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Models for dose assessments

Models adapted to the SFR-area, Sweden

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Abstract

This report presents a model system created to be used to predict dose rates to the most exposed individuals in case of a long-term release of radionuclides from the Final repository for radioactive operational waste (SFR) in Forsmark, Sweden. The system accounts for an underground point source emitting radionuclides to the biosphere, their transport and distribution in various ecosystem types, their uptake by various biota, and calculation of doses to man from a multitude of exposure pathways. Long-term aspects include the consideration of processes at geological time scales, such as land uplift and conversion of marine sediments to arable land. Model parameters are whenever possible based on local conditions and recent literature.

The system has been used for simulations based on geospheric releases varying over time of a mixture of radionuclides. Here, the models have been subjected to various test scenarios, including different radionuclide entry points and sorption properties. Radionuclides released from SFR are efficiently diluted to low concentrations in the water of the coastal model. A large fraction of the nuclides leave the Model Area quickly as a consequence of the rapid water turnover. The total amount of radionuclides in water is about the same independent of particle affinity (K_d), and at most some percents of the amounts retained in the sediments for some time. The latter is also true for the lake model when releases of radionuclides to the water is simulated.

The role of sediments as a radionuclide source seems of minor importance in lakes at least for long-term radiation doses. Modelling the sediments as a source of radionuclides most of the “low K_d radionuclides” will leave the lake whereas the “high K_d nuclides” are still present within the deeper sediments after 1 000 years. The amount of “low K_d radionuclides” present in the water and on suspended matter are about $8 \cdot 10^{-5}$ of the initial inventory in the sediments. For “high K_d nuclides” this fraction is about $9 \cdot 10^{-8}$ of the initial inventory.

When modelling the turnover of long-lived radionuclides in the agricultural land model an unexpected result showed up, indicating a benefit of dynamic modelling: The amounts of nuclides in the top soil layer was highest for radionuclides with an intermediate K_d-value, whereas the fraction of radionuclides present in the top soil layer was low for radionuclides with low as well as high K_d-values, less than 1 % of the total amount added in the simulations.

Sammanfattning

I den här rapporten presenteras ett modellsystem som har tagits fram för att uppskatta doser till de mest exponerade individerna vid ett långtidsutsläpp av radionuklider från Slutförvaret för radioaktivt driftavfall (SFR) vid Forsmark, Sverige. Systemet inkluderar en underjordisk punktkälla som släpper ut radionuklider till biosfären, nuklidernas transport och fördelning i olika ekosystemtyper, upptaget i olika biota och beräkningar av dos till människa från ett antal olika exponeringsvägar. Långtidsaspekter som landhöjning och omvandling av marina sediment till jordbruksmark har inkluderats. Modellparametrarna har, så långt det varit möjligt, baserats på lokala förhållanden och den senaste litteraturen.

Systemet har använts för simuleringar baserade på utsläpp av olika radionuklider från geosfären som varierar över tiden. I denna rapport har modellerna testats i olika scenarier, inkluderande olika utsläppskällor och sorptionsegenskaper. Radionuklider som släpps ut från SFR späds effektivt ut till en låg koncentration i vattnet i kustmodellen. En stor andel av nukliderna lämnar området fort på grund av den snabba vattenomsättningen. Den totala andelen radionuklider i vattnet är ungefär lika stor oberoende av affiniteten till partiklar (K_d), och som mest några procent av den andel som fördröjs i sedimenten för en tid. Det senare gäller även för sjömodellen när utsläpp av radionuklider till vattnet simuleras.

Sedimentens roll som radionuklidkälla verkar vara av liten betydelse i sjöar, i alla fall fall när det gäller långtidsdoser. När sedimenten modelleras som en källa för radionuklider kommer det mesta av de lågsorberande radionukliderna att lämna sjön medan högsorberande nuklider fortfarande är närvarande i djupare sediment efter 1 000 år. Mängden lågsorberande radionuklider i vattnet och på suspenderat material är ungefär $8 \cdot 10^{-5}$ av det initiala inventariet i sedimenten. För högsorberande nuklider är denna fraktion ungefär $9 \cdot 10^{-8}$ av det initiala inventariet.

När omsättningen av långlivade radionuklider modellerades i jordbruksmarksmodellen dök ett oväntat resultat upp som indikerar en fördel med dynamisk modellering. Mängden radionuklider i det översta jordlagret var störst för radionuklider med mellanstor sorptionsförmåga medan fraktionen av radionuklider som fanns i detta jordlager var lägre för såväl låg- som högsorberande radionuklider, dvs lägre än 1 % av den totala mängden som tillförts i simuleringarna.

Contents

1 Introduction

The objective of this study was to construct a model system which could be used to predict dose rates to the most exposed individuals in case of a long-term release of radionuclides from the Final repository for radioactive operational waste (SFR) in Forsmark, Sweden. SFR is a facility for disposal of low and intermediate level radioactive operational waste from the nuclear power plants in Sweden. Low level radioactive waste from industry, medicine and research is also disposed in SFR. The facility is situated in bedrock beneath the Baltic Sea, 1 km off the coast near the Forsmark nuclear power plant in northern Uppland.

SFR was built between 1983 and 1988. An assessment of the long-term performance of the facility was presented in an early safety report /SKB, 1987/. A complementary analysis was presented in a deepened safety report in 1991 /FSA, 1991/.

In the operational license for SFR it is stated that renewed safety assessments should be carried out at least every ten years. In order to meet this demand, SKB launched the project SAFE (Safety Assessment of Final Repository of Radioactive Operational Waste). The aim of the SAFE project is to perform an updated analysis and thereby estimate potential dose to the most exposed individuals.

The models used for the biospheric part of the safety assessment, i.e. the models for transport of radionuclides within the biosphere and the resulting exposure of man to radiation (dose) are presented here. A number of studies have been carried out during the past years to investigate and document the biosphere in the area surrounding the repository. Modelling of shore-level displacement by land rise, coastal water exchange and sedimentation has provided data for prediction of the evolution of the area /Brunberg and Blomqvist, 1999, 2000; Brydsten, 1999a, 1999b; Engqvist and Andrejev, 1999, 2000; Jerling et al, 2001; Kautsky et al, 1999; Kumblad, 1999, 2001/. This material has been used for developing a model system for the area. In this report, the structure and parameter values used as well as the basic performance of the model system are presented. Simulations based on geospheric release scenarios and resulting doses are presented elsewhere /Lindgren et al, 2001/.

1.1 Short description of the area

SFR is located in a coastal area near Forsmark (north of Stockholm) and the repository is situated in the bedrock 1 km off the coast beneath the water of the bay Öregrundsgrepen in the Bothnian Sea, see Figure 1-1. Öregrundsgrepen is shaped as a funnel between the mainland and the islands Gräsö and Örskär to the north-east. The western part is a large shallow water area with many emerging rocks and islands. The bay opens to the Bothnian Sea in the north, but has another more narrow connection at the south-eastern end with the Åland Sea. The largest freshwater inflows to Öregrundsgrepen are the small rivers Olandsån and Forsmarksån. The basin of the bay is formed by a 30 km long, deep channel stretches along Gräsö island (the singölinje fault) with two deeps; Storgrunnan (51 m) and one west of the Engelska grundet (59 m). In the south, the narrowing bay becomes increasingly shallow and has a depth of only 15 to 20 m outside

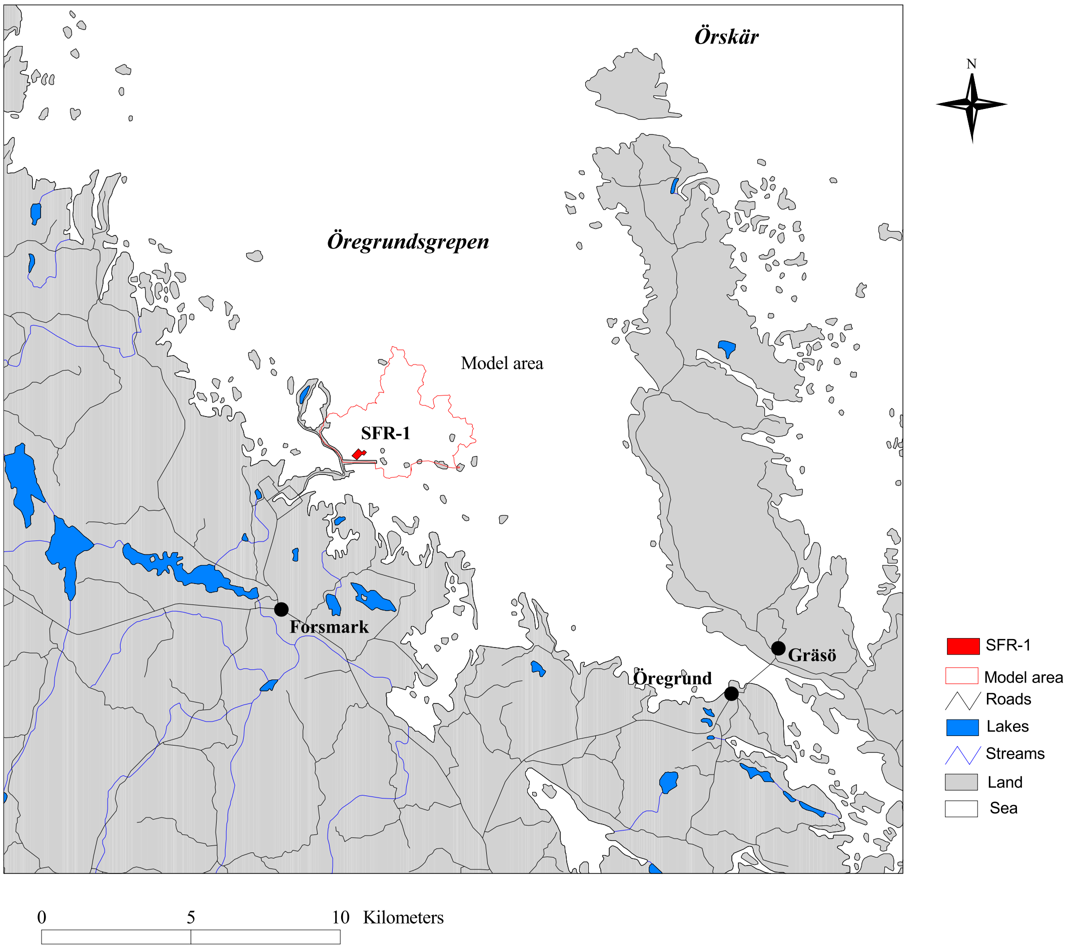

Figure 1-1. *Öregrundsgrepen with the Model Area (marked with a red line) and SFR (marked with red symbol within the Model Area). Permission: The National Land Survey of Sweden. Register number 507-96-1524.*

the town Öregrund and in the north, the channel expands and merges with the Bothnian Sea. The whole area is strongly influenced by the land rise – 0.6 cm/y /Påsse, 1997/ – which is shown on maps in /Brydsten, 1999a/, causing continuous changes of the landscape resulting in new biotopes /Wallström and Persson, 1997; Jerling et al, 2001/.

The bedrock in the area is dominated by granites and gneisses. The glacial and post-glacial soils overlaying the bedrock consist of calcium-rich till and clay deposited during glaciation, and as a consequence the continental ground and surface waters are generally well buffered and relatively nutrient rich. According to the Swedish Meteorological and Hydrological Institute (SMHI), the average (1961–1990) annual precipitation is 544 mm and the average temperature is 5.8°C. The average runoff is 200–300 mm/year /Lindborg and Schüldt, 1998/.

The major part of the terrestrial vegetation in the SFR area consists of low productive, sparse vegetation. In a few places wetlands and dry meadows occur /Jerling et al, 2001/. About 70 % of the area is forest, coniferous forest dominate but deciduous trees occur, especially close to the water. The ground vegetation is abundant. No agricultural areas occur in the near vicinity of the repository area and the amount of agricultural land is low in general (less than 10 % /Brunberg and Blomqvist, 1998/). Most lakes in the area are oligotrophic hardwater lakes.

The area is relatively sparsely populated with about 15 inhabitants per km^2 /SKB, 1997/. Large-scale farming does not occur in this part of the region.

Within 2 000 years (4 000 AD) half of the current water area in Öregrundsgrepen will be land and the water volume will be decreased with two thirds. At 7 000 AD, no brackish water will be present within the whole Öregrundsgrepen area /Brydsten, 1999a/. The area closest to the repository (called the Model Area, see Figure 1-1) will not contain any brackish water at approximately 5 000 AD and at least 20 new basins with fresh water will be formed within this area. Most of them will be shallow and will therefore be transformed to mires quickly /Brydsten, 1999a/.

2 General characteristics of the model system

The models are based on the biosphere models developed for the safety assessment SR 97 /Bergström et al, 1999/. In that study the following recipients for groundwater were included: lakes, running waters, agricultural land, peat bogs, coastal areas and wells. These models, which mostly were generic, i.e. not site-specific have been updated and adapted to describe the turnover of radionuclides within the SFR-area of today and in the future (the peat bog model is here called mire model). All models have a similar concept, that is a dynamic part for calculating the distribution of radionuclides between major physical components of the biosphere like soil, water, and sediments, the results of which are used to predict biological uptake and radiation doses along multiple pathways.

The long-term dynamics of radionuclides in ecosystems are basically controlled by a few of the slowest processes of quantitative significance. Biotic processes rarely belong to this category, since the turnover time of living tissues as well as food chains is on the order of days to years, whereas the focus of a repository model is on time scales of decades to millennia. Therefore, the models account for some important geochemical and physical processes such as sorption and radioactive decay, while processes leading to bioaccumulation and biomagnification usually are accounted for by use of steady state concentration ratios (bioconcentration factors).

The resulting transfers between the different compartments due to various processes are all described by transfer coefficients (TC) or rate constants expressed as the turnover of radionuclides per year. The turnover of radionuclides is calculated with first order differential equations which are solved numerically with ACTIVI from the computer code BIOPATH /Bergström et al, 1982, 1995/.

The concentration of radionuclides in food-stuff is then calculated assuming equilibrium between plants/animals and their environment, an assumption that is justified by the time scales in question and that is usually used in this kind of modelling, e.g. /BIOMOVS II, 1996a; Davis et al, 1993/. Uptake of radionuclides in aquatic biota is therefore calculated using bioaccumulation factors and uptake in terrestrial vegetation is simulated using root-uptake factors and translocation factors (for radionuclides added to the vegetation surface). Specific transfer factors are used when calculating the transfer of radionuclides to milk and cattle meat. The internal exposure of humans is a combination of exposure from radionuclides, taken in by food or water, and those which are inhaled. Exposure from the former ones is calculated by combining the concentrations in foodstuff, consumption rates and nuclide specific dose coefficients for conversion from radioactivity to dose (becquerel to sievert) for ingested radionuclides. By combining concentration in air with inhalation rates and nuclide specific dose coefficients for conversion from activity intake to dose for inhaled radionuclides exposure via inhalation is estimated. External exposure is calculated through consideration of exposure times, concentration of radionuclides in the air or on the ground and nuclide specific dose coefficients for conversion to dose (becquerel per unit of surface or volume to Sievert) for external exposure. The expressions used for calculation of doses are further described in Chapter 9.

In order to estimate the uncertainty of the model results due to uncertainties in parameter values the parameters used (except for the dose coefficients for ingestion, inhalation and external exposure) are assigned statistical distributions and ranges from which random values are generated by Latin-Hypercube sampling /Gardner et al, 1983/. Often, information on true type of distribution for a specific parameter is lacking because of scarcity of data. In these cases the parameters have been given triangular (for narrow ranges) or log-triangular distributions (for ranges over orders of magnitudes), as is recommended by /IAEA, 1995/. Logtriangular distributions are also used for parameters with askewed intervals. The results from uncertainty analyses are less sensitive to distribution type than to the used ranges, which justifies this approach. Some parameters have been correlated to each other in order not to get unrealistic situations. Human consumption rates of fish and meat have been negatively correlated since a high consumption of one of these food stuff implies a lower consumption of the other and vice versa. Positive correlations have also been used. The correlations used are presented for each model in the following chapters.

The number of realisations used is an optimisation between accuracy and time. When the models were used in the SAFE-project 1 000 realisations were run. A comparison between 1000 and 10 000 realisations of results for the coastal model showed that the difference in aritmethic mean value was at most 5 %. For most nuclides the difference was less than 1 %.

2.1 Changes to previous model versions

One major change in the calculations compared to previous versions /Bergström et al, 1999/ is that time dependent source terms can be used instead of the unit releases considered in the earlier study. The coastal model, which is described in detail in Chapter 3, has also been altered to better reflect the conditions in the area. The way of describing sedimentation and resuspension processes, which is also ture for the lake model, has been altered (see Chapter 4). A revision of general data has also been done.

The coastal model has been constructed to reflect the conditions at the site today. The ecosystems within the area changes over time among other things due to land uplift. A prediction of the evaluation of the area has been described elsewhere /Kautsky, 2001/.

2.2 Scenarios

For the modelling of the turnover of radionuclides in the area in the future the following sequence of models has been used: Coast 1 (present conditions, about 2 000 AD–4 000 AD), Coast 2 (decreased water volumes and water turnover, 4 000 AD–5 000 AD), Lake (5 000 AD–8 000 AD) and Agricultural land (8 000 AD–12 000 AD). A possible scenario is that radionuclides are present within the deep sediments of a lake or within the deeper layers of a soil or a mire as a consequence of earlier sorption from surface water (e.g. at the coastal stage) or of contact with contaminated groundwater. The radionuclides may then migrate upwards within the sediments, soil or peat and function as a secondary source. The importance of this process for result from the lake model is discussed in Chapter 10. As a result of this discussion sediments as secondary source of radionuclides has not been modelled in the SAFE-study. The nuclides which accumulated in sediments during the coastal and lake stages were instead considered as a secondary source in the agricultural land stage.

The well model is used for calculating doses to humans based on a scenario assuming that a well had been drilled close to the repository. The structure of the well model is the same as in /Bergström et al, 1999/ but different parameter values have been used, see Chapter 7. This is also true for the mire model (earlier called peat bog model).

3 Coastal model

For calculations of doses to humans from radionuclides potentially escaping from the repository SFR in a near future (about 1 000 years), a model of the area today has been constructed. The model is divided into three sub-areas: the "Model Area" (few km^2 scale) according to earlier studies, e.g. /Kumblad, 1999/, the whole bay of Öregrundsgrepen (hundreds of km^2) (here simply called "Grepen") and the Baltic Sea (thousands of km^2). For each of these areas, compartments for water, suspended matter, the surficial sediment layer (depth of 2 cm) where biological activity occurs (here called upper sediment) and the deeper sediment (here called sediment) are set up, see Figure 3-1. Water and suspended matter leave the Baltic Sea and therefore a sink compartment called "Oceans" is also added.

In the former study /Bergström et al, 1999/ there was no separate compartment for the suspended matter in the coastal model, instead this fraction was included in the water compartment. In this study these two phases are separated in order to give a more realistic picture of the coastal system and the sedimentation process. The upper centimetres of the sediments differs in many ways, e.g. water content, biological activity and stirring, from the lower layers and therefore two sediment compartments are used.

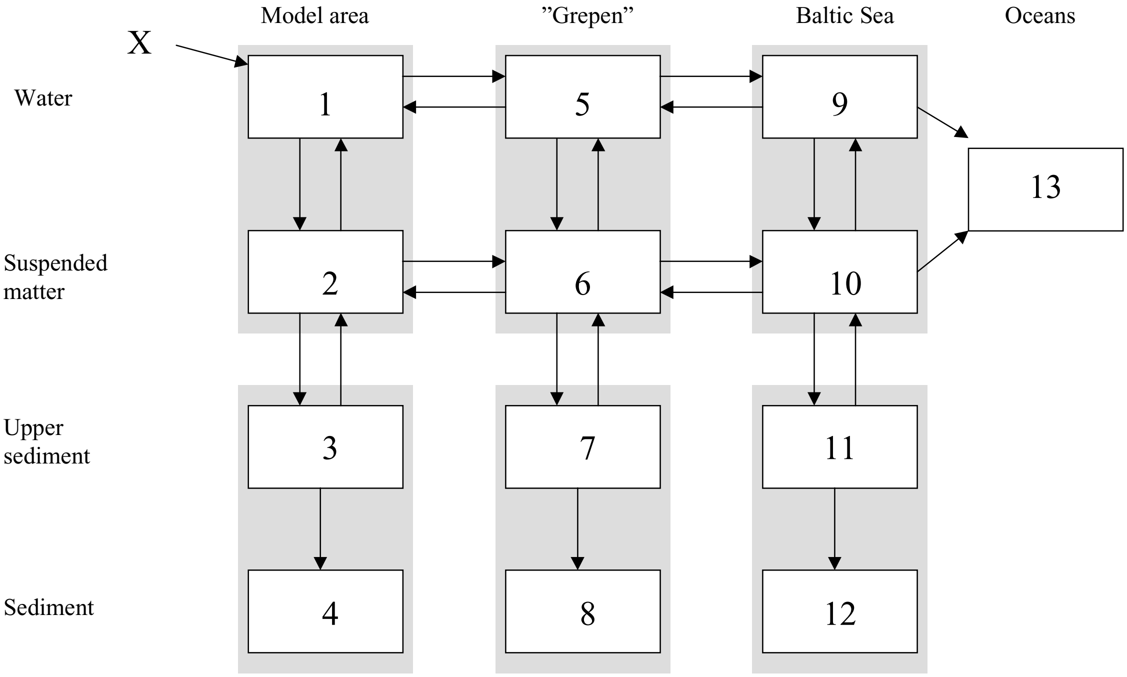

Figure 3-1. *Structure of the coastal model used in this study. Three levels of scale are considered: the Model Area (small, close and dynamic), Grepen (intermediate) and the Baltic Sea (large, remote and less dynamic). Each of these is represented by compartments for water, suspended matter, and two compartments for sediment; the top few centimetres and the deeper layers. A sink compartment (Oceans) is also added. Transfers of radionuclides within the system are marked with arrows. The cross marks the principal source of radionuclides.*

3.1 Turnover of water and suspended matter

The rate constant describing the transfer of water, and thereby radionuclides in solution, between the different water components of the system is obtained from the water retention time within each part. The radionuclides in the water or on suspended matter are assumed to move at the same rate as the carrying medium. The outflow of water (from the Model Area to Grepen, from Grepen to the Baltic Sea and from the Baltic Sea to the Oceans) is described as the fraction of water leaving a given region per unit of time, which is equivalent to the inverse of the water retention time:

$$TC = \frac{1}{RETTIME_X}$$

where

$RETTIME_x$ = Water retention time in the part from which water is transferred [year] (x = Model Area, Grepen or Baltic Sea)

The water inflow to the Model Area from Grepen is related to the volume ratio of the two areas as follows:

$$TC = \frac{A_M \cdot D_M}{A_G \cdot D_G} \cdot \frac{1}{RETTIME_M}$$

where

A_M = Area of Model Area [m^2]
A_G = Area of Grepen [m^2]
D_M = Mean water depth in the Model Area [m]
D_G = Mean water depth in Grepen [m]
$RETTIME_M$ = Water retention time in the Model Area [years]

The water inflow to Grepen from the Baltic Sea is obtained from the same expression but the values for the Model Area are replaced by those for Grepen and the values for Grepen by those for the Baltic:

$$TC = \frac{A_G \cdot D_G}{A_B \cdot D_B} \cdot \frac{1}{RETTIME_G}$$

where

A_B = Area of the Baltic Sea [m^2]
D_B = Mean water depth in the Baltic Sea [m]
$RETTIME_G$ = Water retention time in Grepen [years]

The suspended matter follows the water and therefore the same transfer coefficients are used. Particulate matter as a whole may have a longer turnover time because of settling and resuspension, which is accounted for by the exchange of material between suspended and surficial sediments.

3.2 Interaction of radionuclides between water and particles

Radionuclides are assumed to be released in soluble form into the water in the Model Area. The sorption-desorption transfer of radionuclides between the water and the suspended matter (in the Model Area as well as in Grepen and the Baltic Sea) is assumed to reach an equilibrium, but not instantaneously. Therefore, the transfer from water to suspended matter is described by a rate of sorption that is proportional to the particle concentration and the nuclide sorption affinity, described by a distribution coefficient. The process is time-dependent and therefore a parameter for the half-time to reach sorption equilibrium (here called T_k) has been used:

$$\mathrm{TC} = \frac{\ell n2}{T_k} \cdot \mathrm{Susp}_X \cdot K_d$$

and a simple rate of desorption from suspended matter to water:

$$\mathrm{TC} = \frac{\ell n2}{T_k}$$

where

T_k = Half-time to reach sorption equilibrium [year]

Susp_X = Suspended matter in area X [kg/m^3]
where X can be M = Model Area
G = Grepen
B = Baltic Sea

K_d = Distribution coefficient, ratio of element concentrations in the solid and the dissolved phase [m^3/kg], see Table A-10 in Appendix A

This expression accounts for slow sorption kinetics by including rate constants for sorption and desorption, similar to uptake and elimination when modelling a time dependent uptake and accumulation in fish, and has been used earlier e.g. by /Nordlinder et al, 1997/. If T_k is very short relative to water or particle residence times, the equilibration is quasi-simultaneous and the magnitude of T_k is of no importance.

