption kinetics, the equilibrium concentration in the sediment will differ by a factor of 1.3 when the sedimentation velocity is varied within the range 10–4,000 m/year.

The simulations indicated that the accumulated amount in biota was small in comparison with the accumulated amount in the sediment. The main retention along the stream is therefore assumed to be due to uptake in the sediment. In the calculated example, the amount in biota was less than 4% of the amount in the sediment. The results will of course depend on the assumption of the mass of standing crop in the stream and on which radionuclide that is considered. In the example, calculations were performed for iodine by applying a distribution coefficient and bioconcentration factor valid for that substance.

When equilibrium conditions have been reached, the outflow to the downstream "ecosystem" is given by the radionuclide discharge and flow conditions. This is due to the fact that after a certain time, the sediment will be saturated and equilibrium conditions will prevail. However, transport calculations are important in order to determine the amount accumulated in the sediment and biota, and the time for reaching equilibrium conditions in the system. A prediction of the total amount of radionuclides in sediment and biota is important because it can result in high concentrations of radionuclides, leading to possible future exposure of humans.

Further investigations should be performed in which an important part is the inclusion of site-specific information from field measurements in the stream channel under consideration. This should be done to be able to make certain statements of the transport in the stream and the accumulated amounts in different parts of the stream ecosystem. Also, a further consideration of some parts of the model formulation should be done, in particular the theories behind the advective transport in the sediment and the bed-load transport.

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Appendix A

Estimation of the order of magnitude of the advective velocity in the sediment

For the estimation of the order of magnitude of the advective velocity in the sediment, V_Z, generalisations of the theories and data by /Wörman et al. 2002ab/ is used for the approximation. In their paper a model concept coupling longitudinal solute transport with solute advection along a continuous distribution of hyporheic flow paths is presented and evaluated for Säva Brook (Sävaån) in Uppland. Their intention was to find relationships relating the residence time in the hyporheic zone to measurable parameters of the stream such as geometrical parameters and hydraulic conductivity. For more detailed information about their model see /Wörman et al. 2002ab/. By means of theories of pumping exchange over a flat bed with a harmonic pressure variation, a relationship for the residence time in the sediment due to a pumping exchange is presented in their Equation (20) as:

$$\frac{\langle T \rangle K}{h} = \frac{21}{2\pi^2} \frac{d_b \lambda}{C_1 h^2} \frac{1}{Fr^2} \tag{A1}$$

where T is the total residence time from inlet to exit of hyporheic flow path, K the hydraulic conductivity, λ wavelength of pressure variations along bed, h the hydraulic mean depth (area divided by surface width), d_b the depth of the hyporheic zone, Fr the Froude number (equal to $U/(gh)^{0.5}$), U the flow velocity in the main stream channel and C_1 is a coefficient in pumping theory.

By considering the lumped parameter $\xi = \frac{21}{2\pi^2} \frac{d_b \lambda}{C_1 h^2}$ as an unknown proportional coefficient for the Säva Brook, Wörman et al. evaluated the coefficient by means of data from a performed tracer experiment and yielded two coefficients, one for the reach in agricultural land (ξ=0.14) and one for the reach in coniferous forest (ξ=0.013). Their explanation for the deviation between the two reaches is that the lumped parameter, ξ, probably varies with stream geomorphology.

The corresponding expression for the exchange velocity is formulated as /Wörman et al. 2002ab/:

$$\frac{\langle V_Z \rangle \big|_{z=0}}{K} = C_1 \pi \frac{h}{\lambda} Fr^2 \tag{A2}$$

where V_Z is infiltration Darcy water velocity.

By combining Equation (A1) and (A2) a relation between the residence time and the exchange velocity could be found according to:

$$\langle T \rangle = \frac{21 d_b}{2\pi V_Z} \tag{A3}$$

Substituting Equation (A3) into the relationship (A1) and using the evaluated coefficient on the lumped coefficient ξ, the relationship for the exchange velocity could be formulated as:

$$V_Z = \frac{21 d_b K U^2}{2\pi h^2 g \xi} \tag{A4}$$

By using the order of magnitude of the flow conditions in the present study, with the assumption that the evaluated coefficient for the Säva Brook could be used also for the present watercourse, the order of magnitude of the advective velocity was estimated for the flow conditions in the present study. In this estimation the depth of the hyporheic zone, d_b, is assumed equal to 0.5 m, the hydraulic conductivity equal to 10^{-6}–10^{-5} m/s, the mean velocity in the stream, U, equal to 0.3 m/s, h equal to 0.15 m and ξ =0.14 or ξ =0.013. As can be seen the result is rather strongly influenced by the order of magnitude of the hydraulic conductivity. A decrease in the hydraulic conductivity by a factor of ten yields naturally a decrease also of the advective velocity by a factor of 10. Considering the range in the evaluated lumped coefficient, ξ, given by /Wörman et al. 2002ab/ for the Säva Brook and an assumed range of 10^{-6}–10^{-5} m/s for the hydraulic conductivity, an order of magnitude for the advective velocity is then in the range 7×10^{-6}–8×10^{-4} m/s. Also the effect of varying flow condition will influence the result, however, the uncertainties introduced by the estimation of the hydraulic conductivity and the fact that a generalization is performed of the results from the Säva Brook is probably larger. Also the relationship by /Wörman et al. 2002ab/ has not yet been evaluated for many streams and stream reaches and the exact value of d_b is difficult to estimate in a specific case.

To obtain an estimation of the reliability of the calculated magnitude on the advective velocity, the flow in the sediment can be compared to the flow in the stream water. The water flow in the stream channel is around 1×10^{-6} m^3/year. For an advective velocity of 7×10^{-6} m/s, a gradient of one and a 2 km long stream reach, the flow in the sediment would be around 350,000 m^3/year, i.e. the flow in the sediment will constitute ~35% of the flow in the stream channel. This seems as a reasonable assumption.

However, due to the uncertainties mentioned above, the estimation of the order of magnitude of V_z performed here should be considered highly preliminary. A further investigation of the relationships, possible in connection to field measurements is needed to get a more certain evaluation of the order of magnitude of the advective velocity.

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