3.3 Sedimentation and resuspension processes

A fraction of the suspended matter is assumed to reach the upper sediment through sedimentation (gross sedimentation). This transfer is described by the ratio of the mean settling velocity and the mean water depth:

$$\mathrm{TC} = \frac{V_{SINK}}{D_X}$$

where

V_{SINK} = Particle settling velocity [m/year]
D_X = Mean water depth [m] in area X (X as above)

A fraction of the deposited material is assumed to be transferred back to the water phase by resuspension and the remaining part is assumed to accumulate on the bottom sediments. The sediment is, therefore, described by two compartments, one for the upper, biologically active, sediment and one for the deeper layers. This allows the importance of varying rates of transfer back to the suspended material in water or of accumulation in the sediments to be investigated. The turnover of radionuclides in the upper sediment is described by using the maximum growth rate of this sediment layer (estimated from observed sediment accumulation rates in sheltered coastal areas nearby /Meili et al, 2000a,b/) and the depth of mixing by bioturbation, e.g. /Eckhéll et al, 2000/. The fluxes are partitioned according to the fraction of accumulation bottoms obtained from wave-theory based models of the area /Brydsten, 1999b/. Thus, the transfer from the upper sediment back to the suspended matter (resuspension) is described by scaling the turnover of this sediment layer to the area exposed to strong erosion forces:

$$\mathrm{TC} = \frac{\mathrm{G_S}}{\mathrm{D_S}} \cdot (1 - \mathrm{FRAC_X})$$

and the transfer from the upper sediment to deeper sediment layers (burial) is calculated using an equivalent expression scaled to the sheltered accumulation area:

$$\mathrm{TC} = \frac{\mathrm{G_S}}{\mathrm{D_S}} \cdot (\mathrm{FRAC_X})$$

where

$\mathrm{G_S}$ = Sediment growth rate [m/year]
$\mathrm{D_S}$ = Depth of upper sediment [m]
$\mathrm{FRAC_X}$ = Fraction of accumulation bottoms in area X (X as above)

The influence of the transfer of radionuclides to the sediments is tested by running the model with and without local sedimentation for an element with strong sorption (high K_d). As can be seen in Table 3-1 this does not influence the amount of radionuclides present in the local water. Instead the radionuclides leave Öregrundsgrepen to a larger extent. As the dose to humans is calculated from the concentration of radionuclides in the water of the Model Area (see section 3.5 and Chapter 9) this means that there is no influence of local sedimentation on the calculated dose to humans in the model of **this** coastal area. For the accumulation of radionuclides in the sediment this process is of importance anyway.

Table 3-1. Comparison of the distribution of radionuclides between different areas of the model system (%) with and without local sedimentation in the Model Area and Öregrundsgrepen (K_d=1 000).

Sedimen-tation	Model Area		Öregrundsgrepen		Baltic Sea		Oceans
	water	sediment	water	sediment	water	sediment	total
Yes	$3\cdot10^{-4}$	2	$3\cdot10^{-3}$	21	$8\cdot10^{-2}$	74	3
No	$3\cdot10^{-4}$	–	$3\cdot10^{-3}$	–	$1\cdot10^{-1}$	96	4

3.4 Parameter values

The parameter values used when the coastal model is used for present conditions (Coast 1) are presented in Table 3-2. Values for the element specific distribution coefficient (K_d) can be found in Table A-10 (Appendix A).

Table 3-2. Parameter values used in the coastal model when simulating conditions of today.

"Model Area"

Parameter	Unit	Type of distr	Best estimate	Min	Max	Reference
Area (A_M)	km^2	C	11.2	11.2	11.2	/Kautsky, 2001/ [1]
Mean depth (D_M)	m	T	9.5	8.5	10.5	Estimated from /Engqvist and Andrejev, 1999/ [2]
Suspended matter (Susp_M)	kg/m^3	T (log)	$5\cdot10^{-3}$	$2.5\cdot10^{-3}$	$10\cdot10^{-3}$	[3]
Water retention time (RETTIME_M)	years	T (log)	$2.11\cdot10^{-3}$	$1.05\cdot10^{-3}$	$4.22\cdot10^{-3}$	/Engqvist and Andrejev, 1999/ [4]
Fraction accumulation bottoms (FRAC_M)	–	T	0.22	0	0.44	/Brydsten, 1999b/ [5]
Fine particle settling velocity (V_{sink})	m/year	T (log)	365	73	7 300	[3]
Half-time to reach sorption equilibrium (T_k)	year	T (log)	10^{-3}	10^{-5}	10^{-1}	[6]
Sediment growth rate (G_S)	m/year	T (log)	0.01	0.004	0.02	[7]
Depth of upper sediment (D_S)	m	T (log)	0.02	0.005	0.05	/Eckhéll et al, 2000/ [8]

Öregrundsgrepen

Parameter	Unit	Type of distr	Best estimate	Min	Max	Reference
Area (A_G)	km^2	T	456	433	479	/Kautsky, 2001/
Mean depth (D_G)	m	T	11.2	10.2	12.2	/Kautsky, 2001/ [2]
Suspended matter (Susp_G)	kg/m^3	T (log)	$5\cdot10^{-3}$	$2.5\cdot10^{-3}$	$10\cdot10^{-3}$	[3]
Water retention time (RETTIME_G)	years	T	0.033 (12.1 days)	0.030	0.036	Estimated from /Engqvist and Andrejev, 1999/ [4]
Fraction accumulation bottoms (FRAC_G)	–	T	0.3	0.15	0.60	/Brydsten, 1999b/ [9,10]
Fine particle settling velocity (V_{sink})	m/year	T (log)	365	73	7 300	[3]
Half-time to reach sorption equilibrium (T_k)	year	T (log)	10^{-3}	10^{-5}	10^{-1}	[6]
Sediment growth rate (G_S)	m/year	T (log)	0.01	0.004	0.02	[7]
Depth of upper sediment (D_S)	m	T (log)	0.02	0.005	0.05	/Eckhéll et al, 2000/ [8]

Table 3-2 (cont'd)

Baltic Sea

Parameter	Unit	Type of distr	Best estimate	Min	Max	Reference
Area (A_B)	km^2	T	377 400	370 000	385 000	/SNA, 1992a/ [11]
Mean depth (D_B)	m	T	56	52	60	/SNA, 1992a/ [11]
Suspended matter ($Susp_B$)	kg/m^3	T (log)	$1 \cdot 10^{-3}$	$5 \cdot 10^{-4}$	$2 \cdot 10^{-3}$	[3]
Water retention time ($RETTIME_B$)	years	T	22	15	30	/SNA, 1992a/ [11]
Fraction accumulation bottoms ($FRAC_B$)	–	T	0.3	0.15	0.5	/Jonsson et al, 1990/ [12]
Fine particle settling velocity (V_{sink})	m/year	T (log)	365	73	7 300	[3]
Half-time to reach sorption equilibrium (T_k)	year	T (log)	10^{-3}	10^{-5}	10^{-1}	[6]
Sediment growth rate (G_S)	m/year	T (log)	0.002	0.0008	0.005	[7]
Depth of upper sediment (D_S)	m	T (log)	0.02	0.005	0.05	/Eckhéll et al, 2000/ [8]

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[1] No parameter range is set as the area is defined and delimited within the SAFE-project /Brydsten, 1999a/.

[2] The uncertainty in data was estimated by Lars Brydsten, University of Umeå, Sweden (personal communication spring 2000).

[3] $Susp_B$ and V_{sink}: Gross sedimentation (= $Susp_B \cdot V_{sink}$) and resuspension need to balance observed sediment growth (see [7] below), which is about 1 or a few mm/y or approximately 0.1 kg m^{-2} y^{-1} in the open Baltic /SNA, 1992a; Ilus et al, 1995; Eckhéll et al, 2000/, and about 1 cm/y in the SFR area, similar to other coastal archipelagos /Meili et al, 2000b; Ilus et al, 1995/; $Susp_B$ is supported by e.g. /Pempkowiak et al, 1998/ and /Nordvarg and Johansson, in prep/; V_{sink} is supported by suspended matter concentrations and sediment trap studies /Lindström et al, 1999; Weyhenmeyer et al, 1997; Meili unpublished/ and agrees with Stoke's law: V_{sink} = 0.2…20 m/d for silt 2…20 µm and <0.01…0.3 m/d for plankton 2…20 µm. For compatibility with coastal values, surface water values are given for the open Baltic (high $Susp_B$, low V_{sink}) rather than deep water values (low $Susp_B$, high V_{sink}). The ranges are rough estimates of uncertainty in the annual mean over larger areas, where concentrations and fluxes can vary by over an order of magnitude within days but also among sites /Weyhenmeyer et al, 1997/.

[4] The best estimate for Öregrundsgrepen was estimated from the average retention time in the upper 35 m of the water column, given in Table 4 in /Engqvist and Andrejev, 1999/. The longer retention times of the deepest water layers (50 and 60 m depth) was disregarded due to their small volumes compared to the whole volume of Öregrundsgrepen. Estimation of the uncertainty based on the variation of the calculated average retention time for different layers gives a range of 10 % /Engqvist and Andrejev, 1999/ for the Model Area and about 5 % for Grepen (the range is likely to decrease with increasing volume) for the mean value, all obtained from a hydrological model that has not been thoroughly validated. To be conservative the parameter value is assumed to be uncertain by a factor of two for both areas.

[5] The lower limit (0 %) was set according to /Sigurdsson, 1987/ who found no accumulation bottoms in the investigated area. The upper limit was set as twice the best estimate value from a sediment erosion model /Brydsten, 1999b/ as a rough estimate since no quantification of uncertainty was presented in the referred study.

[6] Best estimate set to a few hours, minimum set to a few minutes, maximum based on the extremely slow cesium sorption to clay calculated from observed desorption rate and partitioning coefficient, e.g. /Meili and Wörman, 1996, 1997/. Note that the values may be element-specific and thus the uncertainty ranges smaller for each element.

[7] Values based on the observed lamination structure and ^{137}Cs dating of over 100 coastal sediment cores in the region, e.g. /Meili et al, 2000b/. The resulting apparent turnover time of the mixed upper sediment layer (around 2 cm, e.g. /Eckhéll et al, 2000/) is about 2 years in coastal accumulation areas with a sediment growth rate of about 1 cm/year, and about 10 years in offshore areas with a growth rate of 0.2 cm/year.

[8] The range 0.5–5.5 cm is given as the total variation in /Eckhéll et al, 2000/.

[9] The value given for the "outer model area" /Brydsten, 1999b/ was used for the whole Öregrundsgrepen.

[10] The range is a rough estimate since no quantification of the uncertainty was presented in the referred study /Brydsten, 1999b/.

[11] The whole Baltic was chosen as one recipient, since the water exchange among its basins is far more rapid than the exchange with the North Sea.

[12] 30 % in the Baltic Proper /Jonsson et al, 1990/, less in other basins.

As time goes by Öregrundsgrepen will become shallower and the strait will close at about 4 000 AD /Kautsky, 2001/. To model these conditions the values of some parameters for the Model area and Öregrundsgrepen are adapted to the assumed conditions at 4 000 AD (Coast 2). This also influences the water turnover and sedimentation conditions of the area which are important parameters when modelling dispersion of radionuclides in aquatic environments. The model structure is not changed. The parameter values used are presented in Table 3-3. The conditions of the Baltic Sea will also be different at that time but this is not considered here, the same parameter values as for the conditions today (Table 3-2) will be used. This only influences the secondary inflow of radionuclides from the Baltic to Öregrundsgrepen, as this flux may be underestimated since the volume in the Baltic is somewhat too large and the water retention time may be too short. As it is assumed that the radionuclides from SFR will reach the system within the Model area the concentrations of radionuclides will be higher here and the inflow of radionuclides from the Baltic will be of minor importance.

Table 3-3. Parameter values used in the coastal model when simulating considered conditions 4 000 AD.

"Model Area"

Parameter	**Unit**	**Type of distr**	**Best estimate**	**Min**	**Max**	**Reference**
Area (A_M)	km^2	T	1.7	0.85	2.55	1, 2
Mean depth (D_M)	m	T	4.3	2.15	6.45	3, 2
Suspended matter ($Susp_M$)	kg/m^3	T (log)	$5 \cdot 10^{-3}$	$2.5 \cdot 10^{-3}$	$10 \cdot 10^{-3}$	4
Water retention time ($RETTIME_M$)	years	T	0.03	0.015	0.045	5, 2
Fraction accumulation bottoms ($FRAC_M$)	–	T	0.95	0.80	0.99	6
Fine particle settling velocity (V_{sink})	m/year	T (log)	365	73	7 300	4
Half-time to reach sorption equilibrium (T_k)	year	T (log)	10^{-3}	10^{-5}	10^{-1}	4
Sediment growth rate (G_S)	m/year	T (log)	0.01	0.004	0.02	4
Depth of upper sediment (D_S)	m	T (log)	0.02	0.005	0.05	4

Table 3-3 (cont'd)

Öregrundsgrepen

Parameter	Unit	Type of distr	Best estimate	Min	Max	Reference
Area (A_G)	km^2	T	228	114	342	7, 2
Mean depth (D_G)	m	T	7.5	3.75	11.25	7, 2
Suspended matter ($Susp_G$)	kg/m^3	T (log)	$5 \cdot 10^{-3}$	$2.5 \cdot 10^{-3}$	$10 \cdot 10^{-3}$	4
Water retention time ($RETTIME_G$)	years	T	0.027	0.0135	0.0405	5, 2
Fraction accumulation bottoms ($FRAC_G$)	–	T	0.9	0.75	0.95	6
Fine particle settling velocity (V_{sink})	m/year	T (log)	365	73	7 300	4
Half-time to reach sorption equilibrium (T_k)	year	T (log)	10^{-3}	10^{-5}	10^{-1}	4
Sediment growth rate (G_S)	m/year	T (log)	0.01	0.004	0.02	4
Depth of upper sediment (D_S)	m	T (log)	0.02	0.005	0.05	4

1 Estimated from /Brydsten, 1999a/.

2 The range of this parameter is set +/– 50 % of the best estimate, a relatively wide interval, since the values are estimations from predictions of the future conditions and therefore associated with large uncertainty.

3 Between 2 000 AD and 5 000 AD when the lake is formed the mean depth decreases from 9.5 to 1.7 m. If this decrease is assumed to be constant the mean depth at 4 000 AD will be about 4.3 m.

4 The same value as used for 2 000 AD (see Table 3.2).

5 Value estimated from volumes of different water layers (for the Model area in /Engqvist and Andrejev, 1999/, for Öregrundsgrepen in /Kautsky, 2001/) and water retention times from /Engquist and Andejev, 2000/.

6 Brydsten L personal communication 2000-12-11. The range has been set by the authors.

7 According to /Brydsten, 1999a/ the water area of Öregrundsgrepen has decreased to half of the area at 2 000 AD and the volume will be decreased with two thirds (cf Table 3.2). The mean depth is calculated according to this statement.

The correlations used in the coastal models are presented in Table 3-4.

Table 3-4. Correlation coefficients used in the coastal model.

Parameter 1	Parameter 2	Correlation coefficient
Meat consumption	Fish consumption	–0.7
K_d	Bioaccumulation factor, fish	–0.7
Sediment growth rate	Depth of upper sediment	0.8[1]

[1] For Model area, Öregrundsgrepen and the Baltic Sea.

3.5 Uptake in biota and exposure pathways

The total amount of radionuclides in the water and suspended matter compartments divided by the water volume is used when calculating concentration in aquatic foodstuff. This is done using bioaccumulation factors, BAF (see Tables A-13 and A-15 in Appendix A, bioaccumulation factors for freshwater fish are presented in Table A-14), i.e. factors illustrating the partitioning of an element between water and aquatic organisms at steady-state conditions:

$$\mathrm{BAF} = \frac{[\mathrm{X}]_{\mathrm{organism}}}{[\mathrm{X}]_{\mathrm{water}}}$$

where

$[\mathrm{X}]_{\mathrm{organism}}$ = The concentration of the element X in the organism [Bq/kg ww]
$[\mathrm{X}]_{\mathrm{water}}$ = The concentration of X in the water (suspended matter included) [Bq/l]

Values of bioaccumulation factors for a given element vary widely among organisms as well as environments. The documentation in literature of how these values are determined is often poor /Karlsson et al, 2001/. One such uncertain factor is how the concentration in water has been obtained as it can be filtered or not before measuring the radioactivity. In this study, the amounts of radionuclides in water and on suspended matter have been added because it has been assumed that the bioaccumulation factors used are valid for unfiltered water. The separation of water and suspended matter in the model structure enables the use of only water concentration if such values are available.

Cattle are assumed to graze on the shores and then drink the contaminated water and eat water plants. The radionuclides they thereby consume are partly transferred to milk and meat and this process is modelled with element specific transfer coefficients (see Tables A-11 and A-12 in Appendix A).

The human exposure pathways considered in this coastal model are:

- Consumption of fish.
- Consumption of milk and meat from cattle drinking the water and eating aquatic plants.

In the former study /Bergström et al, 1999/ consumption of algae was also assumed as an exposure pathway. Parts of algae are today used as gelling agent in i.e. ice creams but are then taken from marine species. Algae can also be used as protein source but the amount eaten are assumed to be very small and therefore exposure from consumption of algae has not been considered in this study. Bioaccumulation factors for algae are presented in Table A-15 in Appendix A enabling this exposure pathway to be considered if interesting.

The expressions used for dose calculations are further described in Chapter 9.

4 Lake model

The lake model consists of compartments for water, suspended matter and two compartments for the sediments; the upper centimetres (called upper sediment) and the deeper layers (here simply called sediment), see Figure 4-1. Radionuclides can be present within the sediments deposited at marine conditions before the formation of the lake from a coastal bay due to land uplift and/or may enter the model system via the incoming water and suspended matter. Irrigation with the lake water was also modelled, see Chapter 8.

In the SAFE-study radionuclides earlier deposited in coastal sediments were not included as a source of radionuclides when the lake model was applied, see discussion in section 10.2.1. Instead radionuclides accumulated in the sediments of this model was added to the pool which was than used as a source in the agricultural land model.

The model contains two sediment compartments in order to allow for long-term sediment retention. Suspended matter is considered separately to account for the influence of sorption (K_d) on the retention and bioavailability of radionuclides in lake ecosystems. Basic concepts and parameters used for the lake model are largely equivalent to those used for the coastal model and are explained to more detail in Chapter 3.

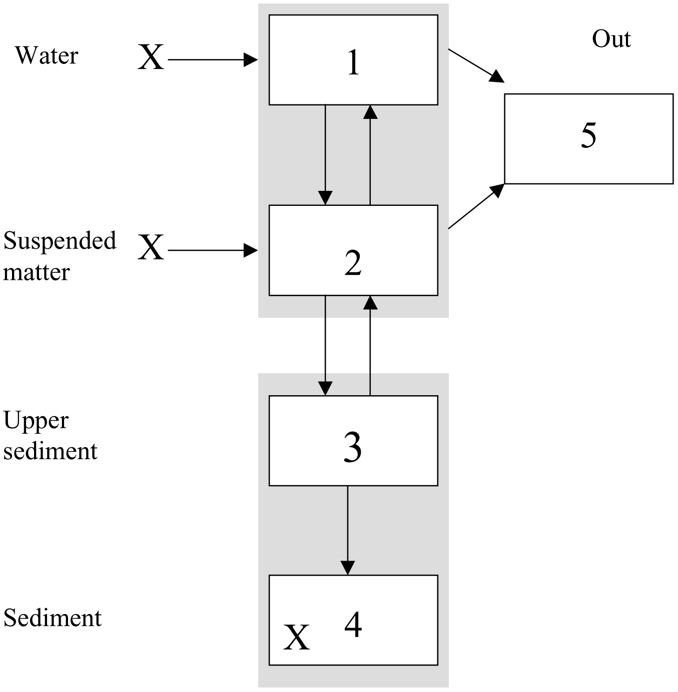


Figure 4-1. *Structure of the lake model. The model consists of compartments for water, suspended matter and two compartments for the sediments; the top few centimetres and the deeper layers. Transfers of radionuclides within the system are marked with arrows. The crosses mark the possible sources of radionuclides.*

4.1 Turnover of water and suspended matter

The inflow of water and suspended matter to the lake depends of course on its position in the drainage area. The inflow of water may be from a watercourse or another lake if the lake is not a springlake. If modelling two lakes connected to each other the outflow of water and suspended matter from the first can be set to enter as inflow in the other one. The radionuclides in the water or on suspended matter are assumed to move with the same rate as the carrying medium.

The outflow of water is a function of the retention time:

$$TC = \frac{1}{RETTIME_X}$$

where

$RETTIME_X$ = Water retention time in lake X [years]

The suspended matter follows the water and therefore the same transfer coefficient is used. Particulate matter as a whole may have a longer retention time because of settling and resuspension, which is accounted for by the exchange between suspended and surficial sediments.

4.2 Interaction of radionuclides between water and particles

The transfer of radionuclides between the water phase and suspended matter is described by the same expression as is used in the coastal model (Chapter 3) but the parameter values used differ. The transfer of radionuclides from the water to the suspended matter is described as:

$$TC = \frac{\ell n2}{T_k} \cdot Susp_X \cdot K_d$$

and the opposite transfer, from the suspended matter to the water as:

$$TC = \frac{\ell n2}{T_k}$$

where

T_k = Half-time to reach sorption equilibrium [year]
$Susp_X$ = Suspended matter in water [kg/m^3]
K_d = Distribution coefficient, concentration of the element on particles in suspended matter relative to dissolved form [m^3/kg]

4.3 Sedimentation and resuspension

A fraction of the suspended matter is assumed to reach the upper sediment through sedimentation. This process is described by the mean particle settling velocity and the mean water depth (cf Chapter 3):

$$TC = \frac{V_{SINK}}{D_X}$$

where

V_{SINK} = Particle settling velocity [m/year]
D_X = Mean water depth [m]

A fraction of the deposited material will be transferred back to the water by resuspension and the other part is assumed to accumulate in the bottom sediments. These processes are also modelled with the same functions as in the coastal model. The transfer from the upper sediment back to suspended matter (resuspension) is described as:

$$TC = \frac{G_S}{D_S} \cdot (1 - FRAC_X)$$

and the transfer from the upper sediment to deeper sediment layers (burial) as:

$$TC = \frac{G_S}{D_S} \cdot (FRAC_X)$$

where

G_s = Sediment growth rate [m/year]
D_s = Depth of upper sediment [m]
$FRAC_X$ = Fraction of accumulation bottoms in lake X [–]

4.4 Parameter values

The evolution of the SFR area has been predicted in the BIOSAFE study /Kautsky, 2001/. Data concerning areas, mean depths and water retention times for lakes, which may appear in the area in the future has been collected from /Brunberg and Blomqvist, 2000/. The large lake, which will appear in the area about 4 900 AD, is in the mentioned study called “Lake No. 4” and the same name is used here, see Table 4-1. Generally wide ranges are used for the parameter values, since the application of observations or estimates from present lakes for predictions in future lakes emerging from the sea should be seen as rough estimates. Values for the element specific distribution coefficients (K_d) can be found in Table A-9. A number of lakes with different sizes will appear and disappear in the area over a time period of 3 000 years from today /Brydsten, 1999a/. In case of a constant release of radionuclides from the repository to Lake 4 the concentration of radionuclides in the lake water will always be highest in this lake and decreasing in the lakes downstream as a consequence of dilution. If the release of radionuclides varies with time it may be the case that the concentrations, at a certain time, may be higher in any of the lakes downstream than in the first lake. The maximum concentrations of radionuclides in lake water for the time period will, anyway, always be in the lake where the release takes place. In this report, parameter values for Lake 4 only are presented but it is possible to calculate doses to humans also from any other of the predicted lakes.

Table 4-1. The parameter values used in the modelling of ”Lake No. 4”.

Lake No. 4						
Parameter	**Unit**	**Type of distr**	**Best estimate**	**Min**	**Max**	**Reference**
Area (A_{L4})	km^2	T	1.06	0.8	1.25	/Brunberg and Blomqvist, 2000/ [1]
Mean depth (D_{L4})	m	T	1.7	1.4	2.1	/Brunberg and Blomqvist, 2000/ [1]
Suspended matter ($Susp_{L4}$)	kg/m^3	T (log)	$2 \cdot 10^{-3}$	$5 \cdot 10^{-4}$	$5 \cdot 10^{-3}$	/Lindström et al, 1999/ [2]
Water retention time ($RETTIME_{L4}$)	years	T	0.24 (89 days)	0.17	0.31	/Brunberg and Blomqvist, 2000/ [1]
Fraction accumulation bottoms ($FRAC_{L4}$)	–	T	0.2	0.0	1.0	[3]
Fine particle settling velocity (V_{sink})	m/year	T (log)	183 (0.5m/day)	36.5	3 600	[4]
Half-time to reach sorption equilibrium (T_k)	year	T (log)	10^{-3}	10^{-5}	10^{-1}	[5]
Sediment growth rate (G_S)	m/year	T (log)	0.004	0.001	0.010	[6]
Depth of upper sediment (D_S)	m	T (log)	0.02	0.005	0.05	/Eckhéll et al, 2000/ [7]

[1] No estimate of the uncertainty in data was presented in the study. Predictions of this sort can be seen as rough estimates, so a wide range was used.

[2] The range and mean value are estimated from data in /Lindström et al, 1999/.

[3] Since no information about the extension of accumulation sediment in the future lakes is available, the whole range (0–100 %) was assumed to be possible.

[4] The particle settling velocity is i.a. depending on the size and density of the settling particles. In lakes a large fraction of the particles and sediments is organic matter which tends to sink at a slower rate than mineral particles of the same size (cf Table 3.2). A mean value half as large as that used in the coastal model was therefore used, and the range was adopted accordingly (see Table 3-2).

[5] Best estimate set to a few hours, minimum set to a few minutes, maximum based on the extremely slow caesium sorption to clay calculated from observed desorption rate and partitioning coefficient, e.g. /Meili and Wörman 1996, 1997/. Note that the values may be nuclide-specific and thus the uncertainty ranges narrower for each nuclide.

[6] As in the coastal model (cf Table 3.2) this value was estimated from sediment mass balance calculations of the lake. A growth rate of 0.4 cm/year corresponds to a turnover time for the upper sediment layer (2 cm) of about 5 years.

7 The range 0.5–5.5 cm is given as the total variation in /Eckhéll et al, 2000/.

The correlations used in the lake model are presented in Table 4-2.

Table 4-2. Correlation coefficients used in the lake model.

Parameter 1	Parameter 2	Correlation coefficient
Meat consumption	Fish consumption	–0.7
K_d soil	Root uptake factor, root crops	–0.8
K_d soil	Root uptake factor, cereals	–0.8
K_d soil	Root uptake factor, pasturage	–0.8
Root uptake factor, cereals	Root uptake factor, pasturage	0.64
Root uptake factor, cereals	Root uptake factor, root crops	0.64
Root uptake factor, pasturage	Root uptake factor, root crops	0.41
Porosity of top soil	Porosity of deep soil	0.7
Consumption of cereals	Consumption of root crops	–0.7
Bioaccumulation factor, fish	K_d suspended matter in lake	–0.7
Sediment growth rate	Depth of upper sediment	0.8

4.5 Uptake in biota and exposure pathways

The total amount of radionuclides in the water and suspended matter compartments is divided by the water volume when calculating the concentration in aquatic foodstuff with bioaccumulation factors (see Tables A-13 and A-15 in Appendix A), see section 3.5.

The human exposure pathways considered are:

- Consumption of water.
- Consumption of fish.
- Consumption of milk and meat from cattle which has been grazing along the shores where they consume aquatic plants as well as water (radionuclide redistribution within cattle are described in section 3.5).

In the former study /Bergström et al, 1999/ consumption of crustacean was also assumed as an exposure pathway. In this study this is not considered since the lakes which may form in the area in the future are thought to be to shallow and muddy to harvest crustacean of interest as food for humans. Bioaccumulation factors for crustacean are presented in Table A-15 in Appendix A enabling this exposure pathway to be considered if interesting.

Irrigation of a garden plot with lake water has also been included in this model (see description in Chapter 8). The radionuclides within the irrigation water are transferred to the topsoil where they are available for root uptake (see section 6.7). A fraction of the radionuclides is also assumed to be retained on the vegetation surface. Transfer to edible parts (for cereals and root crops) is then calculated with a translocation factor (see Table A-6 in Appendix A).

The additional exposure pathways due to irrigation are:

- Consumption of vegetables and root crops grown on irrigated soil.
- Intake of soil via for example insufficiently washed vegetables.
- External exposure from contaminated ground.
- Inhalation of resuspended particles from garden plots.

The expressions used for dose calculations are further described in Chapter 9.

5 Agricultural land model

According to the investigations of the evolution of the SFR area /Kautsky, 2001/ there will most probably not be any large areas suitable for cultivation of crops in the future. The only area which may reasonably be usable for large-scale cultivation is the former sediments of the large lake.

The agricultural land model describes a soil with a near horizontal flow of groundwater where the water table is situated about one meter below the surface. The radionuclides can be assumed to be present within the soil as a consequence of accumulation in former sediment when the area was below water. Inflow of radionuclides with groundwater may also be simulated. The simulated agricultural fields are assumed to be ditched or have draining-tiles so the groundwater level could be assumed to be relatively constant. Nuclides are transported from the saturated zone up to the upper soil layers via processes as capillary rise, root-uptake and diffusion. The saturated zone (below the groundwater table) is divided into two compartments; one representing the pore water and one the solid material, see Figure 6-1. The unsaturated zone consists of a surface layer (here called top soil) which represents the part which is influenced by ploughing and a deeper layer (deep soil). Outflow of radionuclides from the system may occur with groundwater or through erosion processes. The crops grown on the soil are used as food for humans as well as cattle.

Radionuclides accumulated in the sediments during the coastal and lake stage are present within the soil as a secondary source. The fraction accumulated during the lake stage is present in the top soil layer whereas those from the coastal stages are added to the deep soil and the solid fraction of the saturated zone.

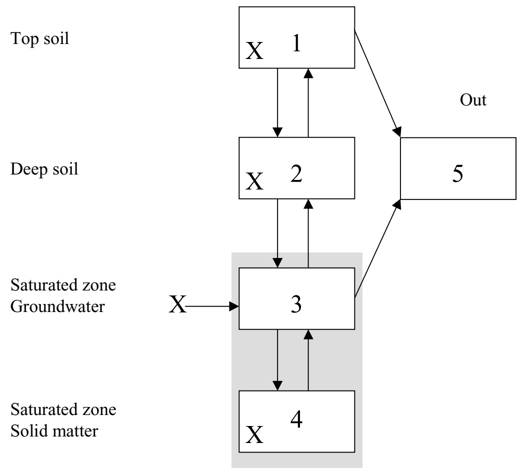


__Figure 5-1.__ Structure of the agricultural land model. The model consists of two zones; the upper unsaturated and the lower saturated zone. The saturated zone consists of one compartment for the groundwater and one for the solid matter. The unsaturated zone consists also of two compartments; one for the uppermost layer and the other for the layer beneath. Transfers of radionuclides within the system are marked with arrows. The crosses mark the possible sources of radionuclides.

The structure of the soil model is comparable to the models for surface waters (coastal and lake) using two compartments to simulate what happens in the inflow areas (in this model the saturated zone). Thereafter only two compartments are used to describe the exchange of the elements in the upper soil horizon. This is the general approach for dose assessments models for high-level waste disposal /BIOMOVS II, 1996a/.

Comparison of concentrations in soil was carried out using only one compartment for the saturated zone and the same expression as was used for the turnover between the two upper soil compartments when describing the transfer between saturated zone and deep soil. The resulting concentrations in top soil were similar to those when the saturated zone was divided into two compartments, highest discrepancy was a factor of two. Neglecting any adsorption in the aquifer leads, on the other hand, to substantially higher concentrations in the upper soil, from which the exposures to humans are calculated. This approach was not selected, however, because it is most probable that a fraction of the elements attaches to the solid phase as is considered for transport calculations in the geosphere. The division of the saturated zone into two compartments was also made in order to prepare for improvements of the model.

5.1 Transfers between solid and soluble fractions in saturated zone

The distribution of elements between dissolved and solid fractions in the saturated zone is described by one parameter (here called K_d) although it involves chemical, biological and physical processes. These are time-dependent processes and therefore a parameter for the half-time to reach sorption equilibrium is used. The transfer coefficients from dissolved to solid matter is scaled to the relation between solid matter and pores in soil and is given by the expression:

$$TC = \frac{K_d \cdot \ln(2)}{T_k} \cdot \frac{(1-\varepsilon_{sa})D_{sa} \cdot A \cdot \rho_{sa}}{\varepsilon_{sa} \cdot D_{sa} \cdot A} = \frac{K_d \cdot \ln(2)}{T_k} \cdot \frac{(1-\varepsilon_{sa}) \cdot \rho_{sa}}{\varepsilon_{sa}}$$

and from particulate to dissolved fraction by:

$$TC = \frac{\ln(2)}{T_k}$$

where

K_d = Distribution coefficient, concentration of the element on solids relative to dissolved [m^3/kg], see Table A-7
T_k = Half-time to reach sorption equilibrium [year]
ε_{sa} = Porosity in saturated zone [m^3/m^3]
D_{sa} = Depth of saturated zone [m]
A = Area of agricultural land [m^2]
ρ_s = Density of soil particles [kg/m^3]

5.2 Transfer from saturated zone

The transfer coefficient describing the horizontal flow of dissolved radionuclides in groundwater out from the saturated zone is based on water balance and becomes:

$$TC = \frac{R}{\varepsilon_{sa} \cdot D_{sa}}$$

where

R = Runoff $[m^3/(m^2 \cdot year)]$
ε_{sa} = Porosity of saturated zone $[m^3/m^3]$
D_{sa} = Depth of saturated zone [m]

5.3 Transfers between saturated zone and deep soil

The direction of water flow between saturated and unsaturated zones varies with time /BIOMOVS II, 1996b/. Processes as diffusion and capillary rise cause an upward flow from the saturated to the unsaturated zone, during dry periods, while precipitation generates a flow in the opposite direction. The upward flow, $F_{sa,ds}$ is estimated to 200 mm per year, using results from lysimeter experiments in which the saturated level was held constant /BIOMOVS II, 1996b/. The transfer coefficient from saturated to deep unsaturated soil considering the fraction of nuclides in soluble form in the soil pores becomes:

$$TC = \frac{F_{sa,ds}}{\varepsilon_{sa} \cdot D_{sa}}$$

where

$F_{sa,ds}$ = Upward flow $[m^3/(m^2 \cdot year)]$
ε_{sa} = Porosity of saturated zone $[m^3/m^3]$
D_{sa} = Depth of saturated zone [m]

The downward transfer of radionuclides from the deep soil to the saturated zone is expressed by:

$$TC = \frac{(R + F_{ds,sa})}{\varepsilon_{ds} \cdot D_{ds}} \cdot Ret$$

where

$F_{ds,sa}$ = Downward flow $[m^3/(m^2 \cdot year)]$
R = Runoff $[m^3/(m^2 \cdot year)]$
ε_{ds} = Porosity of deep soil $[m^3/m^3]$
D_{ds} = Depth of deep soil [m], (see below)
Ret = Retention, (see below) [–]

$$D_{ds} = 1 - D_{ts}$$

where

D_{ts} = Depth of top soil [m]

$$\mathrm{Ret} = \frac{1}{1 + K_d \cdot \rho_p \cdot \frac{1-\varepsilon_i}{\varepsilon_i}}$$ /Andersson et al, 1982/

where

K_d = Distribution coefficient, concentration of the element on solids relative to dissolved [m^3/kg], see Table A-7
ρ_s = Density of soil particles [kg/m^3]
ε_i = Porosity of the matter in compartment i (here deep soil) [m^3/m^3]

5.4 Transfers between deep soil and top soil

The upward flow from deep soil to top soil is estimated from the same lysimeter experiment as mentioned above /BIOMOVS II, 1996b/. An important process for transport of soil, and thereby radionuclides, between deep soil and top soil is bioturbation, mainly caused by earthworms /Müller-Lemans and van Dorp, 1996/. The transfer coefficient from deep soil to top soil, considering these factors, becomes:

$$TC = \frac{F_{ds,ts}}{\varepsilon_{ds} \cdot D_{ds}} \cdot \mathrm{Ret} + \frac{\mathrm{BioT}}{(1-\varepsilon_{ds}) \cdot \rho_p \cdot D_{ds}}$$

where

$F_{ds,ts}$ = Upward flow [$m^3/(m^2 \cdot year)$]
ε_{ds} = Porosity of deep soil [m^3/m^3]
D_{ds} = Depth of deep soil [m], see above
Ret = Retention, see above [–]
BioT = Bioturbation, (transport of soil by earthworms) [kg/($m^2 \cdot$year)]
ρ_p = Density of soil particles [kg/m^3]

When using the expression above all the pores in the unsaturated zone are assumed to be able to hold water and therefore the porosity is used in the expression. This simplification may underestimate the migration rates as, in reality, some pores are not water filled. This leads to an underestimation of the upward transport of water and radionuclides but also to an underestimation of the transport in the different direction (see below).

The transfer coefficient from top soil to deep soil is described by the following equation:

$$TC = \frac{(R + F_{ds,ts})}{\varepsilon_{ts} \cdot D_{ts}} \cdot Ret + \frac{BioT}{(1 - \varepsilon_{ts}) \cdot \rho_p \cdot D_{ts}}$$

where

R = Runoff [$m^3/(m^2 \cdot year)$]
$F_{ds,ts}$ = Downward flow [$m^3/(m^2 \cdot year)$]
ε_{ts} = Porosity of top soil [m^3/m^3]
D_{ts} = Depth of top soil [m]
Ret = Retention (see above) [–]
BioT = Bioturbation (transport of soil by earthworms) [$kg/(m^2 \cdot year)$]
ρ_s = Density of soil particles [kg/m^3]

5.5 Erosion

There is a loss of elements due to erosion or removal of soil during agricultural practices. Geographical conditions, soil types and meteorological parameters such as wind and rain influence the magnitude of erosion. The size and density of the soil particles are also of importance. Land covered with vegetation are less exposed to erosion since the roots bind the material and the plants may cover parts of the surfaces. The loss of radionuclides due to erosion is estimated as:

$$TC = \frac{Rem}{D_{ts}(1 - \varepsilon_{ts}) \cdot \rho_p}$$

where

Rem = Removal of soil [$kg/(m^2 \cdot year)$]
D_{ts} = Depth of top soil [m]
ε_t = Porosity of soil in the top soil (plough layer) [m^3/m^3]
ρ_p = Density of soil particles [kg/m^3]

5.6 Parameter values

Few site-specific data for the area has been used. Instead the parameter values from /Bergström et al, 1999/ are used (see Table 6-1). The parameter values used in the agricultural land model can be seen in Table 6-1. Values for the element specific distribution coefficient (K_d) can be found in Table A-7.

Table 5-1. Parameter values used in the agricultural land model.

Agricultural land

Parameter	Unit	Type of distr	Best estimate	Min	Max	Reference
Runoff (R)	$m^3/(m^2 \cdot year)$	T	0.25	0.20	0.30	/Lindborg and Schüldt, 1998/ [1]
Depth of top soil (D_{ts})	m	T	0.25	0.20	0.30	/Haak, 1983/ [2]
Depth of top and deep soil	m	C	1.0	–	–	[3]
Depth of saturated zone (D_{sa})	m	T	3	2	4	[4]
Top soil porosity (ε_{ts})	m^3/m^3	T	0.5	0.4	0.6	/Wiklander, 1976/ [5]
Deep soil porosity (ε_{ds})	m^3/m^3	T	0.5	0.4	0.6	[6]
Saturated zone porosity (ε_{sa})	m^3/m^3	T	0.3	0.25	0.40	[7]
Soil density (ρ_s)	kg/m^3	T	2 650	2 600	2 700	[8]
Bioturbation (BioT)	$kg/(m^2 \cdot year)$	T	2	1	3	/Müller-Lemans and van Dorp, 1996/ [9]
Water transport from groundwater to deep soil ($F_{sa,ds}$)	$m^3/(m^2 \cdot year)$	T(log)	0.2	0.1	0.3	/BIOMOVS II, 1996b/ [10]
Water transport from deep soil to top soil ($F_{ds,ts}$)	$m^3/(m^2 \cdot year)$	T	0.10	0.05	0.20	[11]
Water transport from deep soil to groundwater ($F_{ds,sa}$)	$m^3/(m^2 \cdot year)$	T	0.2	0.1	0.3	[12]
Area of agricultural land (A)	m^2	T	530 000	400 000	625 000	[13]
Half-time to reach sorption equilibrium (T_k)	year	T(log)	10^{-3}	10^{-5}	10^{-1}	[14]
Soil removal (Rem)	$kg/(m^2 \cdot year)$	T(log)	0.005	0.002	0.020	[15]

[1] The range is given in /Lindborg and Schüldt, 1998/.

[2] According to /Haak, 1983/ the depth of the plough layer in agricultural soils varies between 20–30 cm.

[3] It is assumed that the unsaturated zone of the soil horizons have a depth of 1 m, which is a plausible value for drained soils in agricultural areas.

[4] The soil used for agricultural purposes is assumed to consist of the sediments deposited in accumulation zones when the area was a coastal area and later a lake /Kautsky, 2001/. The coastal and lake stages are assumed to remain for about 2 500 years each. The sedimentation in accumulation areas during the coastal period is about 1.8 kg dw/($m^2 \cdot$year) (see Table 3-2) and with a density of about 2 600 kg dw/m^3 and a porosity of about 50 % the sediment accumulated during these 2 500 year will be about 3.5 m deep. During the lake period another 0.5 m is added (sedimentation = 0.4 kg dw/($m^2 \cdot$year), calculated from data in Table 4-1). As the upper two layers (top soil and deep soil) together has a depth of 1 m the depth of the saturated zone is here set to 3 m, varying from 2 to 4 m.

[5] According to /Wiklander, 1976/ most agricultural soils have a porosity varying from 0.4 to 0.6, but the porosity can be even higher in organogenic soils.

[6] In lack of better information the same porosity as for the upper soil layer was used.

[7] The porosity in aquifers may vary considerable between 30 to 60 % in loose deposits /Grip and Rodhe, 1985/. In order not to underestimate the upward transport of radionuclides 30 % was selected as a mean value, subjectively ranging from 25 to 40 %.

[8] In most literature an average value of 2 650 kg/m^3 is given for the density of soil particles /Hillel, 1980; Wiklander, 1976; FitzPatrick, 1980/. If the soil contains a large fraction of heavy minerals, the value will be higher while it decreases with increasing fraction of organic matter. In lack of more precise data this average value is selected with narrow ranges as it is assumed that no extreme soil types are present in the area.

[9] The effect of bioturbation depends on the biological activity within the soil which can vary very much and therefore a wide range is used.

[10] The amount of water transported upwards from the groundwater zone to the unsaturated zone (200 mm) was estimated from data obtained in lysimeter experiments /BIOMOVS II, 1996b/.

[11] A transport of 100 mm from deep soil to the top layer, compared to 200 mm from groundwater to deep soil, was selected as e. g. the influence of capillary rise will decrease when the distance to the groundwater table increases. The upper soil layer also receives water through precipitation. The upward transport of water in soil is also driven by the water uptake of vegetation. Part of the water is taken up directly by the plants but a fraction will also remain within the upper soil layer (which is the fraction modelled).

[12] Equilibrium is assumed to have been established so the upward flow of water from the groundwater zone to the deep soil is assumed to be of the same size as the flow in opposite direction. This assumption is reasonable since the soil was assumed to be drained so the level of the groundwater surface is rather constant.

[13] The area used for agricultural purposes is assumed to be available after drainage of the large lake which will arise within the area in the future /Kautsky, 2001/. It is assumed that half of the lake area is drained and cultivated. The lake area is set to 1.06 km^2 (best estimate, range: 0.8–1.25 km^2), see Table 4-1, which gives an agricultural area of 530 000 m^2, ranging from 400 000 to 625 000 m^2.

[14] Best estimate set to a few hours, minimum set to a few minutes, maximum based on the extremely slow cesium sorption to clay calculated from observed desorption rate and partitioning coefficient, e.g. /Meili and Wörman, 1996, 1997/. Note that the values may be nuclide-specific and thus the uncertainty ranges narrower for each nuclide.

[15] The importance of erosion for transport of soil varies considerable. An investigation of erosion in southern Sweden presents losses of 0.5 to 300 tons per ha /Alström and Bergman, 1986/. A later investigation showed much lower values, from 0.001 to 80 tons per ha /Alström and Bergman Åkerman, 1991/. Erosion has also been studied as transport of suspended matter in running waters /Nilsson, 1972/. The results indicate annual losses from 10 up to hundreds of kg per ha. When the parameter range was estimated the extreme values were excluded because the area has low relief.

The correlation coefficients used in the agricultural land model are presented in Table 5-2.

Table 5-2. Correlation coefficients values used in the agricultural land model.

Parameter 1	Parameter 2	Correlation coefficient
K_d soil	Root uptake factor, root crops	–0.8
K_d soil	Root uptake factor, cereals	–0.8
K_d soil	Root uptake factor, pasturage	–0.8
Root uptake factor, cereals	Root uptake factor, pasturage	0.64
Root uptake factor, cereals	Root uptake factor, root crops	0.64
Root uptake factor, pasturage	Root uptake factor, root crops	0.41
Porosity of top soil	Porosity of deep soil	0.7
Consumption of cereals	Consumption of root crops	–0.7

5.7 Uptake in biota and exposure pathways

The concentration of radionuclides within the top soil compartment is used when calculating the concentration in crops grown on the agricultural field. This is done using root uptake factors, RUFs (see Tables A-2–A-5 in Appendix A), i.e. factors illustrating the partitioning of an element between vegetation and soil at steady-state conditions:

$$\mathrm{RUF} = \frac{[\mathrm{X}]_{\mathrm{vegetation}}}{[\mathrm{X}]_{\mathrm{soil}}}$$

where

$[\mathrm{X}]_{\mathrm{vegetation}}$ = The concentration of the element X in the vegetation
$[\mathrm{X}]_{\mathrm{soil}}$ = The concentration of X in dry soil

The effect of root uptake varies dependent on species, soil type, climate and element specific properties /IAEA, 1983/. In addition, cultivation practices of soil such as ploughing, fertilisation and irrigation influence the root uptake /IAEA, 1994/. Data for different plants are available for most elements, though the values vary considerably for each element /IUR, 1984/. The root uptake factors are inversely correlated to the distribution factors in soil (K_d-values) in the calculations as high sorption reduces the bioavailability.

In this study crops are divided into three main groups; (leafy) vegetables, cereals and root crops. These groups cover most of the species in the Scandinavian diet /Becker et al, 1985/.

The crops are assumed to be eaten by humans and cereals are also eaten by cattle. Part of the area is used for grazing. The following exposure pathways are considered:

- Consumption of cereals.
- Consumption of root crops.
- Consumption of vegetables.
- Intake of soil via for example insufficiently washed vegetables.
- Consumption of milk and meat from cattle which have been eating cereals, hay and also unintentionally soil from the field (radionuclide transfer to milk and meat within cattle is described in section 3.5).
- Inhalation of resuspended particles.
- External exposure from contaminated soil.

The expressions used for dose calculations are further described in Chapter 9.

6 Mire model

Mires contain peat which is an accumulation zone for many elements /Statens energiverk, 1985/. When burning peat which has been contaminated with radionuclides, releases of nuclides to air may occur, or the nuclides may be enriched in ash /Nordlinder, 1989/. The structure of the mire model is shown in Figure 6-1. When modelling the transport of radionuclides within and out of a mire area it can be assumed that the radionuclides are present within the peat at the start of the modelling as a consequence of earlier accumulation in sediments when the area was covered with water. This was not considered in the SAFE-study. Inflow of radionuclides may also be modelled. The mire is assumed to be exploited and a fraction of the contaminated peat is used as fuel in a household. Another fraction is drained and used for farming purposes. The crop grown on the peat is then assumed to be used as food for humans as well as cattle.

The structure of the mire model is very simple, consisting of only two compartments which represent the two physical phases present in this kind of ecosystem. It may be argued that the conditions are not homogeneous but there is no data available to support a more detailed model structure. Besides, when the peat is taken away for use as fuel some stirring is certain to occur.

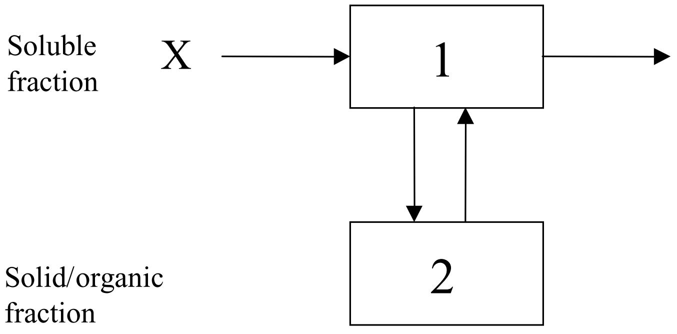


Figure 6-1. *Structure of the mire model. The model consists of two compartments; one for the soluble fraction and one for the solid/organic fraction. Transfer of radionuclides within the system is marked with arrows. The cross marks the principle sources of radionuclides.*

6.1 Transfers between solid and soluble fraction

The distribution of elements between dissolved and solid phases in peat is described by one parameter (here called K_d) although it involves chemical, biological and physical processes. These are time-dependent processes and therefore a parameter for the half-time to reach sorption equilibrium is used. If the reaction velocity is known, the transfer coefficients from dissolved to solid fraction can be given by the expression:

$$TC = \frac{K_d \cdot \ln(2)}{T_k} \cdot \frac{D_p \cdot \rho_p}{\varepsilon_p \cdot D_p} = \frac{K_d \cdot \ln(2)}{T_k} \cdot \frac{\rho_p}{\varepsilon_p}$$

and from particulate to dissolved fraction by:

$$TC = \frac{\ln(2)}{T_k}$$

where

K_d = Distribution coefficient, concentration of the element on solids relative to dissolved [m^3/kg], see Table A-8
T_k = Half-time to reach sorption equilibrium [year]
D_p = Depth of peat in mire [m]
ρ_p = Density of peat [kg/m^3]
ε_p = Porosity in peat [m^3/m^3]

6.2 Outflow of soluble fraction

The transfer coefficient for the horizontal flow is based on water balance and becomes:

$$TC = \frac{R}{\varepsilon_p \cdot D_p}$$

where

R = Runoff [$m^3/(m^2 \cdot year)$]
ε_p = Porosity of peat [m^3/m^3]
D_p = Depth of peat in mire [m]

6.3 Parameter values

The mire model is used when estimating dose to humans in the SFR area in a far future (4 000 AD–12 000 AD). Generally wide ranges are used for the parameter values, since predictions for future mire areas emerging from the sea should be seen as rough estimates.

The general parameter values used can be seen in Table 6-1 below. Values for the element specific distribution coefficient (K_d) can be found in Table A-8. In lack of data specific for peat, data for organic soils were used.

Table 6-1. General parameter values used in the mire model.

Mire

Parameter	**Unit**	**Distr**	**Best estimate**	**Min**	**Max**	**Reference**
Runoff (R)	$m^3/(m^2 \cdot year)$	T	0.25	0.20	0.30	/Lindborg and Schüldt, 1998/ [1]
Density (ρ_p)	kg dw/m^3 peat	T	100	80	120	/Steinmann and Shotyk, 1997/ [2]
Porosity (ε_p)	m^3/m^3	T	0.90	0.80	0.95	/Sharma and Forster, 1993/ [3]
Peat depth (D_p)	m	U	–	0.7	2.1	[4]
Mire area (A)	km^2	U	–	0.24	1.25	[5]
Half-time to reach sorption equilibrium (T_k)	years	T (log)	10^{-3}	10^{-5}	10^{-1}	[6]

[1] The range is given in /Lindborg and Schüldt, 1998/.

[2] The peat density varies with depth but in this model this is not considered, instead an average density for the whole peat is used. The two peat profiles examined in /Steinmann and Shotyk, 1997/ did not differ much in average density which seems rather logical since the density of the organic matter in different mires should not differ very much. The parameter range used here is 20 % of the best estimate.

[3] The porosity is related to the peat density and therefore a relatively narrow range is used.

[4] Most Swedish peat bogs are about 2 m deep (www.torvproducenterna.se/basfakta.html, 2000-08-04). The minimum depth to be classed as peat-bogs are 0.3 m. In this case the mire depth is set according to the depth of the lake basin which is predicted to be gradually transformed into a mire /Kautsky, 2001/. It is assumed that the depth of the mire will be at least half of the lake depth (see Table 4-1), i.e. 0.7 m (half of the minimum lake depth), and at most of the same depth, i.e. 2.1 m (maximum lake depth). A uniform distribution is used since it is not possible to make a better estimation.

[5] It is predicted that the large lake which will form in the area in the future /Kautsky, 2001/ will be gradually transformed into a mire. To estimate the extension of the mire is not easy. Here it is assumed that 30–100 % of the former lake basin (see Table 4-1) will be transformed into a mire, hence the size of the mire is set to minimum 0.24 km^2 (30 % of the minimum lake area) and maximum 1.25 km^2 (100 % of maximum lake area). A uniform distribution is used since it is not possible to make a better estimation.

[6] Best estimate set to a few hours, minimum set to a few minutes, maximum based on the extremely slow caesium sorption to clay calculated from observed desorption rate and partitioning coefficient, e.g. /Meili and Wörman 1996, 1997/. Note that the values may be nuclide-specific and thus the uncertainty ranges narrower for each nuclide.

The correlations used in the mire model are presented in Table 6-2.

Table 6-2. Correlation coefficients used in the mire model.

Parameter 1	Parameter 2	Correlation coefficient
K_d peat	Root uptake factor, root crops	–0.8
K_d peat	Root uptake factor, cereals	–0.8
K_d peat	Root uptake factor, pasturage	–0.8
Root uptake factor, cereals	Root uptake factor, pasturage	0.64
Root uptake factor, cereals	Root uptake factor, root crops	0.64
Root uptake factor, pasturage	Root uptake factor, root crops	0.41
Consumption of cereals	Consumption of root crops	–0.7

6.4 Uptake in biota and exposure pathways

The concentration of radionuclides within the compartment for the solid fraction of the mire is used when calculating the concentration in crops grown in the peat. This is done using the same root uptake factors as in the agricultural land model (see section 6.7).

The crops are assumed to be eaten by humans as well as cattle. The human exposure pathways considered are:

- Consumption of cereals.
- Consumption of root crops.
- Consumption of vegetables.
- Consumption of milk and meat from cattle which has been eating cereals, grass and hay grown on the peat and also unintentionally consuming peat when grazing (radionuclide redistribution within cattle are described in section 3.5).
- External exposure from the mire.

A fraction of the contaminated peat is assumed to be used as fuel in a household. The additional exposure pathways because of this are:

- Inhalation of resuspended particles from the mire.
- Inhalation of nuclides in flue gas produced when peat is burned.

The expressions used for dose calculations are further described in Chapter 9.

7 Well model

In the well model the water is assumed to be used for consumption by man and cattle. Irrigation of a small garden plot with the well water is also modelled, for description see Chapter 8.

7.1 Water turnover

The model is very simple and consists of one compartment for water. The radionuclides reaching the well are assumed to be evenly distributed in the water volume.

7.2 Parameter values

A possible scenario is that a well is drilled within the plume of groundwater which has passed the repository. This is modelled with the well model. The annual volume for mixing is an important parameter since it decides the concentration of radionuclides in the well water. This volume is assumed to be the same as the annual amount of water taken from the well. The critical group for which the dose is calculated consists of 5–10 people, living on a small farm. It is assumed that these people drink 1.6 litres of water per day (600 l/year). In addition to this each person consumes 200 l/day for washing, showering etc. The total water consumption for the group becomes 1–2 m^3/day. It is also assumed that the farm has 5–10 cows and that each cow consumes 65–75 l/day, which gives a total equal to 325–750 l/day. Furthermore, the water from the well is used for irrigation of a garden plot. It is assumed that the garden plot is irrigated 6–14 times per year. The volume of water used at each time is between 0.014–0.067 m^3 of water per m^2 of garden. The area of the garden is assumed to be between 150–250 m^2. Hence, the total volume used for irrigation will be between 12.6 m^3/year and 234.5 m^3/year. Thus, the total use of water by the small farm is between 1.4–3.4 m^3/day (496–1 237 m^3/year). The arithmetic average value of these two numbers is 2.34 m^3/day (867 m^3/year). See Table 7-1.

Mixing volumes for radionuclides in groundwater reaching wells have been discussed during many years. The volume has been determined in various ways over the years. The annual infiltration amount of water over an area the of size of a deep repository has e.g. been used /Bergman et al, 1977/. This approach gave about hundred times larger volumes than the ones used in /Bergström et al, 1999/. In /Bergström et al, 1999/ measured well capacities were taken from the Swedish Well Archive. These measured capacities were transformed to annual mixing volumes by assuming that they were constant all over the year. This lead to narrow ranges of the mixing volumes and a potential for overestimating the volumes, an approach which can be discussed. The same approach as used in this study, i.e. an estimation of mixing volume based on the water need of the users was also used in the safety assessment SKB 91 /Bergström and Nordlinder, 1990b/. The volume used was larger since the group using the well water was assumed to consist of 25 persons with a livestock of 8 cattle.

The correlations used in the well model are presented in Table 7-2.

Table 7-1. General parameter values used in the well model.

Parameter	Unit	Distr	Best estimate	Min	Max	Reference
Well						
Water capacity	m^3/year	T	867	496	1 237	Estimated as described above

Table 7-2. Correlation coefficients used in the well model.

Parameter 1	Parameter 2	Correlation coefficient
K_d soil	Root uptake factor, vegetables	–0.7
K_d soil	Root uptake factor, root crops	–0.56
Root uptake factor, vegetables	Root uptake factor, root crops	0.8
Porosity of top soil	Porosity of deep soil	0.8

7.3 Uptake in biota and exposure pathways

The concentrations of radionuclides in the well water are used when calculating dose to humans in this model. The human exposure pathways considered in the well model are:

- Consumption of water.
- Consumption of milk and meat from cattle drinking well water (radionuclide redistribution within cattle are described in section 3.5).

The fate of the radionuclides within the irrigation water is described in section 4.4. The concentrations of radionuclides in the top soil and on the vegetation surfaces are used when calculating additional exposures due to irrigation. The additional exposure pathways via the irrigation model are:

- Consumption of vegetables and root crops grown on irrigated soil.
- Intake of soil via for example insufficiently washed vegetables.
- External exposure from the contaminated garden plot.
- Inhalation of resuspended particles from the garden plot.

The expressions used for dose calculations are further described in Chapter 9.

8 Sub-model irrigation

Radionuclides in various types of water can be transferred to vegetation and soil through irrigation and therefore a sub-model handling this process is used. The sub-model can be used within the well and lake models. Two different cases are possible to model; irrigation of a garden plot or irrigation of an agricultural field where crops are grown or cattle are grazing. In the SAFE study irrigation of a garden plot is included in the lake and well models whereas irrigation of a field has not been considered.

The radionuclides within the water are transferred to vegetation and the upper soil layer in a garden plot, see Figure 8-1. From the top soil the radionuclides can be transferred to vegetation through root uptake but they can also be transferred in the opposite direction through shedding. The radionuclides may also migrate downwards to deeper soil layers and finally reach the well water again. Transfer of radionuclides from the top soil to the fresh water occurs through erosion of the top soil. The need for irrigation is, of course, governed by the water need of the vegetation and the annual precipitation. This model simplifies that because the same large annual average amount of irrigation water is assumed to be used each year.

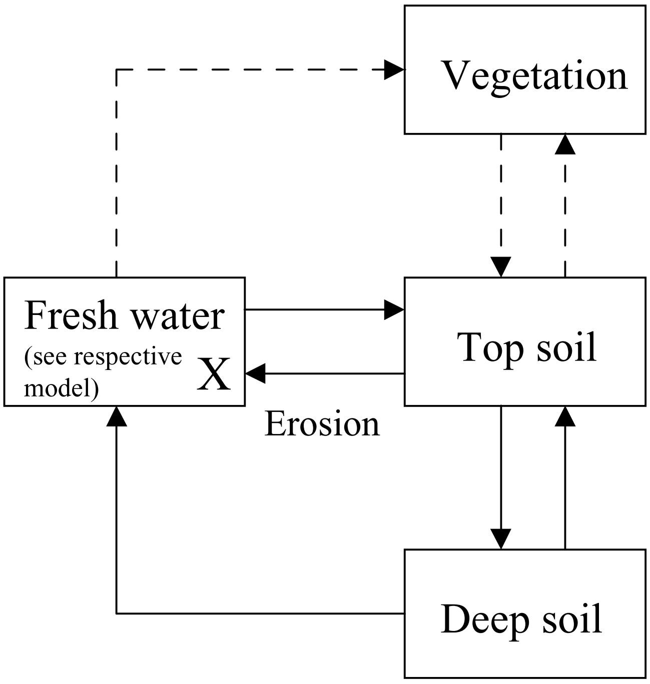


Figure 8-1. *Structure of the sub-model irrigation. The model consists of two compartments for the soil; one for the surface layer and one for the deeper layer, and one for the fresh water (including suspended matter). The vegetation part of the system (broken arrows) is not modelled in the dispersion model but is included later in the dose calculations. Transfer of radionuclides within the system is marked with arrows. The cross marks the source of radionuclides.*

The soil part of the model is comparable to the agricultural land model, i.e. the top and deep soil compartments. The outflow of radionuclides from the irrigated soil is assumed to be to the source (well or lake, respectively). The consequence of this feedback is that no radionuclides are lost from the system due to irrigation. That the groundwater zone of the soil is not included in the model is a simplification, normally drainage from soil to freshwater systems occurs as groundwater outflow. This simplification may overestimate the transfer of radionuclides from soil to surface water which is a conservative approach when dose to humans is calculated.

8.1 Irrigation

The rate constant describing the transfer of radionuclides from surface water to soil due to irrigation is obtained from the following expression:

$$TC = \frac{V_{IRR} \cdot Nr_{IRR}}{VW} \cdot A$$

where

V_{IRR} = Water amount used at each irrigation event [m^3/m^2]
Nr_{IRR} = Number of irrigation events [number/year]
VW = Volume of water from which irrigation water is taken [m^3]
A = Area of irrigated soil [m^2]

It is assumed that the suspended matter follows the water and therefore the same transfer coefficient is used for the transfer of suspended matter to soil in the lake model.

8.2 Initial retention

Retained irrigation water with its contents of radionuclides contaminates the surfaces of vegetation. This is handled as a non-continuous process where about 3 mm of water is retained on the surface of vegetation at each irrigation occasion. The 3 mm water layer is an average value based on a leaf area index (LAI, depends on e.g. the shape of the leafs) of 6 and a specific interception storage capacity of 0.5 mm/LAI /Persson, 1997/. The concentration of radionuclides on vegetation surfaces decreases due to growth and effects of wind and precipitation, which effect is described by a weathering half-life /IAEA, 1994/. A new irrigation occasion causes an additional retention while there is an exponential decrease in activity during the time passing between the irrigation occasions. The amounts of radionuclides on the surfaces of vegetation are therefore a function of irrigation frequencies and time for harvest.

The concentration of radionuclides in vegetation due to irrigation is calculated somewhat different for cereals and root crops compared to vegetables and pasturage as the later are harvested during the irrigation period whereas the former are harvested once per year. For vegetables and pasturage a mean concentration of radionuclides during the whole irrigation period is calculated. For cereals and root crops the radionuclide concentration is calculated depending on the number of irrigation occasions in relation to the time for harvest. Another important difference between these two kinds of crops is that the edible parts of cereals and root crops are not those that are exposed to irrigation water as is the case of vegetables and pasturage. Instead a transfer from vegetation surface to edible parts (grain and root respectively) is modelled using a translocation factor. For further descriptions see sections 9.1.1 and 9.1.2.

In this model no difference is made between fractions of radionuclides soluble in the irrigation water or sorbed on particles in this water. In the text above it is stated that about 3 mm of water is retained on the vegetation surfaces but it is not known if the amount of the suspended matter which is retained is proportional to this amount. If this approach underestimates or overestimates the amount of radionuclides which is retained on the vegetation surfaces is therefore not clear.

8.3 Turnover in soil

After deposition on soils, nuclides migrate with varying rates due to their physical/chemical properties and soil conditions. In addition, processes like bioturbation and erosion cause a redistribution of the radionuclides. The leakage or migration rate of nuclides from the top soil to the deep soil compartment is described by a transfer coefficient considering vertical transport due to percolation of water and transport of soil by (earth)worms. This expression is taken from another study /BIOMOVS II, 1996a/:

$$\mathrm{TC} = \frac{\mathrm{R}}{\varepsilon_t \cdot \mathrm{D}_{ts}} \cdot \mathrm{Ret} + \frac{\mathrm{BioT}}{\mathrm{D}_{ts}(1-\varepsilon_t) \cdot \rho_p}$$

where

$$\mathrm{Ret} = \frac{1}{\left[1 + \mathrm{K}_d \cdot \rho_p \cdot (1-\varepsilon_t)/\varepsilon_t\right]} \qquad \text{/Andersson et al, 1982/}$$

and

R = Runoff (Precipitation – Evapotranspiration) [$\mathrm{m^3/(m^2 \cdot year)}$]
ε_t = Porosity of soil in the top soil [$\mathrm{m^3/m^3}$]
D_{ts} = Depth of top soil [m]
BioT = Transport due to bioturbation [$\mathrm{kg/(m^2 \cdot year)}$]
ρ_p = Density of soil particles [$\mathrm{kg/m^3}$]
K_d = Distribution factor, concentration of the element on solids relative to dissolved [$\mathrm{m^3/kg}$]

The rates for bio-transport are obtained from an annual transport of soil, divided by soil masses; that is masses for top and deep soil, respectively as it is also considered as a reverse transport from deep soil to top soil:

$$TC = \frac{BioT}{D_{ds}\left(1-\varepsilon_d\right)\rho_p}$$

where

D_{ds} = Depth of deep soil layer [m], see below
ε_d = Porosity of soil in the deep soil [m^3/m^3]

$$D_{ds} = 1 - D_{ts}$$

where

D_{ts} = Depth of top soil [m]

Radionuclides in deep soil are assumed to leak back to the aquifer from which the irrigation water is taken (well or lake). This is described by the same expression as above, but without bioturbation and with depths and porosities according to deep soil.

Data on soil erosion in garden plots are not available and in order to use a conservative approach this process has not been considered in that case. When modelling irrigation of an agricultural field the loss of radionuclides from the top soil layer due to erosion is described as follows:

$$TC = \frac{Rem}{D_{ts}(1-\varepsilon_t)\cdot\rho_p}$$

where

Rem = Removal of soil [kg/(m^2·year)]
D_{ts} = Depth of top soil [m]
ε_t = Porosity of soil in the top soil (plough layer) [m^3/m^3]
ρ_p = Density of soil particles [kg/m^3]

8.4 Parameter values

Few site-specific data is used, instead the parameter values from /Bergström et al, 1999/ are adopted, see Tables 8-1 and 8-2.

Table 8-1. General data used in the irrigation sub-model when it is applied to an agricultural soil.

Irrigation of agricultural soil

Parameter	Unit	Distr	Best estimate	Min	Max	Reference
Water amount used at each irrigation event (V_{IRR})	m^3/m^2	T(log)	0.030	0.014	0.067	Estimated after /Linner, 1997/ [1]
Irrigation events (Nr_{IRR})	number per year	T	5	3	7	Estimated after /Linner, 1997/ [1]
Irrigation period, fraction of year (t_{tot})	$year^{-1}$	T	0.21 (75 days per year)	0.17	0.25	[2]
Irrigation area (A)	m^2	T	530 000	400 000	625 000	[3]
Retention of irrigation water (I)	$m^3/m^2 = m$	T	0.003	0.001	0.005	/Persson, 1997/ [4]
Runoff (R)	$m^3/(m^2 \cdot year)$	T	0.25	0.20	0.30	/Lindborg and Schüldt, 1998/ [5]
Depth of top soil layer (D_{ts})	m	T	0.25	0.20	0.30	/Haak, 1983/ [6]
Depth of top soil and deep soil layer	m	C	1.0	–	–	[7]
Soil particle density (ρ_p)	kg/m^3	T	2 650	2 600	2 700	[8]
Soil porosity,						
top soil (ε_t)	m^3/m^3	T	0.5	0.4	0.6	/Wiklander, 1976/ [9]
deeper soil (ε_d)	m^3/m^3	T	0.5	0.4	0.6	[10]
Weathering half-life ($T_{½w}$)	day	T	15	10	20	/IAEA, 1994/ [11]
Bioturbation (BioT)	$kg/(m^2 \cdot year)$	T	2	1	3	/Müller-Lemans and van Dorp, 1996/ [12]
Soil removal (Rem)	$kg/(m^2 \cdot year)$	T	0.005	0.002	0.020	[13]

[1] According to Linner, the annual water amount used for irrigation is about 0.15 m^3/m^2. In /Bergström et al, 1999/ this value was given a range of 0.1 to 0.2 m^3/m^2. Assuming that the number of irrigation events per year is 5 (3–7) the water amount used at each irrigation occasion is 0.030 m^3/m^2 varying from 0.014 m^3/m^2 (0.1 m^3/m^2/7 times) and 0.067 m^3/m^2 (0.2 m^3/m^2/3 times).

[2] The growing period in this part of Sweden is 180–200 days per year /SNA, 1992b/. If irrigation occurs it is usually applied during the first months, not close to harvest so the irrigation period has been set to maximum 90 days per year (May, June, July) and minimum about 2 months.

[3] The parameter value is taken from Table 5-1 (area of the cultivated land). The radionuclide concentrations in soil, leading further to doses are not dependent on the size of this area as it is the amount of irrigation water per m^2 which is important for the exposure to humans.

[4] This parameter was estimated from two other parameters (leaf area index and interception storage capacity, see section 9.2) for which data for only a small number of crops were available and therefore a relatively wide range is used.

[5] The range is given in /Lindborg and Schüldt, 1998/.

[6] According to /Haak, 1983/ the depth of the plough layer in agricultural soils varies between 20 and 30 cm.

[7] It is assumed that the unsaturated zone of the soil horizons has a depth of 1 m, which is a plausible value for soils in agricultural areas.

[8] In most literature an average value of 2 650 kg/m^3 is given for the density of soil particles /Hillel, 1980; Wiklander, 1976; FitzPatrick, 1980/. If the soil contains a large fraction of heavy minerals, the value will be higher while it decreases with increasing fraction of organic matter. In lack of more precise data this average value is selected with narrow ranges as it is assumed that no extreme soil types are present in the area.

[9] According to /Wiklander, 1976/ most agricultural soils have a porosity varying from 0.4 to 0.6, but the porosity can be even higher in organogenic soils.

[10] In lack of better information the same porosity as for the upper soil layer is used.

[11] The parameter range is estimated from /Hoffman and Baes, 1979/.

[12] This parameter value depends on the biological activity within the soil which can vary very much and therefore a wide range is used.

[13] The importance of erosion for transport of soil varies considerable. An investigation of erosion in southern Sweden presents losses of 0.5 to 300 tons per ha /Alström and Bergman, 1986/. A later investigation showed much lower values, from 0.001 to 80 tons per ha /Alström and Bergman Åkerman, 1991/. Erosion has also been studied as transport of suspended matter in running waters /Nilsson, 1972/. The results indicate annual losses from 10 up to hundreds of kg per ha. When the parameter range was estimated the extreme values were excluded.

Table 8-2. General data used in the irrigation sub-model when it is applied to a garden plot. All values are assumed to be triangularly distributed.

Irrigation of garden plot

Parameter	**Unit**	**Distr**	**Best estimate**	**Min**	**Max**	**Reference**
Water amount used at each irrigation event (V_{IRR})	m^3/m^2	T(log)	0.030	0.014	0.067	[1]
Irrigation events (Nr_{IRR})	number per year	T	10	6	14	[2]
Irrigation period, fraction of year (t_{tot})	$year^{-1}$	T	0.21 (75 days per year)	0.17	0.25	[3]
Irrigation area (A)	m^2	T	200	150	250	[4]
Retention of irrigation water (I)	$m^3/m^2 = m$	T	0.003	0.001	0.005	/Persson, 1997/ [5]
Runoff (R)	$m^3/(m^2 \cdot year)$	T	0.25	0.20	0.30	/Lindborg and Schüldt, 1998/ [6]
Depth of top layer (D_{ts})	m	T	0.25	0.20	0.30	/Haak, 1983/ [7]
Soil particle density (ρ_p)	kg/m^3	T	2 650	2 600	2 700	[8]
Soil porosity,						
top soil (ε_t)	m^3/m^3	T	0.5	0.4	0.6	/Wiklander, 1976/ [9]
deeper soil (ε_d)	m^3/m^3	T	0.5	0.4	0.6	[10]
Weathering half-life ($T_{½w}$)	days	T	15	10	20	/IAEA, 1994/ [11]
Bioturbation (BioT)	kg/$(m^2 \cdot year)$	T	2	1	3	/Müller-Lemans and van Dorp, 1996/ [12]

[1] In lack of data the same amounts as for an agricultural field is used, see Table 8-1.

[2] It was assumed that garden plots are irrigated more frequently than agricultural fields. A number of 10 irrigation events per year equals about one irrigation event per week during the summer period when garden plots are in use. A quite wide range is set since the number of events varies with crop as well as prevailing weather conditions.

[3] Cultivation in garden plots takes place during the summer months (second part of May, June, July and the first part of August) and irrigation was assumed to take place during this time. A maximum irrigation period of 90 days per year is therefore used and a minimum of 2 months.

[4] This area should be of sufficient size to produce the annual amount of foodstuff for a family. The calculated doses are not dependent on the size of this area as it is the amount of irrigation water per m^2 which is important for the exposure to humans.

[5] This parameter was estimated from two other parameters (leaf area index and interception storage capacity, see section 9.2) for which data for only a small number of crops were available and therefore a relatively wide range is used.

[6] The range is given in /Lindborg and Schüldt,1998/.

[7] According to /Haak,1983/ the depth of the plough layer in agricultural soils varies between 20 and 30 cm.

[8] In most literature an average value of 2 650 kg/m^3 is given for the density of soil particles /Hillel, 1980; Wiklander, 1976; FitzPatrick, 1980/. If the soil contains a large fraction of heavy minerals, the value will be higher while it decreases with increasing fraction of organic matter. In lack of more precise data this average value is selected with narrow ranges as it is assumed that no extreme soil types are present in the area.

[9] According to /Wiklander, 1976/ most agricultural soils have a porosity varying from 0.4 to 0.6, but the porosity can be even higher in organogenic soils.

[10] In lack of better information the same porosity as for the upper soil layer is used.

[11] The parameter range is estimated from /Hoffman and Baes, 1979/.

12 This parameter value depends on the biological activity within the soils which can vary very much and therefore a wide range is used.

8.5 Uptake in biota and exposure pathways

The exposure pathways added when irrigation is considered in the lake and well models can be seen in sections 4.5 and 7.3, respectively.

Surface deposited radionuclides can be translocated from surfaces of vegetation to the edible parts. This is described by an element specific translocation factor (see Table A-6 in Appendix A). The factor expresses the concentration of radionuclides (Bq/kg wet weight) in edible parts of the crop compared to the amount deposited on the crop surface (Bq/m^2).

For cesium and strontium it has been shown that the concentration in the edible parts due to translocation is strongly related to the time period between contamination and harvest /Aarkrog, 1994/. Translocation has been included for root crops and cereals in this model.

For vegetables the retained radionuclides on the vegetative surface are consumed by man and no translocation factor is used. The same is also true for pasturage consumed by cattle.

The expressions used for dose calculations are further described in Chapter 9.

9 Methods for calculation of doses to humans

Humans can be exposed externally as well as internally from radiation, see Figure 9-1. **External exposure** may occur from ground deposition of radionuclides or immersion in contaminated air or water. However, results from earlier safety analyses show that the only external exposure contributing significantly to the total dose is from contaminated ground /Bergman et al, 1977, 1979; Bergström, 1983/. Hence this is, the only external exposure pathway considered in this study.

Inhalation and intake of radionuclides lead to **internal exposure**. Nuclides in soil, water and sediment may be taken up by living organisms and be further transported along the food-chains to man. The type of ecosystem, to which radionuclides enter, determines which exposure pathways are to be considered. Vegetation is contaminated by two main pathways, root-uptake and retention of radionuclides on surfaces due to deposition or irrigation. The radionuclides deposited on vegetation surfaces can be transferred to edible parts by translocation.

The assumed human diet in this study consists of meat, fish, vegetables, root crops, cereals and milk. All food is produced locally in the area where the radionuclides from the repository enter the biosphere. This is not the general situation in Sweden today as fractions of the food usually is produced in other parts of the country as well as abroad. For the dose calculations this approach is thus conservative as "dilution" with uncontaminated food does not occur. The consumption rates used are not extreme,

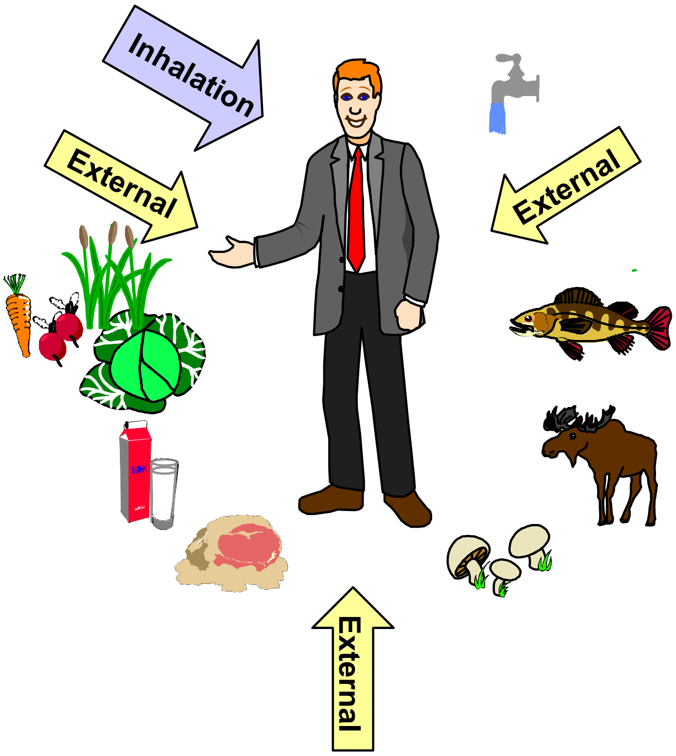


Figure 9-1. Potential exposure pathways to man; internal by inhalation and consumption of water and food; external from radionuclides in air and on the ground.

instead an estimated average diet has been used (see Table 9-3). The values have been given normal distributions with a standard deviation of 10 %. It was assumed that beef represents the total annual amount of meat consumed by humans. The consumption of milk includes all milk products such as butter and cream. Consumption of entrails as kidney and liver are excluded.

Radiation doses from different exposure pathways are calculated based upon concentrations of each radionuclide in relevant compartments in the dispersion models described earlier. The method for dose calculations follows the principles recommended by IAEA for radiological impact assessment /IAEA, 1994/. Here follow the equations for the dose calculations. Parameter values used are presented in tables following the text.

9.1 Ingestion

Doses via ingestion (D_{ing}) are calculated using the concentration in the food item, the consumption rate and element specific dose coefficients for ingestion:

$$D_{ing} = HC_i \cdot U_i \cdot DC_{ing}$$

where

HC_i = Consumption rate for pathway i [kg or litre per year], see Table 9-3

U_i = Concentration in foodstuff i [Bq per kg or litre] (expressions given below)

DC_{ing} = Dose coefficients for ingestion [Sv/Bq] according to /EU, 1996/, see Table A-1

Human consumption of agricultural products is represented by five groups of food items, i.e. milk, meat, vegetables, root crops and cereals. Fish represent food from aquatic systems. Consumption of algae and fresh water crustacean is also possible to consider even though this is not adequate for the studied area. For each of these food items the concentration of radionuclides is calculated as follows.

9.1.1 Milk and meat

Transfer of radionuclides to milk and meat is based on cattle's intake of contaminated fodder, soil and water. The concentration in milk (U_{milk}) and meat (U_{meat}) is obtained from:

$$U_{milk} = In \cdot F_{milk}$$

$$U_{meat} = In \cdot F_{meat}$$

where

In = Daily intake of nuclides [Bq/day] (expression given below)

F = Element specific transfer coefficient for milk and meat, respectively [day/litre, day/kg], see Tables A-11 and A-12

Cattle are assumed to eat three different kinds of fodder; concentrated fodder (here represented by cereals), grass (fresh when grazing on the pasturage and as hay, harvested from the pasturage, the rest of the year, here combined and simply called pasturage) and plants growing on the shore of a lake or watercourse (hereafter called water plants). Additionally, some inadvertent consumption of soil when grazing is assumed. Radionuclides may also be intaken through consumption of contaminated water. Pasturage and cereals are contaminated through root uptake and retention of radionuclides on vegetation surfaces whereas water plants are contaminated through their uptake and transpiration of contaminated water. Different uptake pathways are considered in the different models, see Table 9-1. The cattle's daily intake of radionuclides (In) is calculated as a sum of products:

$$In = MC_x \cdot UC_x \ \ldots . + MC_y \cdot UC_y$$

where

$MC_{x/y}$ = Daily consumption of food item, water and soil, respectively [kg or litre per day], see Table 9-2

$UC_{x/y}$ = Concentration of radionuclides in food item, water and soil, respectively [Bq per kg or litre] (expressions given below)

The concentrations of radionuclides in water and soil are obtained from the dispersion models (Chapters 3–8). The concentration of radionuclides in cereals that are used as concentrated fodder to cattle is assumed to have the same concentration as the cereals for human consumption, which is obtained according to the expression given below (see section 9.1.2). In the agricultural land and mire models it is assumed that pasturage is grown on contaminated areas and radionuclides are transferred to the grass via root uptake. The concentration in pasturage (UC_p) is calculated as:

$$UC_p = C_i \cdot RUF_p$$

where

C_i = Concentration of radionuclides in soil or peat [Bq/kg], from the dispersion model

RUF_p = Root uptake factor for pasturage [(Bq/kg dw)/(Bq/kg dw)], see Table A-2

Table 9-1. An overview of considered pathways (X) for intake of radionuclides in cattle in the different models.

Model	Water	Pasturage	Cereals	Water plants	Soil
Coastal	X			X	
Lake	X			X	
Agricultural land		X	X		X
Mire		X	X		X
Well	X				

The consumption of water plants from a contaminated recipient (UC_{wp}) does also contribute to the contamination of milk and meat. As the cattle is assumed to graze only part of the year the uptake of radionuclides via this pathway depends on, among other things, the number of days that they spend on the shore line:

$$UC_{wp} = \frac{CG}{365} \cdot \frac{C_w \cdot TR \cdot GD \cdot 10^{-6} \cdot 24}{Y_{wp}}$$

where

CG = Cattle grazing period on shores [days/year], see Table 9-2

C_w = Concentration of radionuclides in water [Bq/m^3] (radionuclides in suspended matter included), from the dispersion model

TR = Transpiration of water plants [$g/(m^2 \cdot h)$], see Table 9-2

GD = Average time for water plant transpiration before animal's consumption [days/year], see Table 9-2

Y_{wp} = Annual production of water plants [$kg/m^2 \cdot year$], see Table 9-2

10^{-6} = m^3/g

24 = h/day

365 = days/year

9.1.2 Crops

Crops are represented by cereals, root crops and green vegetables. In the agricultural land and mire models it is assumed that these crops are grown on contaminated soil or peat and radionuclides are transferred to the vegetation via root uptake. The concentration in the crops (U_i) is calculated as:

$$U_i = C_i \cdot RUF_i$$

where

C_i = Concentration of radionuclides in soil or peat [Bq/kg dw], from dispersion model

RUF_i = Root uptake factor for crop i (see below)[(Bq/kg dw)/(Bq/kg ww)], see Table A-3–A-5

i = c for cereals
i = r for root crops
i = v for vegetables

In other models, i.e. the lake and well models, contamination may occur through irrigation with contaminated water. In the SAFE study irrigation of a garden plot where root crops and vegetables are grown are simulated. Root crops are then contaminated via root uptake as well as retention of radionuclides on vegetation surfaces and thereafter translocation to edible parts. As more radionuclides are supplied at each irrigation occasion the radionuclide concentration depends on the number of irrigation events. Root crops are harvested after the irrigation period and the radionuclides supplied during the season are added up. The expression for resulting concentration in root crop products (U_i) is:

$$U_i = C_s \cdot RUF_R + \sum_{o}^{Nr_{IRR}} I \cdot TL \cdot C_w$$

where

C_s = Concentration of radionuclides in soil [Bq/kg dw] from the dispersion model

RUF_R = Root uptake factors for root crops (soil to plant transfer factors) [(Bq/kg ww)/Bq/kg dw)], see Table A-4

Nr_{IRR} = Number of irrigation occasions [year^{-1}], see Tables 8-1 and 8-2

I = Remaining water on the vegetation after each irrigation occasion [m^3/m^2 = m], see Table 8-1

TL = Translocation from plant surface to edible parts of plant, [(Bq/kg ww)/(Bq/m^2)], see Table A-6

C_w = Concentration of radionuclides in irrigation water [Bq/m^3], from the dispersion model

Vegetables are also contaminated from root uptake and surface contamination due to retention of contaminated irrigation water. The harvest of green vegetables is assumed to occur during the whole growing period (180–200 days per year /SNA, 1992b/). Therefore the mean concentration of surface contamination during the period is calculated and used in the dose calculations. The expression for the resulting content of radionuclides in vegetables (U_v) is:

$$U_v = C_s \cdot RUF_v + \frac{C_w}{Y_v} \cdot \frac{I}{t_{tot}} \cdot \sum_{Nr_{IRR}} \int_0^{t_n} e^{-\tau \cdot t_n} dt$$

where

C_s = Concentration of radionuclides in soil [Bq/kg dw], from the dispersion model

RUF_v = Root uptake factor for vegetables (soil to plant transfer factor) [(Bq/kg ww)/Bq/kg dw)], see Table A-5

C_w = Concentration of radionuclides in irrigation water [Bq/m^3], from dispersion model

Y_v = Yield of vegetables [kg/m^2·year], see Table 9-5

I = Remaining water on the vegetation after each irrigation occasion [m^3/m^2 = m], see Tables 8-1 and 8-2

t_{tot} = Irrigation period, fraction of year [year^{-1}], see Tables 8-1 and 8-2

Nr_{IRR} = Number of irrigation occasions per year [year^{-1}], see Tables 8-1 and 8-2

t_n = Time between last irrigation occasion and harvest [days]

τ = $\ln 2/T_{½w}$ where $T_{½w}$ = weathering half-life [day], see Tables 8-1 and 8-2

9.1.3 Food from aquatic systems

Food from aquatic systems is represented by fish (algae and crustaceans can also be used). Concentrations of radionuclides in those organisms are obtained by use of bioaccumulation factors for edible parts of the species relative to the total concentration in the water (i.e. the amounts of radionuclides in water and on suspended matter). In order to use available databases which often are based on empirical concentration ratios, the water includes here the radionuclide fraction associated with suspended matter (which usually is a minor fraction). The bioaccumulation factors are valid for steady-state conditions and implicitly consider all uptake paths from the ambient environment. The expression for estimating the content of radionuclides in fish, crustaceans and algae (U_i) is:

$$U_i = BAF_i \cdot C_w$$

where

BAF_i = Element specific bioaccumulation factor from water to edible parts of species i [(Bq/kg ww)/(Bq/l)], see Tables A-13 to A-15

i = f for fish
i = c for crustacean
i = a for algae

C_w = Concentration of radionuclides dissolved and suspended in the ambient water [Bq/l], from the dispersion model

9.2 Inhalation

Doses via inhalation (D_{inh}) are calculated using the concentration of radionuclides in the air, the inhalation rate, the exposure time and element specific dose coefficients for inhalation:

$$D_{inh} = C_a \cdot IH \cdot H_i \cdot DC_{inh}$$

where

C_a = Concentration of radionuclides in air [Bq/m^3], see expressions below

IH = Inhalation rate [m^3/h], see Table 9-3

H_i = Exposure time [h/year], see Table 9-3

DC_{inh} = Dose coefficients for inhalation [Sv/Bq] according to /EU, 1996/, see Table A-1

Radionuclides in air emanate from three sources; dust in air from soil or peat resuspension and flue gases produced when peat is used as fuel in a household. Particles in air are assumed to have the same radionuclide concentration as soil. The concentration of radionuclides in air due to contaminated soil resuspension ($C_{a,s}$), peat resuspension ($C_{a,p}$) and contaminated peat combustion ($C_{a,f}$) is expressed by:

$$C_{a,s} = C_s \cdot S$$

$$C_{a,p} = C_p \cdot S$$

$$C_{a,f} = C_p \cdot FC \cdot RC \cdot FE$$

where

C_s = Concentration of radionuclides in soil [Bq/kg dw], from the dispersion model

S = Dust content in air [kg/m^3], see Table 9-5

C_p = Concentration of radionuclides in peat [Bq/kg dw], from the dispersion model

FC = Fuel load [kg dw/s], see Table 9-4

RC = Relative concentration as an annual mean value [s/m^3], see Table 9-4

FE = Fraction of nuclides which leaves the combustion apparatus via exhaust gases (efficiency of the filter system), see Table 9-4

9.3 External exposure

Doses via external exposure from soil or mire (D_{ext}) are calculated using the concentration of radionuclides in soil/peat, the soil/peat density, the exposure time and element specific dose coefficients for external exposure:

$$D_{ext} = C_i \cdot \rho_i \cdot H_i \cdot DC_{ext}$$

where

C_i = Concentration of radionuclides in soil or peat [Bq/kg dw], from the dispersion model

ρ_i = Soil or peat density [kg/m^3], see Table 5-1/6-1

H_i = Exposure time [h/year], see Table 9-3

DC_{ext} = External dose coefficients [(Sv/h)/(Bq/m^3)], see Table A-1

9.4 Data

The consumption rates used for cattle and man are shown in Tables 9-2 and 9-3. In addition, inhalation rates and exposure times are shown in Table 9-3. Parameter values used for dose calculations for combustion of peat can be seen in Table 9-4. Other generic data for the dose calculations are summarised in Table 9-5.

Table 9-2. Data to calculate cattle's intake of radionuclides.

Parameter	Unit	Distr	Best estimate	Min	Max	Reference
Water consumption (C_w)	l/day	T	70	65	75	/Morén, 2000/ [1]
Consumption of pasturage/water plants ($C_{p/wp}$)	kg dw/day	T	8.5	8	9	/Morén, 2000/ [2]
Consumption of cereals (concentrated food) (C_c)	kg ww/day	T	11	10	12	/Morén, 2000/ [3]
Soil consumption (C_s)	kg dw/day	T	0.3	0.15	0.5	[4]
Transpiration of water plants (TR) [5]	g/(m^2·h)	T(log)	100	50	300	/Jörgensen et al, 1991/, Table 1-392
Average time for water plant transpiration before cattle's consumption (GD) [5]	days/year	T	100	80	120	[6]
Annual production of water plants (Y_{wp}):						
coastal model	kg/(m^2·year)	T	0.3	0.1	0.5	[7]
fresh water models	kg/(m^2·year)	T	0.3	0.1	0.5	[8]
Shore grazing period	days/year	T	30	20	40	[9]

[1] The values given are for dairy cows, water consumption of beef cattle are much lower (20–60 l/day).

[2] According to Morén cattle need 8–9 kg food (in dry weight) each day together with the concentrated food.

[3] According to Morén milk producing cattle consume 10 to 12 kg of concentrated food each day during the producing period (6 to 7 months per year). To assume that this amount is consumed during the whole year is a somewhat conservative assumption.

[4] The consumption of soil adhering to "plant food" was calculated by /Davis et al, 1993/ and the value is extracted from that study. A relatively wide parameter range is set because of the difficulties to verify such data.

[5] In brackish as well as fresh water.

[6] It is here assumed that the plant transpiration starts early in the year when light gets more abundant, i.e. in March and that outdoor grazing for cattle starts in May or June. It is not possible to verify these data within this study so a relatively wide parameter range (20 %) is used.

[7] Data for the annual production of plants on the shore is not available, instead the production value for pasturage in the region is set as the maximum level. The primary production of macrophytes in the Model Area given in /Kumblad, 1999/ is used as a minimum value since the production of plants growing on the shores is larger than the production of these species off shore.

[8] In lack of data the same values are used as for the coastal model.

[9] Part of the time which cattle spend outdoors are assumed to be spent near shores of a coastal bay, a lake or a watercourse. During the remaining time it is assumed that grazing occurs elsewhere. A relatively wide parameter range is used since this varies a lot depending on e.g. the shore line length and the production of shore plants.

Table 9-3. Human consumption rates, external exposure rates and inhalation rates used in the dose calculations.

Parameter	Distr[1]	Unit	Best estimate	Std		Reference
Water (HC_w)	N	l/year	600	10 %[2]		/Bergström and Nordlinder, 1990b/
Milk (HC_{mi})	N	l/year	300	10 %[2]		[3, 4]
Meat (HC_{me})	N	kg/year	70	10 %[2]		[3]
Vegetables (HC_v)	N	kg/year	60	10 %[2]		[3]
Root crops (HC_r)	N	kg/year	70	10 %[2]		/Bergström and Nordlinder, 1990b/
Cereals (HC_c)	N	kg/year	80	10 %[2]		/Bergström and Nordlinder, 1990b/
Soil (HC_s)	N	kg/year	0.1	10 %[2]		[5]
Fish (HC_f)	N	kg/year	30	10 %[2]		/Bergström and Nordlinder, 1990b/
Exposure time (H_i)[6, 7]	T	h/year	100	50	150	[8]
Inhalation rate (IH)[6]	T	m^3/h	1	0.8	1.2	/ICRP, 1974/[9]

[1] Type of distribution: N = normal distribution, T = triangular distribution.

[2] No extreme diets are considered in this study so a relatively narrow range is used.

[3] Information from the Swedish Board of Agriculture (www.sjv.se) September, 2000.

[4] This parameter value includes the consumption of all milk products, e.g. milk, cheese and butter. According to the Swedish Board of Agriculture the average consumption of milk and cheese for adults in Sweden is 115 l/year and 16.5 kg/year respectively. According to Arla (Swedish milk producers) (www.arla.se) it takes 10 l of milk to produce 1 kg cheese.

[5] Extracted from /Davis et al, 1993/.

[6] Triangularly distributed values; best estimate, min and max values.

[7] Time of exposure from contaminated agricultural land or garden plot, i.e. time for external exposure from soil, and for inhalation doses' duration in contaminated air.

[8] Values estimated for work with soil i.e. agricultural practices. The same time is used for work on an agricultural field as for work in a garden plot. A garden plot is visited much more often then a field but the time spent on each occasion is much shorter. When working on a field a vehicle is used which should decrease the external exposure. The range used is relatively wide as such information is hard to evaluate.

[9] In lack of data a variation of 20 % is used.

Table 9-4. Parameter values used in the dose calculations for combustion of peat.

Parameter	Unit	Distr [1]	Best estimate	Min	Max	Reference
Fuel load (FC)	kg dw/s	T	$1.0 \cdot 10^{-4}$ (0.4 kg dw/h)	$0.5 \cdot 10^{-4}$ (0.2 kg dw/h)	$2.0 \cdot 10^{-4}$ (0.8 kg dw/h)	[2]
Filter efficiency (FE)	–	C	1			Not considered
Relative concentration (RC)	seconds/(m^3)	T(log)	$1 \cdot 10^{-5}$	$3 \cdot 10^{-6}$	$3 \cdot 10^{-5}$	/Widemo and Gyllander, 1979/ [3]
Exposure time (H_i)	h/year	T	8 000	6 500	8 760	[4]

[1] T = Triangular distribution, T(log) = Logtriangular distribution, C = Constant

[2] This parameter gives the amount of peat combusted per time unit. An average value for energy content in peat is 22 MJ/kg (R Samuelsson, Swedish University of Agricultural Sciences in Umeå, personal communication). In general 20 000 kWh/year is used for heating of a family-house (personal experience). If distributing this evenly over the year, about 0.4 kg peat is used per hour. This value is selected as mean value. The minimum value is set to half of that and the upper to twice the value.

[3] This parameter is a factor describing the dispersion at atmospheric releases. It varies with i.a. chimney height, meteorological conditions and distance from source. In this model these conditions are not known and a relatively wide range is therefore used.

[4] In lack of better data very conservative parameter values and a relatively wide range are used.

Table 9-5. Data for common parameters in the models used in the dose calculations. All values are assumed to be triangularly distributed.

Parameter	Unit	Best estimate	Min	Max	Reference
Yield of pasturage (Y_p)	kg dw/($m^2 \cdot$year)	0.5	0.4	0.6	/Haak, 1983/ [1]
Yield of cereals (Y_c)	kg dw/(m^2 year)	0.5	0.4	0.6	[2]
Yield of root crops (Y_r)	kg dw/($m^2 \cdot$year)	2.3	1.8	2.8	[3]
Yield of vegetables (Y_v)	kg dw/($m^2 \cdot$year)	2	1.5	4	[4]
Dust concentration in air (S)	kg/m^3	$1 \cdot 10^{-4}$	$3 \cdot 10^{-5}$	$3 \cdot 10^{-4}$	/Haak, 1983/ [5]

[1] According to /Haak, 1983/ the lowest production value is for pasturage in the area ("betesvall") whereas the highest value is for agricultural land where hay is harvested ("slåttervall").

[2] The mean value selected is based on reported standard norm harvests for Uppsala county 1998 /SCB, 1999/. The mean value for 5 crops 1998 was 0.474 kg dw/$m^2 \cdot$year which was rounded to 0.5. The range is set to include the inter-annual variations given, i.e. up to 10–15 %.

[3] No information on yield sizes of potatoes was found for Uppsala county. Standard yields were however calculated for the area 1998 /SCB, 1999/. This value is used with about 20 % variation. The yield of carrots is higher per surface area but the total production is considerably lower, about 10 % of that for potatoes. In addition much of carrots are used as fodder and not for human consumption.

[4] The selected values are from 1996 /SCB, 1999/. The mean value is for lettuce (1.9 kg dw/$m^2 \cdot$year rounded to 2 kg dw/$m^2 \cdot$year). The range is set to include the values for cauliflower (about 1.5 kg dw/$m^2 \cdot$year) and white cabbage (about 4 kg dw/$m^2 \cdot$year).

[5] This parameter varies with i.a. distance from source and meteorological conditions. In this model these conditions are not known and a relatively wide range is therefore used.

10 Effects of different sorption properties and contamination pathways

10.1 Coastal model

In order to obtain a rough estimate of the fate and distribution of radionuclides with various sorption properties, the coastal model was run deterministically using different K_d-values and a long physical half-life. The radionuclide concentrations in water, suspended matter and upper sediments over time when simulating a constant release of 10 000 Bq/y over a period of 1 000 years are presented in Figure 10-1 for long-lived radionuclides with K_d-values of 1 and 1 000 m^3/kg, respectively. For all model parts (and for both K_d-values) except the water in oceans, an equilibrium is established almost immediately. In Figures 10-2 to 10-5) the concentrations, inventories and fluxes within and between the different parts of the system are shown for radionuclides with different K_d-values, after simulation of a constant release of 10 000 Bq/y after a period of 1 000 years.

As can be seen in Figures 10-2 to 10-5, mobile elements (low K_d-values) leave Öregrundsgrepen to a larger extent than elements with higher K_d-values, since the former elements have a lower inclination to sorb to settling suspended matter. Similarly, the accumulation of these elements somewhere else in the Baltic is less likely at low K_d-values, since the fraction of the elements leaving Öregrundsgrepen is diluted in the large water volume of the Baltic Sea (about 200 000 km^3) or sorbed in the large amount of sediment. Even after reaching a steady state, the concentration of these elements will decrease with the distance from the outflow point due to dilution with clean waters and suspended matter entering the Baltic. When comparing pools and sinks rather than concentrations, the fraction found in sediments is as expected highest for elements that are prone to sorb to solid matter.

The dose calculations in the coastal model are basically related to the concentration of radionuclides in the water, which includes the often minor fraction of radionuclides associated with suspended matter, since the concentrations in food-stuff (fish, algae and crustaceans) are calculated from empirical bioaccumulation factors based on the total concentration of radionuclides in the water. As can be seen in the figures the concentration of radionuclides in the water of the Model Area and Öregrundsgrepen are of the same order of magnitude even if the K_d-values vary.

The fractions present within the sediments are instead of more interest in a following step when the sediments are assumed to be drained and used as agricultural land. Then these radionuclides can be taken up in crop and reach cattle and humans. The solubility is then, of course also dependent on the element's behaviour in different media. Elements with a strong tendency to sorb to solid matter in the marine environment can be expected to show the same behaviour in soil, even if the K_d-value may be different in absolute terms. Accordingly, even if large amounts of those elements can be expected to be present within the soil after the transformation from marine sediment to soils or freshwater sediment, this will not necessarily lead to high concentrations in crop as K_d is inversely correlated with root uptake. Exposure through inhalation may be of concern anyway.

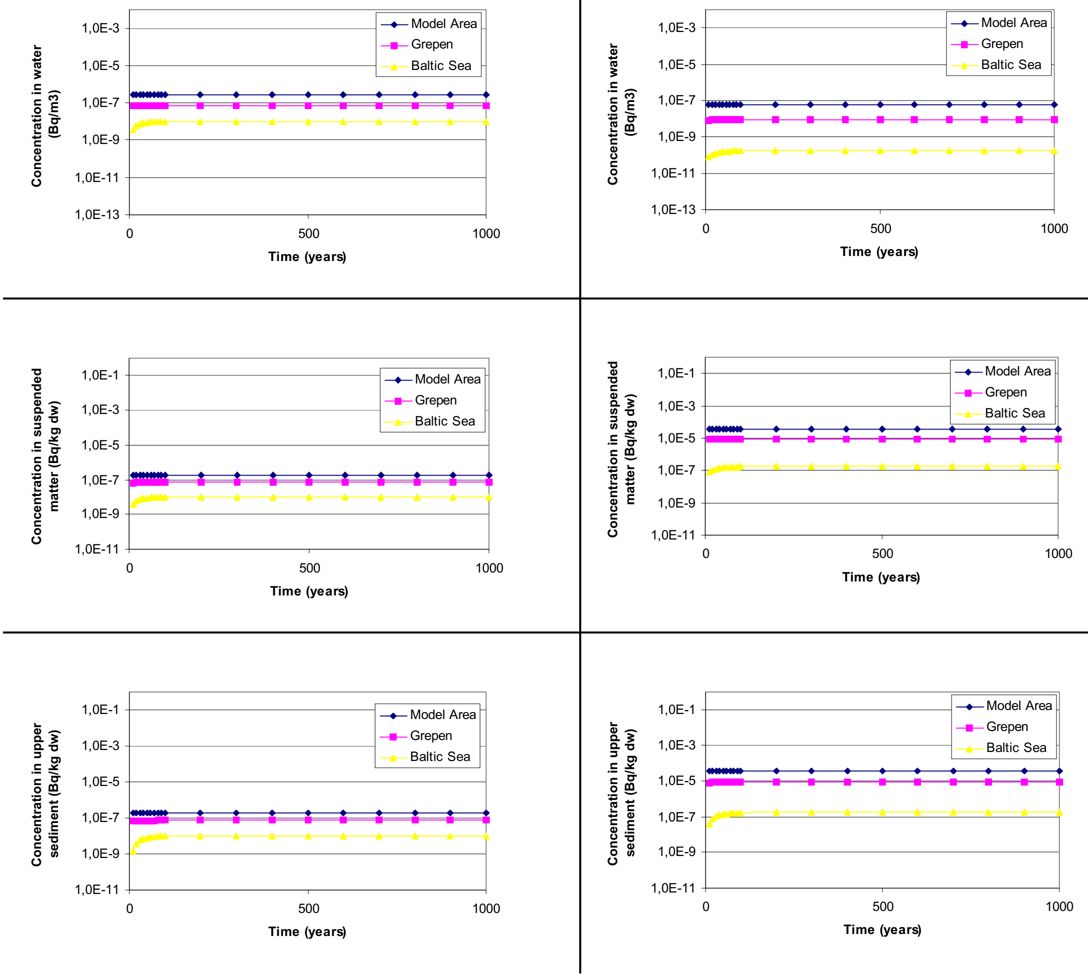


Figure 10-1. *Radionuclide concentrations in water (Bq/m3, (dissolved)), suspended matter (Bq/kg dw) and upper sediment (Bq/kg dw) of the coastal model as a function of time when simulating a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 1 m^3/kg (left) and 1 000 m^3/kg (right).*

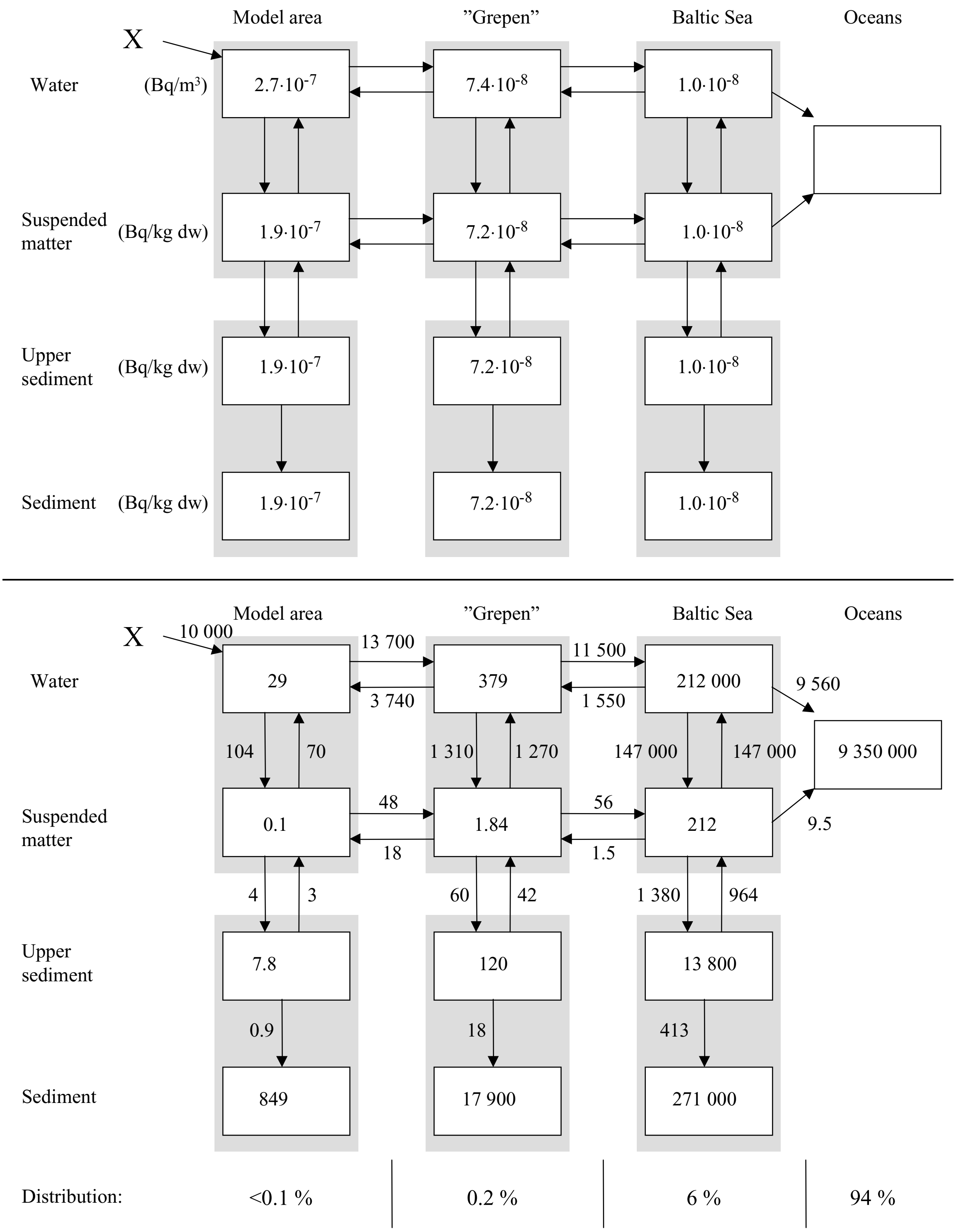


Figure 10-2. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the coastal model at the end of a 1 000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 1 m^3/kg. The concentrations in the sediments are the same as in the upper sediment layer and the suspended matter since no mineralisation or other dilution processes are considered. No concentration is given for the Ocean compartment since data about the recipient is not of interest in this case.*

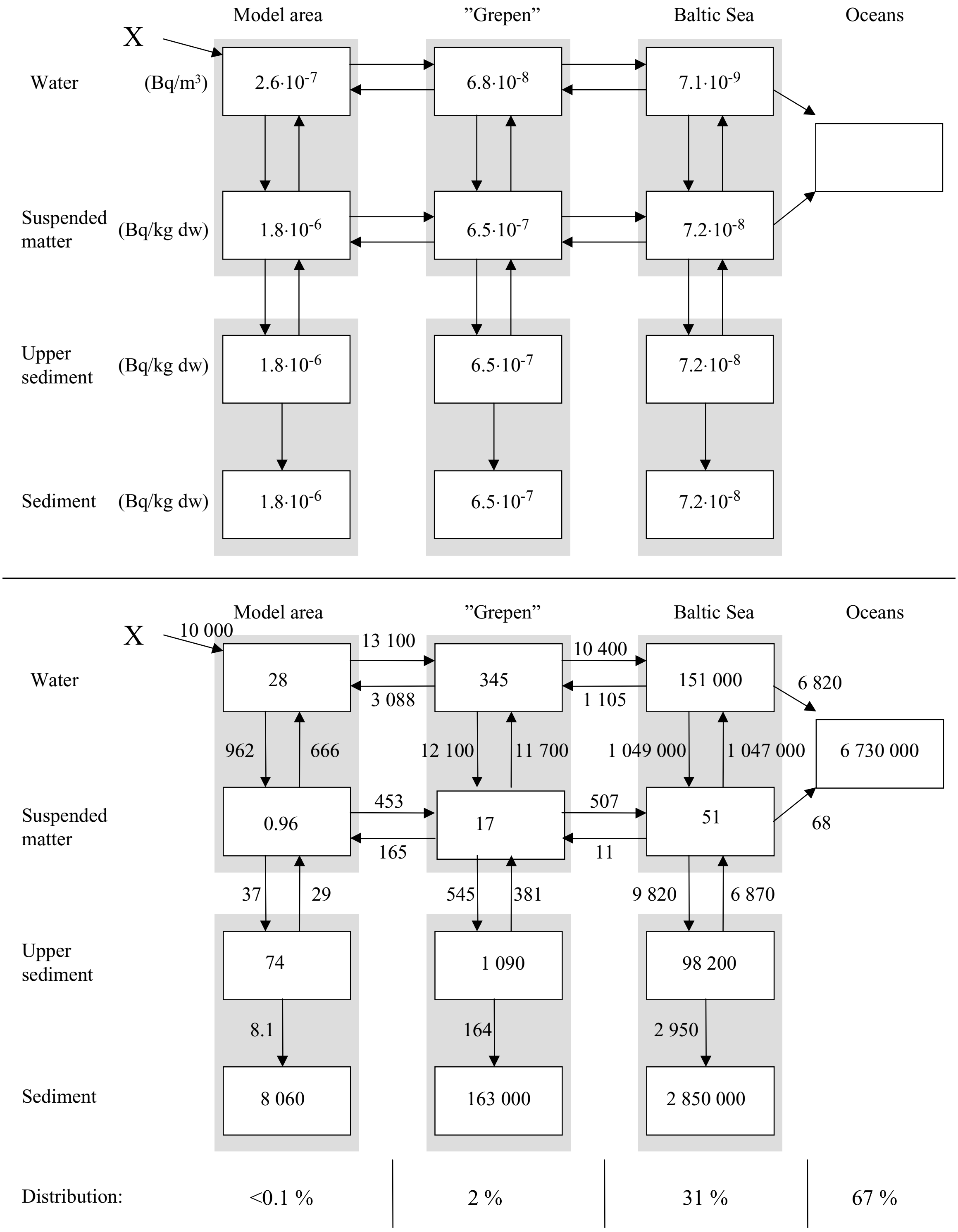


Figure 10-3. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the coastal model at the end of a 1 000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 10 m^3/kg. The concentrations in the sediments are the same as in the upper sediment layer and the suspended matter since no mineralisation or other dilution processes are considered.*

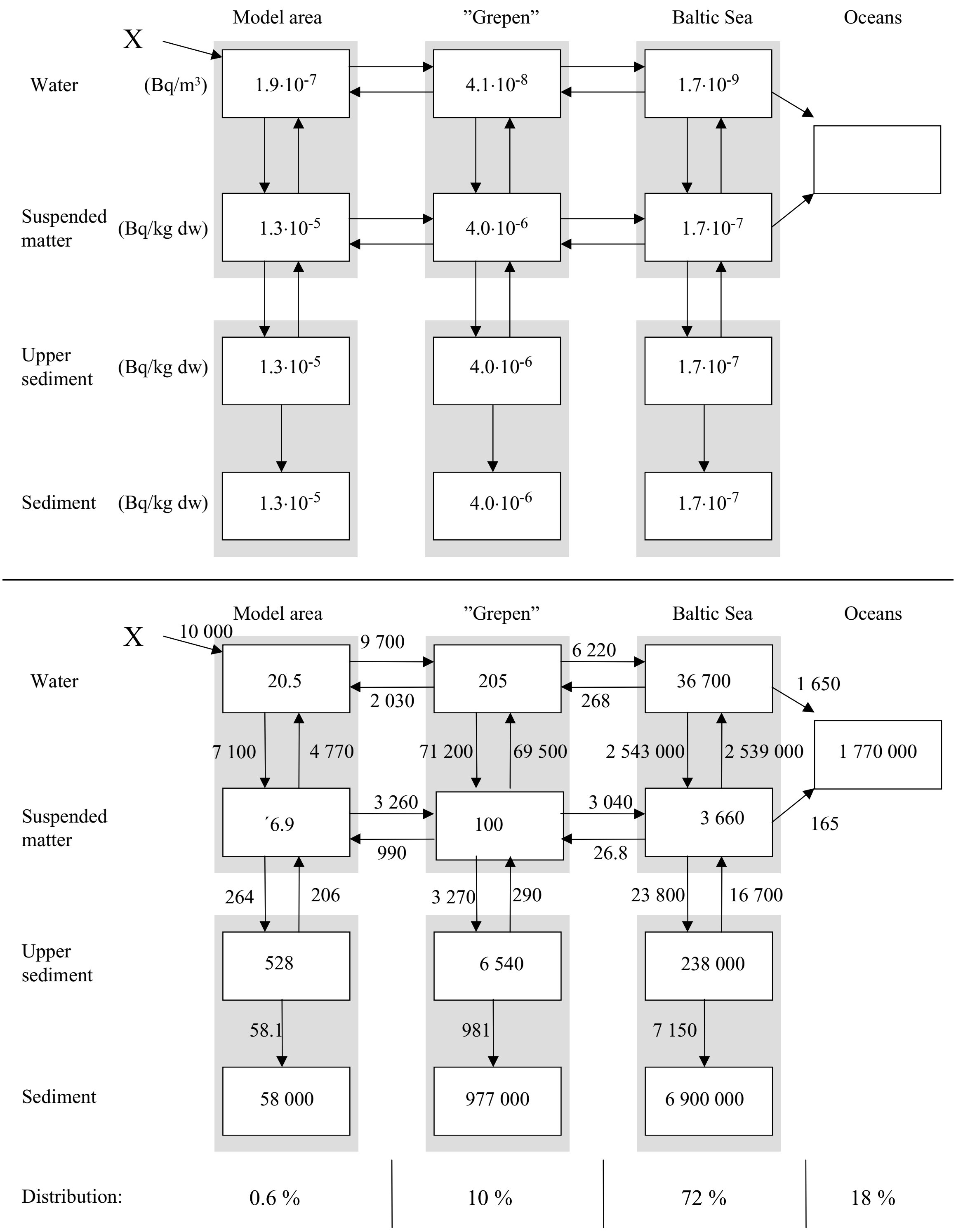


Figure 10-4. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the coastal model at the end of a 1000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 100 m^3/kg. The concentrations in the sediments are the same as in the upper sediment layer and the suspended matter since no mineralisation or other dilution processes are considered.*

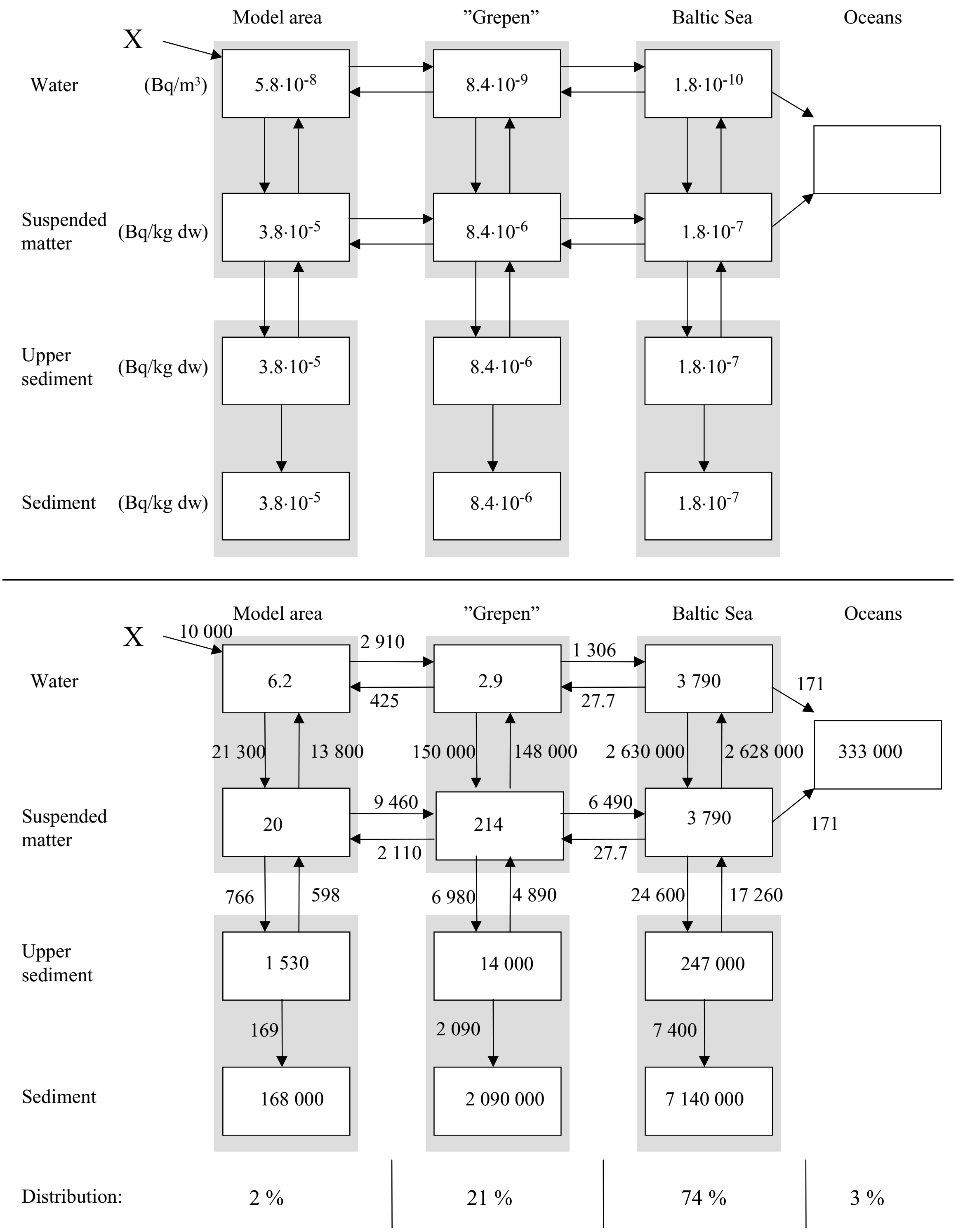


Figure 10-5. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the coastal model at the end of a 1 000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 1 000 m^3/kg. The concentrations in the sediments are the same as in the upper sediment layer and the suspended matter since no mineralisation or other dilution processes are considered.*

10.2 Lake model

The distribution of radionuclides within the lake model was examined at the end of a simulation of a constant release of 10 000 Bq/y to the lake water during a period of 1 000 years. As with the coastal model (see above) four different K_d-values were used (1, 10, 100 and 1 000 m^3/kg). As can be seen in Figures 10-6 and 10-7 there was no dramatic change in water inventory/concentration when increasing the K_d-value until a value of 1 000 m^3/kg was used. When increasing the K_d-value from 100 to 1 000 m^3/kg the dissolved inventory decreased about six-fold, while the particulate fraction almost doubled. The fraction of radionuclides sorbing to suspended matter increased with increasing K_d and as a consequence the fraction of the radionuclides leaving the lake sorbed to particles compared to those leaving with the water also increased. With a K_d-value of 1 000 m^3/kg the outflowing fraction associated with suspended matter is about twice the size of the dissolved fraction leaving the system.

The major sinks (inventories) in the system are shifted with increasing K_d from the outflow (low K_d) to the sediment (high K_d). It should be noted that upward transfer from the deeper sediment layer was not modelled specifically in this case, but is accounted if the flux from surface to deeper sediments is considered as a net flux, i.e. the difference of gross burial and remobilisation. The opposite, a net upward flux from deep sediments, was tested separately (see section 10.2.1).

The change in concentrations in the different model parts over time is presented in Figure 10-8. For all model parts (and for both K_d-values), an equilibrium is established almost immediately.

In the above mentioned test simulations of the lake model the sub-model irrigation was not included since the fraction removed from the system via the irrigation water is very small and insignificant. When modelling irrigation of an area of 1 000 000 m^2 (the size of Lake 4, see Table 4-1) with an annual amount of 150 mm of water the amount of lake water radionuclides transferred to the irrigated area was 2 % of the released amount ($1.93 \cdot 10^5$ Bq/$1 \cdot 10^7$ Bq) when applying a K_d-value of 1 m^3/kg and 0.4 % of the released amount ($4.43 \cdot 10^4$ Bq/$1 \cdot 10^7$ Bq) when K_d was set to 1 000 m^3/kg. On the other hand the levels of radionuclides in soils should increase due to a continuous build-up.

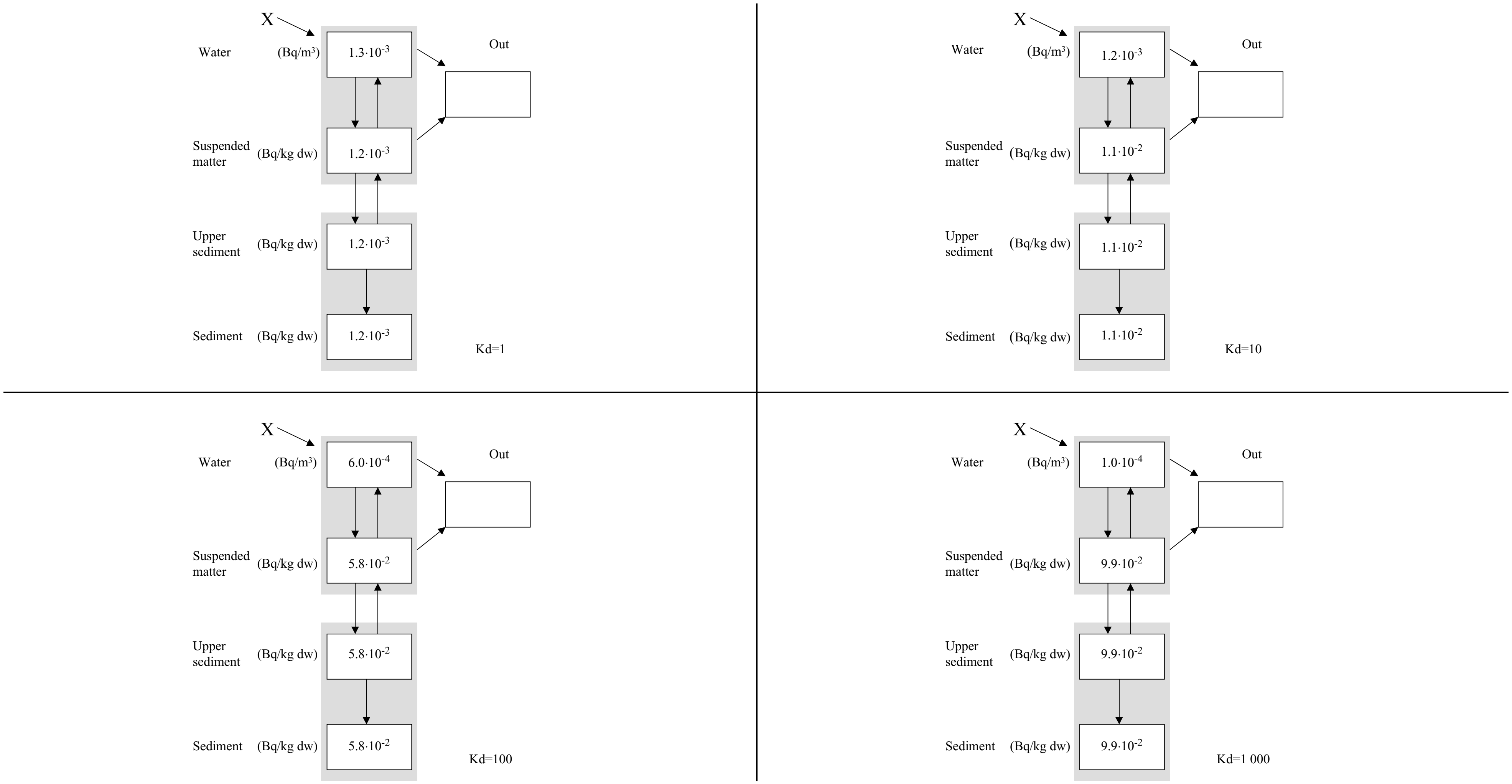


Figure 10-6. *Radionuclide concentrations in lake water (Bq/m^3(dissolved)), suspended matter (Bq/kg dw), upper and deeper sediments (Bq/kg dw) of the lake model at the end of a 1 000-year period with a constant discharge (10 000 Bq/y) of long-lived radionuclides with different K_d-values (1, 10, 100 and 1 000 m^3/kg). The concentrations in the sediments are the same as in the upper sediment layer and the suspended matter since no mineralisation or other dilution processes are considered. No concentrations are given for the outflow since data about the recipient is not of interest in this case.*

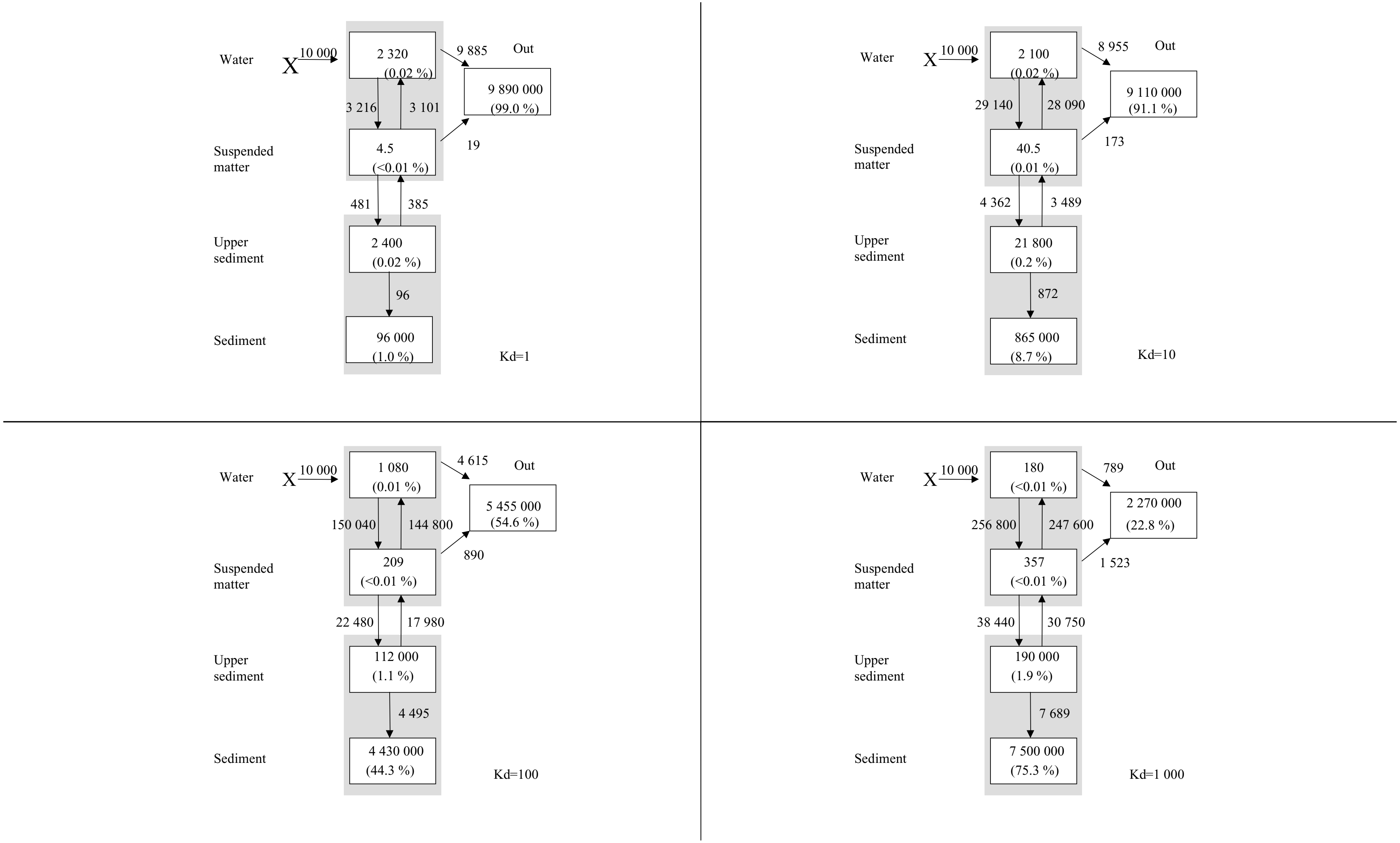
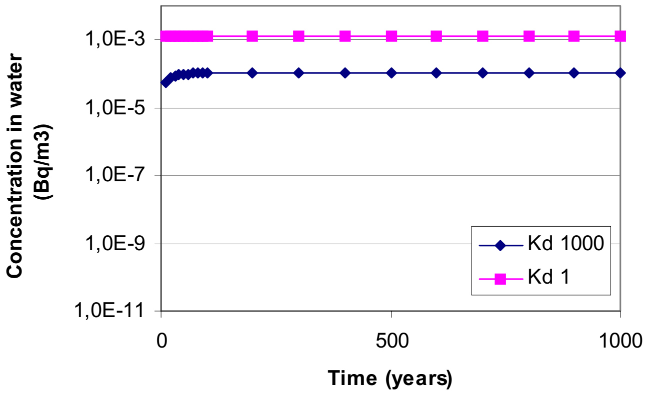


Figure 10-7. *Inventories (Bq) and annual fluxes (Bq/y) in the lake model at the end of a 1 000-year period with a constant discharge (10 000 Bq/y) of long-lived radionuclides with different K_d-values (1, 10, 100 and 1 000 m^3/kg).*



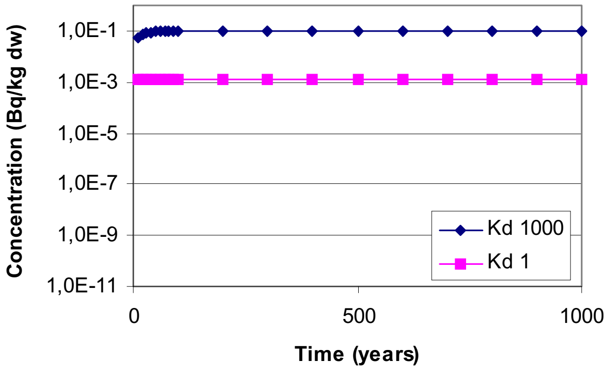


Figure 10-8. *Radionuclide concentrations in water (Bq/m^3(dissolved), above) and suspended matter (Bq/kg dw, below) in the lake model as a function of time when simulating a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 1 m^3/kg and 1 000 m^3/kg respectively. The concentration in upper sediment is not shown since it is the same as for the suspended matter for both K_d-values.*

10.2.1 Sediments as a secondary source of radionuclides in a lake

A possible scenario is that radionuclides are present within the deep sediments of a lake as a consequence of earlier sorption and deposition from marine surface water (at the coastal stage) or of contact with contaminated groundwater. The radionuclides may then diffuse upwards within the sediment and act as a source. The importance of this process is not well quantified but a field study of spiked sediments cores during three years /Andersson et al, 1992/ confirmed a high sorption for elements with varying properties, about the same orders of magnitude as could be expected for clay minerals. The study also showed a high sorption of elements at sediment surfaces. No explanation was given, but the results were probably due to incorporation of elements in organic matter.

The diffusive flux is thus difficult to estimate as it differs between radionuclides (with different sorption characteristics) and also with time as new sediment is deposited. Some simple and probably conservative estimates can be derived, however, assuming that the contamination of the sediments occurred by equilibration with a contaminated water and is followed by a leakage to a clean water, and that the mobility of radionuclides is controlled by diffusion of the dissolved fraction.

The diffusive mobility of radionuclides within the sediments is proportional to the concentration of the radionuclides in the dissolved phase. This is calculated as (see e.g. /IAEA, 1982/):

$$\%\text{dissolved} = \frac{1}{(1 + K_d \cdot S)} \cdot 100$$

where

K_d = Distribution coefficient, ratio of element concentrations in the solid and the dissolved phase [m^3/kg]

S = Concentration of solids [kg/m^3]

Similar to soils, the magnitude of the dissolved fraction is related to the availability of sorption surfaces also in water systems. Therefore, an efficient sorption (a high K_d-value) does not necessarily mean that the dissolved fraction is smaller than the particulate fraction. In surface waters with a low concentration of suspended matter of $1 \cdot 10^{-3}$ kg/m^3, 99.9 % of the radionuclides with a K_d-value of 1 m^3/kg will be dissolved and about 50 % of the radionuclides with a K_d-value of 1 000 m^3/kg. If the concentration of suspended matter instead is $1 \cdot 10^{-2}$ kg/m^3, 99 % will be dissolved at a K_d of 1 m^3/kg but only 10 % at a K_d of 1 000 m^3/kg.

The same equation applies to the distribution of radionuclides in sediments. On the other hand, S is several orders of magnitude higher in sediments than in the water, and the dissolved radionuclide fraction is always much smaller than the particulate fraction except for the most mobile radionuclides such as Cl-36. Note, however, that the sediment K_d may be different from the one in the water, often 1–2 orders of magnitude lower.

The water content in upper sediments where radionuclides may accumulate typically varies between 60 and 95 % /Meili et al, 2000b; Håkanson and Jansson, 1983/. At a solid density of 2 500 kg/m^3, this corresponds to a particle concentration of about 500 kg/m^3 and 50 kg/m^3, respectively. Using the equation above, the dissolved fraction ranges from about 2 % at a particle concentration of 50 kg/m^3 and a K_d of 1 m^3/kg to 0.0002 % at a particle concentration of about 500 kg/m^3 and a K_d of 1 000 m^3/kg.

A worst case scenario is that both desorption and diffusion of radionuclides in sediments are rapid enough to maintain the same concentration in the overlying lake water as in the pore water of the sediment. In principle, this requires a combination of an instantaneous sorption equilibration in the sediment and an infinitely long water residence time in the lake. In this case, the lake water concentration in a given system is basically controlled by the K_d value in the sediment. Since freshwater K_d often is equal to or higher than in marine systems (Table A-9 and A-10 in Appendix A), the maximum concentration in the lake will be the same as in the sea during the precedent contamination. This scenario is unlikely, even more so as it requires that no soil erosion or other input of sorbing matrices enters the lake to dilute or cover the radionuclides buried in the sediment. Furthermore, natural freshwater throughflow may result in a rapid depletion of radionuclides in the sediment and thus to a water contamination over only a limited period of time. The slower the release, the lower the concentration in the lake, but the longer the duration of the contamination.

Another less conservative but more realistic way is to assume an apparent pore water diffusive flow velocity of about 1 m/year (diffusive gradient of 1 cm, no resuspension considered) and a supply of radionuclides from a sediment layer with an average depth of about 1 m. In this case, the apparent pore water residence time is about 1 year, which implies an annual gross loss of radionuclides (thus excluding resedimentation) that is the same as the dissolved pool in the sediments over 1 m. According to the reasoning above this pool is unlikely to exceed 1 % of the total sediment inventory, which implies a duration on the order of a few decades or centuries until depletion. However, more likely is a much lower flux (<0.1 %/year) and a correspondingly longer duration (>1 000 years). The nuclides released are then diluted to low concentrations by the annual flow of water through the lake, which in this area can be expected to be several times larger than the lake volume (Table 4-1).

In the test runs of the coastal model, the maximum pool of radionuclides in the water of the Model Area was about 30 Bq when using a low K_d-value of 1 m^3/kg (see Figure 10-2). Given a water volume of $1 \cdot 10^8$ m^3 (see Table 3-2) this corresponds to a maximum brackish radionuclide concentration of $3 \cdot 10^{-7}$ Bq/m^3. Assuming that sorption equilibrium is established this corresponds to an initial sediment concentration of $3 \cdot 10^{-7}$ Bq/kg. With a sediment mass of about $5 \cdot 10^7$ kg dw (95 % water content, based on data in Table 4-1 and a sediment depth of 1 m after land uplift), the initial sediment pool of radionuclides will be 15 Bq, instead assuming a water content of 60 % the sediment mass is about $5 \cdot 10^8$ kg dw which gives an inventory of 150 Bq. If assuming that the concentration of radionuclides within the water is the same also for radionuclides with high K_d (fairly reasonable, cf sum of box 1 and 2 in Figures 10-2 to 10-5), i.e. $3 \cdot 10^{-7}$ Bq/m^3, the same reasoning suggests that the initial sediment pool of radionuclides with a K_d of 1 000 m^3/kg would be 15 000 Bq if the sediment water content is set to 95 % and 150 000 Bq at a water content of 60 %.

According to above, the release rate is less than 0.2 % per year for radionuclides with a low K_d and less than 0.0002 % per year for those with a high K_d-value if the water content of the sediment is 60 %. At an initial inventory of "low K_d radionuclides" of maximum 150 Bq and a release rate of <0.2 % per year, the maximum input to the lake water is less than 0.3 Bq/year. The maximum input of "high K_d radionuclides" will be <0.0002 % of 150 000 Bq and thus the same as above, that is less than 0.3 Bq/year. The maximum input to lake water is thus not depending on the K_d. Moreover, it does not depend on sediment water content either since <2 % of 15 Bq (K_d=1 m^3/kg) and <0.002 % of 15 000 Bq (K_d =1 000 m^3/kg) also equals less then 0.3 Bq/year. With an annual water flow of minimum $4 \cdot 10^6$ m^3/year (see Table 4-1) the maximum lake water concentration of radionuclides will be on the order of $1 \cdot 10^{-7}$ Bq/m^3 or less.

The distributions of radionuclides within the lake model where radionuclides are released from previously contaminated sediments were examined after a 1 000-year period. Two different K_d-values were tested; 1 and 1 000 m^3/kg. According to the reasoning above the initial inventory of radionuclides in the sediments was 150 Bq for the "low- K_d nuclide" and 150 000 Bq for the "high- K_d nuclide". The release rates were set to 0.2 and 0.0002 % of the inventory, respectively. The inventories and fluxes of radionuclides in and between different parts of the lake model at the end of the simulations are shown in Figures 10-9 and 10-10. As expected most of the "low K_d radionuclides" will leave the lake whereas most of the "high K_d nuclides" are still present within the deeper sediments. The amount present in the water and on suspended matter is about the same irresponsible of high or low K_d. The concentration is, however, low; about $1\cdot10^{-2\ \mathrm{Bq}}/\mathrm{m}^3$ representing about $8\cdot10^{-5}$ ($K_d = 1\ \mathrm{m}^3/\mathrm{kg}$) and $9\cdot10^{-8}$ ($K_d = 1\ 000\ \mathrm{m}^3/\mathrm{kg}$) of the initial inventories. Because of this the sediments were not modelled as a source of radionuclides in the lake model used in the SAFE-study. Instead radionuclides accumulated in the sediments during the lake stage were added to the pool which was than used as a source in the agricultural land model.

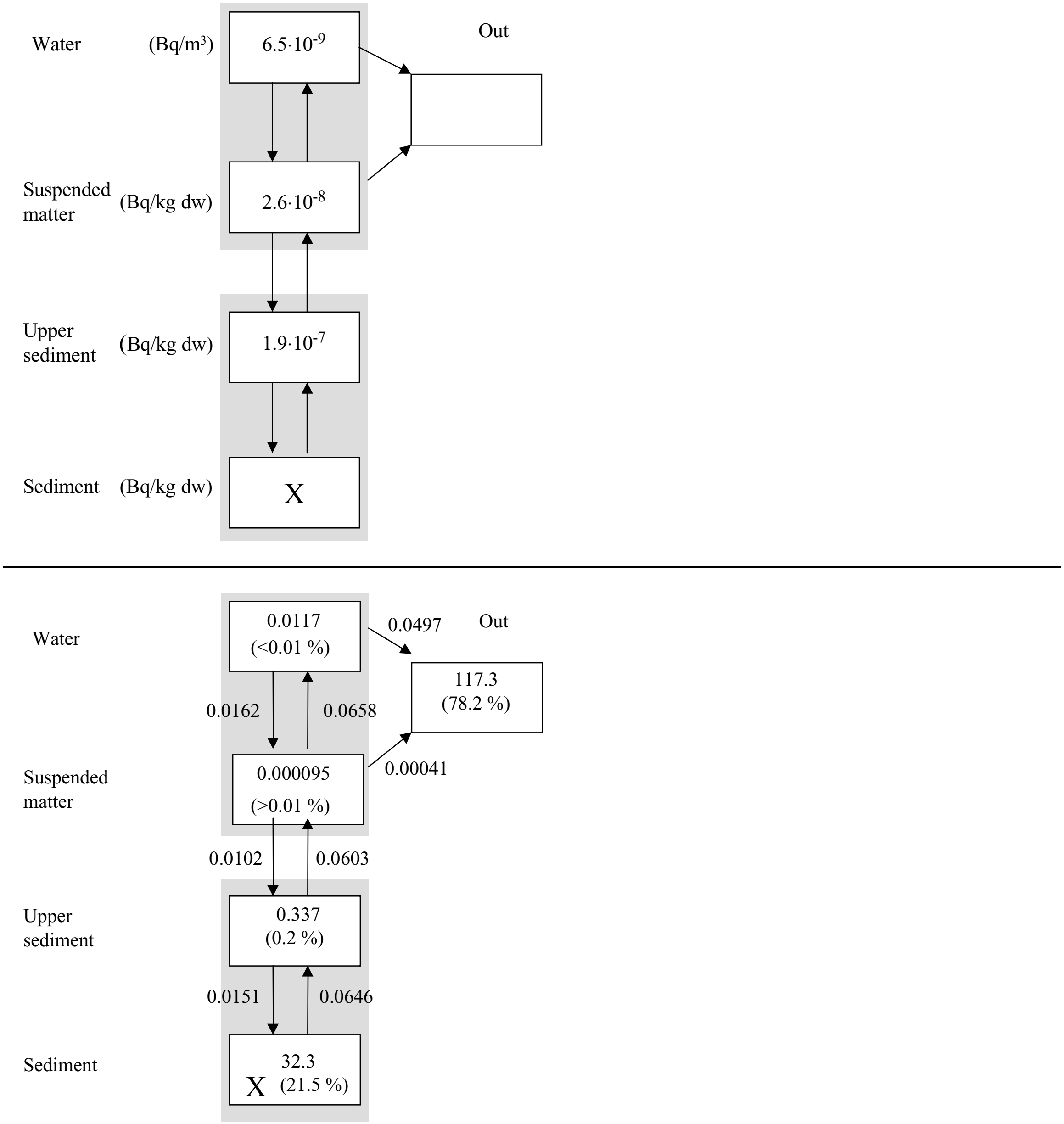


Figure 10-9. *Above concentrations (Bq/m³ and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/year) in and between the different compartments in the lake model the last year when simulating a time period of 1 000 years and using a K_d-value of 1 m³/kg. The sediments are assumed to act as a source of radionuclides with an initial inventory of 150 Bq. An annual leakage rate of 0.2 % of the inventory was used as reasoned in the text. The concentration in the deeper sediment layer is not given since the amount of sediments is not specified in this case. No concentration is given for the outflow since data about the recipient is not of interest in this case.*

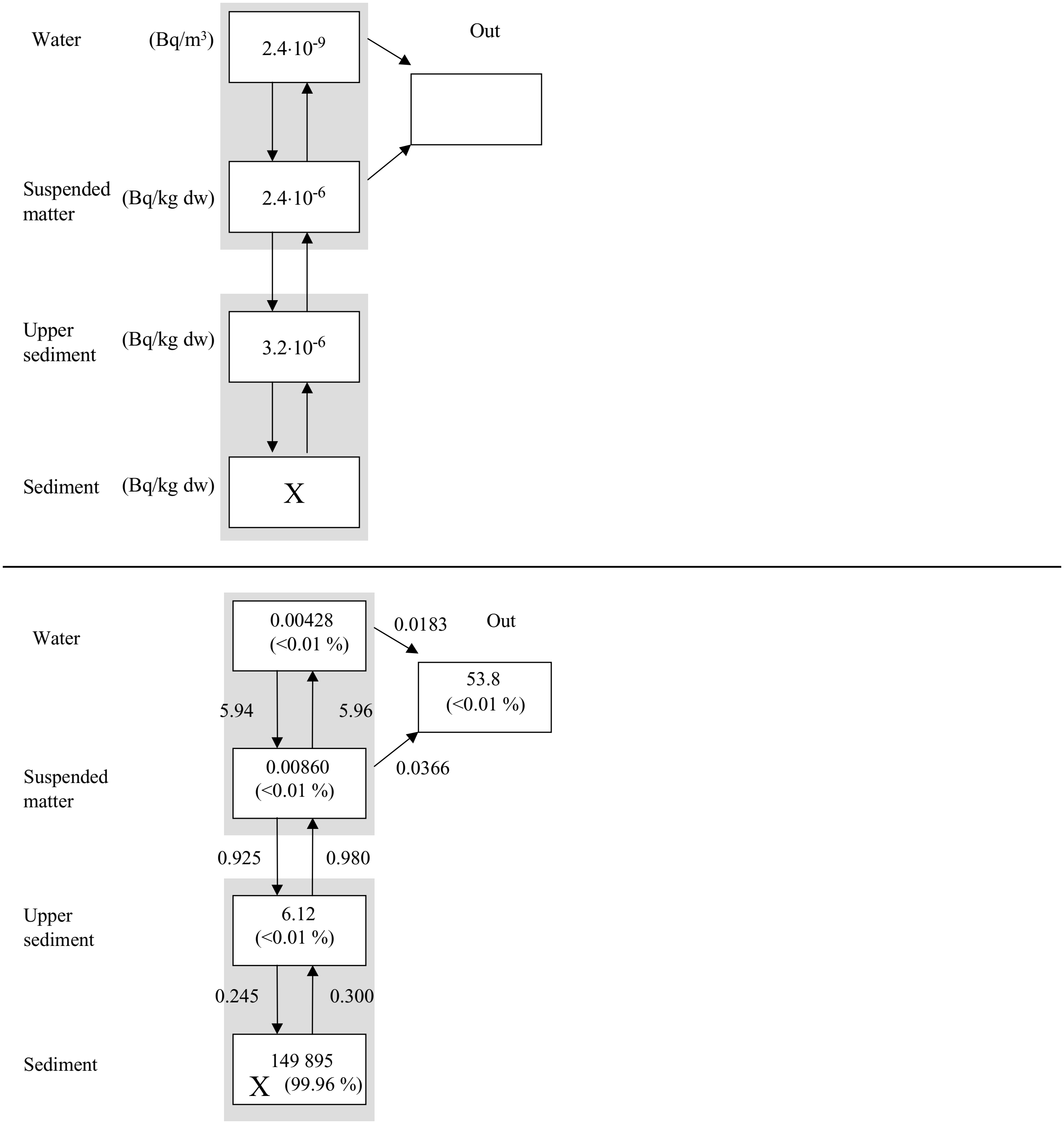


Figure 10-10. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/year) in and between the different compartments in the lake model the last year when simulating a time period of 1 000 years and using a K_d-value of 1 000 m^3/kg. The sediments are assumed to act as a source of radionuclides with an initial inventory of 150 000 Bq. An annual leakage rate of 0.0002 % of the inventory was used as reasoned in the text. The concentration in the deeper sediment layer is not given since the amount of sediments is not specified in this case. No concentration is given for the outflow since data about the recipient is not of interest in this case.*

The change in concentrations in the different model parts over time is presented in Figure 10-11. For radionuclides with a K_d-value of 1 000 m^3/kg an equilibrium is established almost immediately whereas it takes longer time for radionuclides with a K_d-value of 1 m^3/kg. For the latter, the concentrations reach a maximum in water as well as suspended matter and upper sediment after about 25 years and than decrease steadily during the whole time period. This is not surprising since such mobile nuclides will leak out from the lake to a large extent (see Figure 10-9).

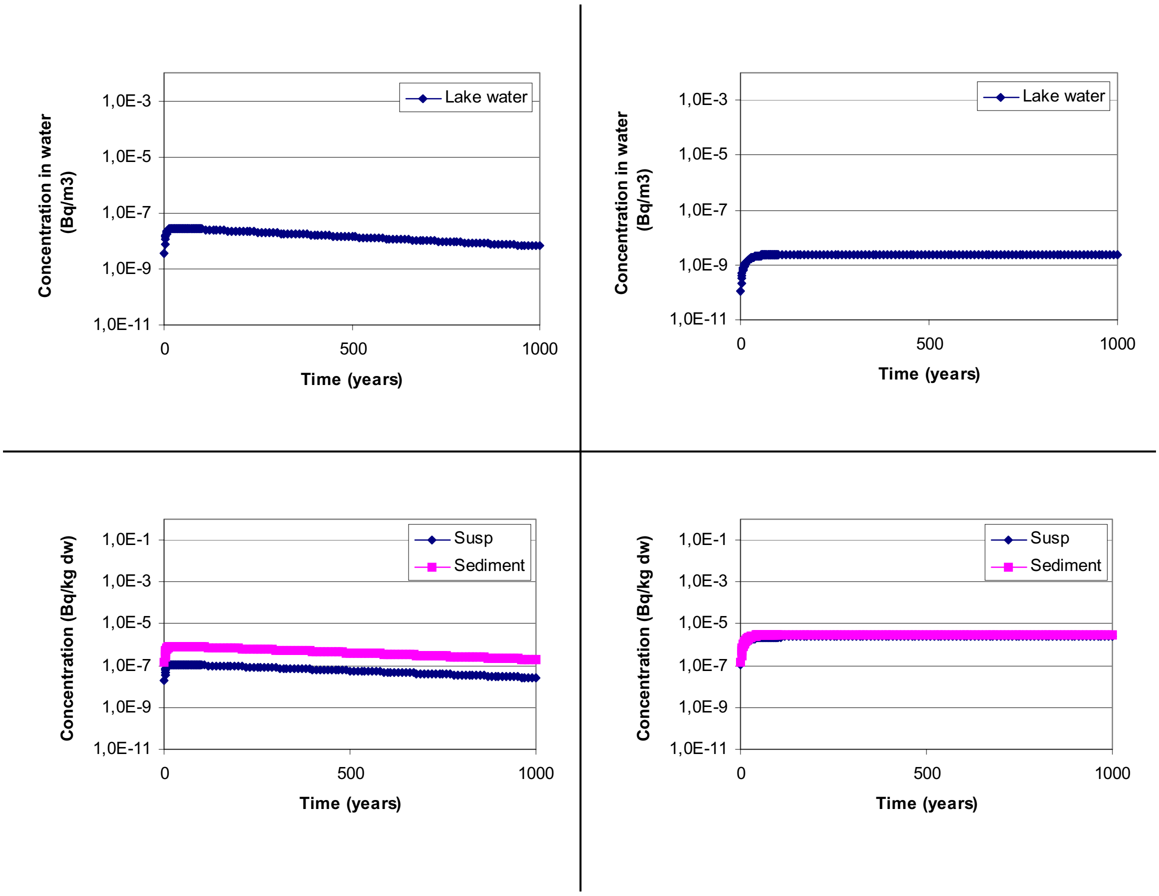


Figure 10-11. *Radionuclide concentrations in water (Bq/m^3(dissolved), above), suspended matter (Bq/kg dw) and upper sediment (Bq/kg dw), below, of the lake model as a function of time when simulating the sediment as a source of radionuclides (see text) with a K_d-value of 1 m^3/kg (left) and 1 000 m^3/kg (right).*

10.3 Agricultural land model

The distributions of radionuclides within the compartments included in the agricultural land model were examined after a 10 000-year period with an annual release of 10 000 Bq entering the system via the groundwater. Three different K_d-values were tested; 0.01, 1 and 100 m^3/kg. Lower K_d-values were used compared to the tests with the coastal and lake models since soil K_d is typically lower than sediment K_d (cf Tables A-7, A-9 and A-10). In the agricultural land model the mobile radionuclideswill migrate upwards with the water flow and reach the upper layers where they may be taken up by vegetation whereas radionuclides with high sorption will stay in the saturated zone sorbed to the solid matter. In the surface water models (lake and coastal models) mobile nuclides leave the system quickly and are then not available for local exposure. Compared to the tests of the surface water models a longer time period is simulated when testing the agricultural land model. This was done since it takes longer time to reach equilibrium in this system compared to the surface water systems. Note that the masses of the different compartments in the model are different from what is gained when using the figures in Table 5-1 (relevant for concentrations).The calculations presented in this chapter were performed before the values of all parameters were finally decided. This is of no relevance as the concentrations in the figures should be used for comparison between the figures.

The inventories and fluxes of radionuclides in and between the different parts of the model at the end of the simulation are shown in Figures 10-12 to 10-14. The highest "fluxes" occur between the solid and soluble phases of the saturated zone reflecting the rapid sorption-desorption equilibrium where the transfers in both directions are the same. The outflow of radionuclides, both with groundwater and through erosion, is largest with low K_d and decreases with increasing K_d-value. With a K_d of 0.01 m^3/kg a large fraction of the radionuclides leaves the soil and the pool sorbed to the solid phase in the saturated zone is about 100 times lower than the lost fraction. With a K_d of 1 m^3/kg these two fractions are of about the same size. For radionuclides with a K_d of 100 m^3/kg the sorbed fraction is about 100 times larger than the leaving fraction.

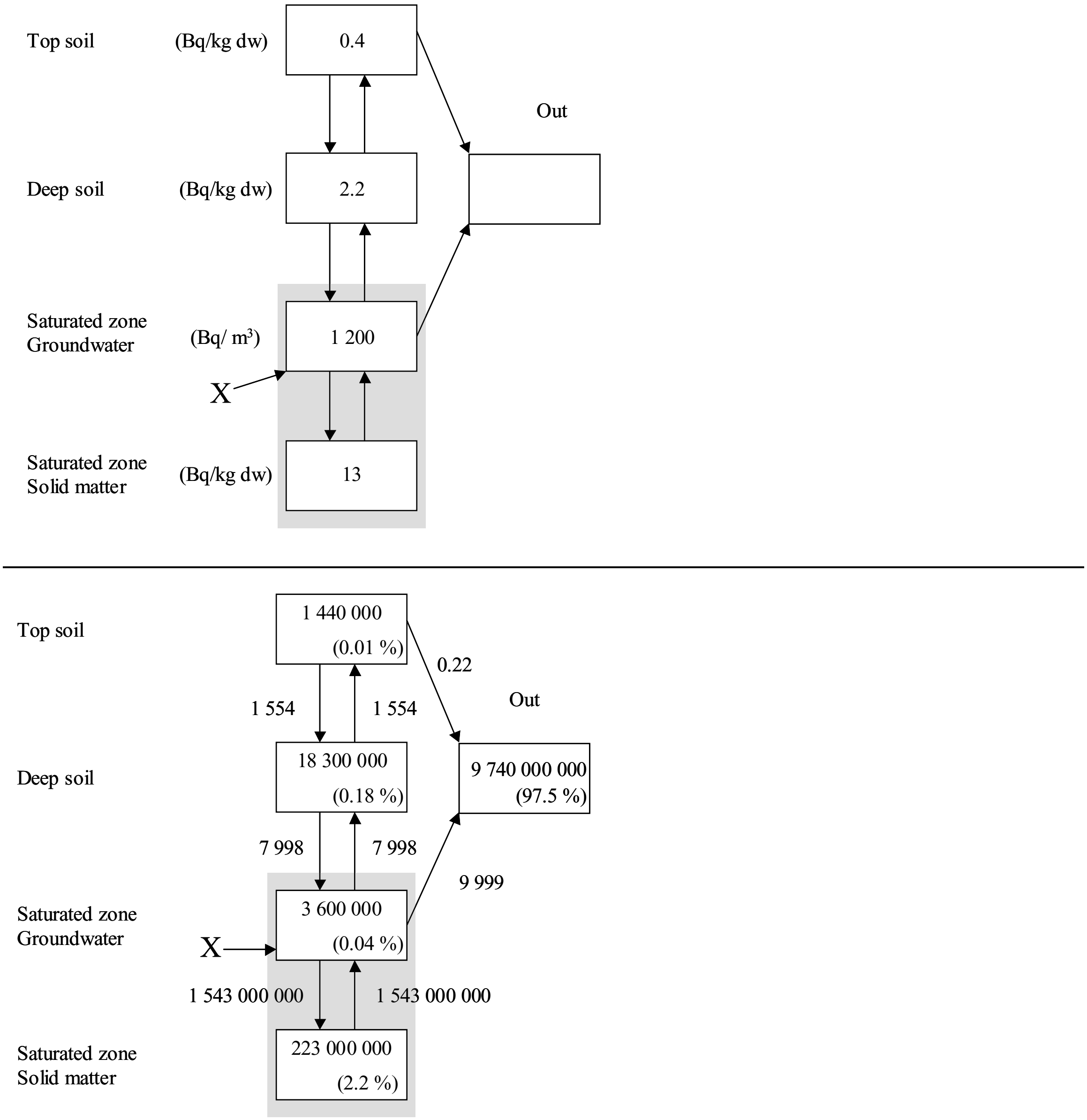


Figure 10-12. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the agricultural land model at the end of a 10 000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 0.01 m^3/kg. The large figures for the transfers between the solid matter and groundwater in the saturated zone is a consequence of the large number of time steps used when solving the differential equations, it should be seen as a transport back and forth. No concentration is given for the outflow since data about the recipient is not of interest in this case.*

An unexpected effect can be seen if comparing the inventory of the top soil in the three cases. The largest inventory is found for a radionuclide with a K_d-value of 1 m^3/kg whereas this inventory is of lower size for radionuclides with K_d-values of 0.01 and 100 m^3/kg. This is an example of a non-linear response to a gradual parameter change, which may be difficult to predict without the use of complex and dynamic models.

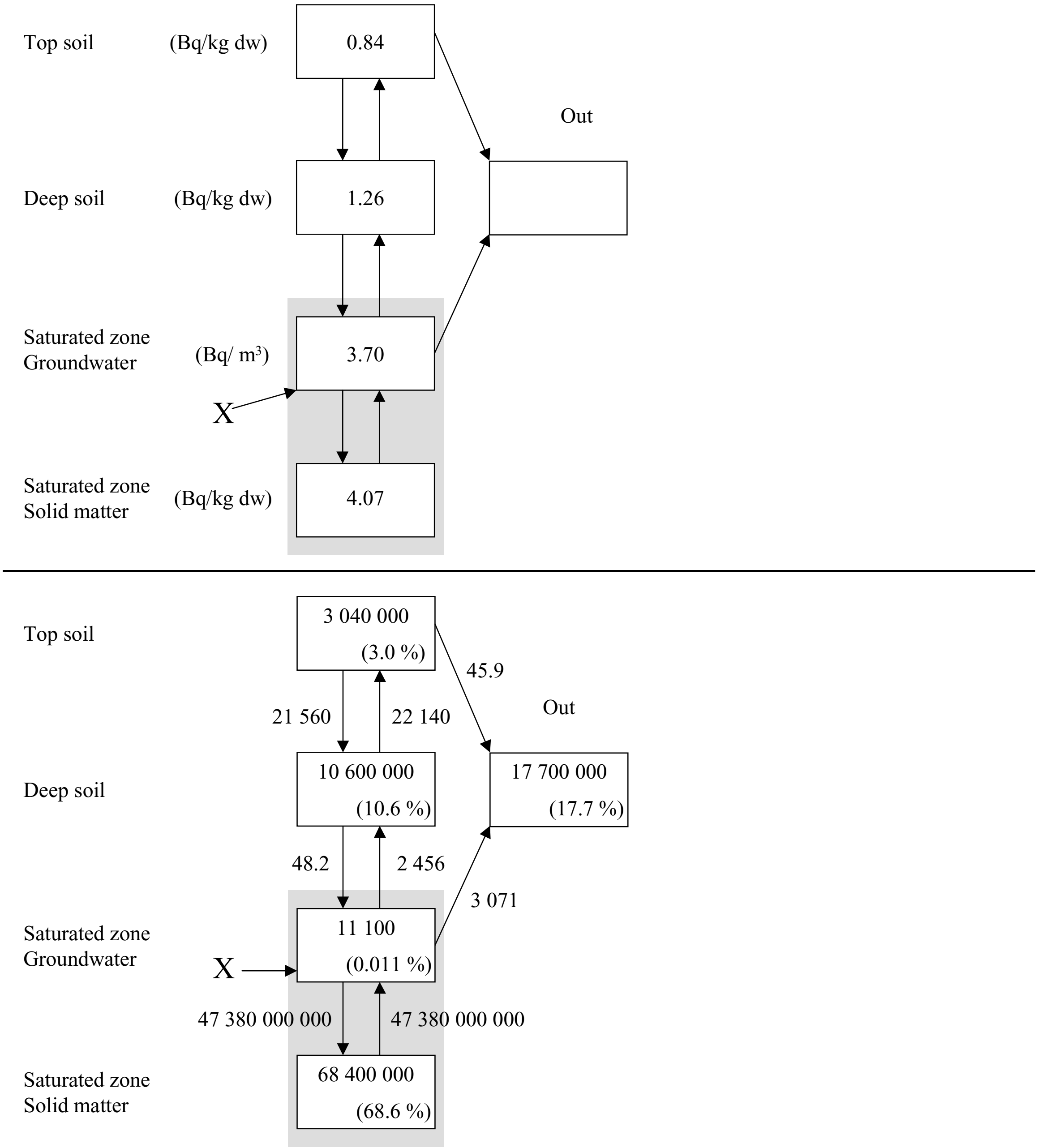


Figure 10-13. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the agricultural land model at the end of a 10 000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 1 m^3/kg. The large figures for the transfers between the solid matter and groundwater in the saturated zone is a consequence of the large number of time steps used when solving the differential equations, it should be seen as a transport back and forth. No concentration is given for the outflow since data about the recipient is not of interest in this case.*

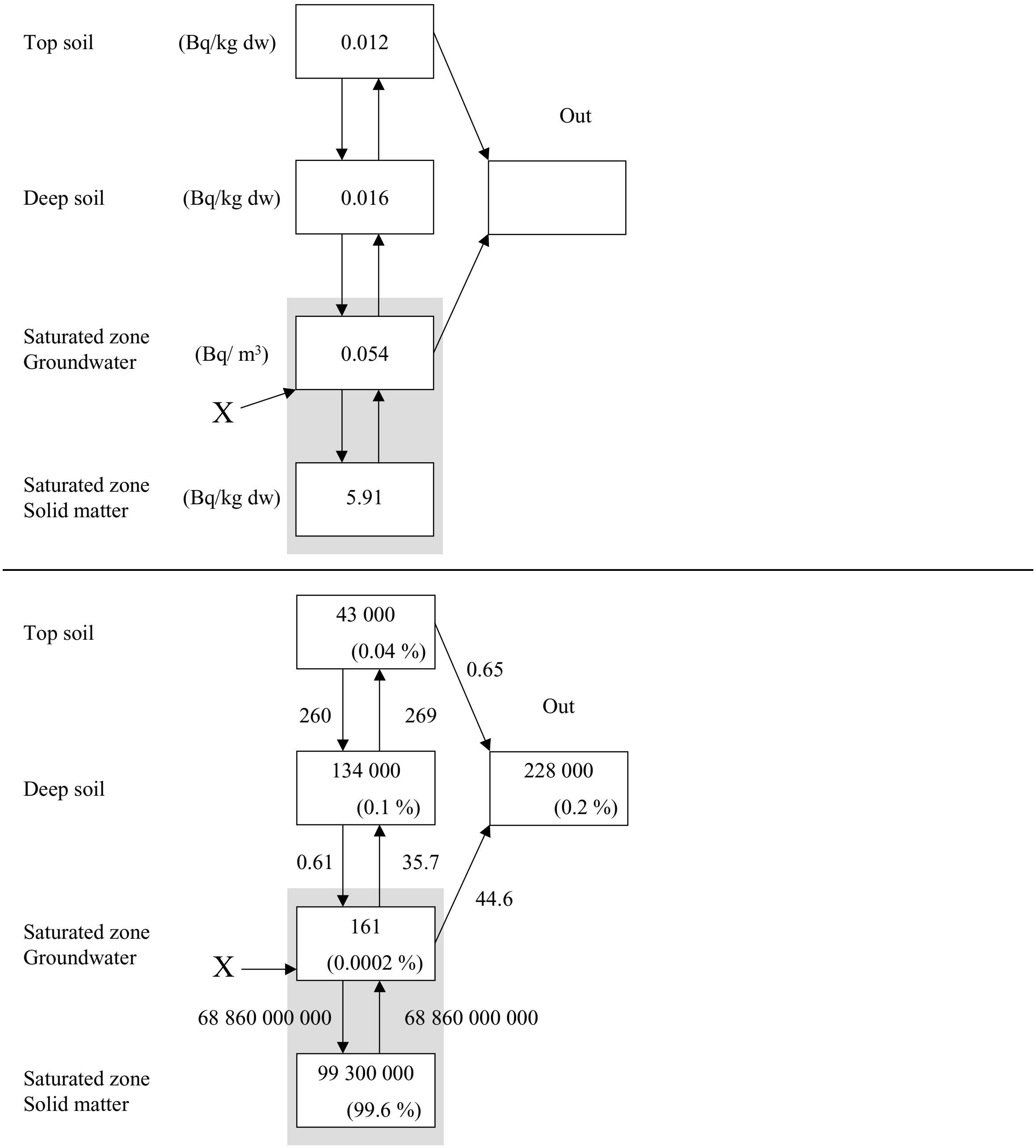


Figure 10-14. *Above concentrations (Bq/m^3 and Bq/kg dw) and below inventories (Bq) and annual fluxes (Bq/y) in the agricultural land model at the end of a 10 000-year period with a constant discharge (10 000 Bq/y) of a long-lived radionuclide with a K_d-value of 100 m^3/kg. The large figures for the transfers between the solid matter and groundwater in the saturated zone is a consequence of the large number of time steps used when solving the differential equations, it should be seen as a transport back and forth. No concentration is given for the outflow since data about the recipient is not of interest in this case.*

The change in concentrations in the different model parts over time is presented in Figure 10-15. The figures show that although a time period of 10 000 years was simulated, equilibrium has not been clearly established for any of the model parts except for the radionuclide concentrations for a nuclide with a K_d-value of 0.01 m^3/kg. Longer time period than this will probably not be simulated when calculating dose to humans from exposure via agricultural land.

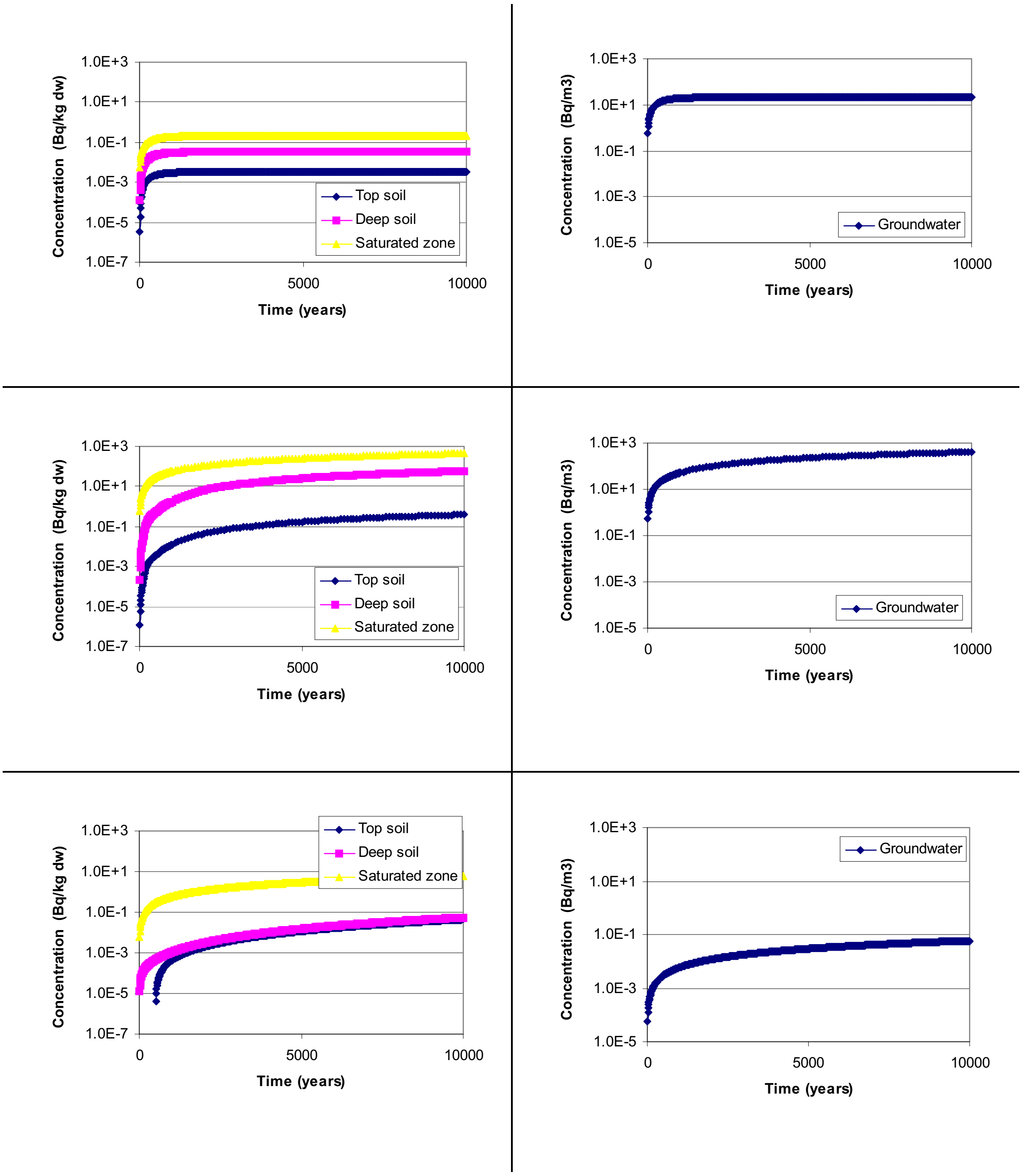


Figure 10-15. Radionuclide concentrations in different soil layers (left, Bq/kg dw) and groundwater (right, Bq/m^3) of the agricultural land model as a function of time when simulating a constant discharge (10 000 Bq/y) of long-lived radionuclides with K_d-values of 0.01 (upper figures), 1 (middle figures) and 100 m^3/kg (lower figures) during 10 000 years.

11 Summary and discussion

The models set up in this study have been subjected to various test scenarios, including different radionuclide entry points and sorption properties. The simulations show that most of the nuclides leave the Model Area and Öregrundsgrepen quickly as a consequence of the fast water turnover (about 90 % of the radionuclides with high sorption and even larger amounts of the less sorbing ones). The total amount of radionuclides in water is about the same independent of sorption properties and at most some percents of the amounts in the sediments. The latter is also true for the lake model when releases to the water was simulated.

An estimate of the turnover of radionuclides within lake sediment which has former been part of a contaminated coastal area was done. For radionuclides with low sorption to suspended matter the inventory in sediments will be low but the annual fraction released to the lake water will be higher compared to radionuclides with higher sorption. The amount of radionuclides in the water will therefore be of the same order of magnitude irresponsible of sorption properties. That fraction is anyway very small compared to the initial inventory; about $8 \cdot 10^{-5}$ and $9 \cdot 10^{-8}$ of the inventory for radionuclides with K_d =1 m^3/kg and 1 000 m^3/kg respectively. Radionuclides with low K_d-values tend to leave the lake rapidly whereas nuclides with high K_d to a large extent stays in the sediment.

The structure of the agricultural land model is very different from that of the surface water models (lake and coastal models). In the latter mobile nuclides leave the system quickly and are then not available for local exposure. In the agricultural land model the mobile radionuclides will migrate upwards with the water flow and reach the upper layers where they may be taken up by vegetation or are transported out of the system through erosion or groundwater movement. Radionuclides with high sorption will stay in the saturated zone sorbed to the solid matter. The highest concentration in the top soil layer will actually be of those radionuclides with intermediate sorption in soil. Another aspect is that it takes longer time to reach equilibrium in this system compared to the surface water systems. The test showed that equilibrium had not established after a modelling time period of 10 000 years. Longer periods than that was not run in the SAFE-study.

11.1 Uncertainties in data

The parameters used in the biosphere models can be divided into five sets according to their properties. **Geometrical** and **physical/chemical parameters** are used to calculate the dispersion of radionuclides in and between different parts of the models. **Biological and radiological parameters** as well as those describing **living habits of humans and cattle** are used when calculating the exposure to man. When estimating data for geometrical parameters (e.g. areas and depths of the different physical components) investigations at the site as well as model prognoses for the evolution of current conditions at the site have been used. This is also true for some physical/chemical parameters such as water retention times whereas data about e.g. suspended matter, fine particle sinking velocity, water transport in soil layers, soil density and porosity have been taken from studies of other relevant areas. For most biological parameters generic data have been used. These are often given a wide range i.a. to include the large variety which occur between different areas. One exception to this is the yield values for which data relevant for the area has been used when available. Data about living habits have been chosen to reflect Swedish conditions of today which should also include those at the SFR-area. The radiological parameters used are dose conversion factors and half-lives of radionuclides. For the former, data specified by the authorities have been used. The source chosen by SKB has been used for the half-lives.

The **geometrical data** used in the coastal model are connected with the lowest uncertainties as they describe the situation of today which has been investigated. For such data in the other models, results from models predicting the future evolution has been used and the uncertainties increase.

Comparing geometrical parameters with **physical/chemical and biological parameters** for all models it is clear that data for the former to a much larger extent are site-specific and therefore are given narrower distributions. The physical/chemical and biological ones are, with some exceptions, generic but the ranges are set up so that they should include the reasonable values describing the conditions in the area today as well as in the future.

Data about **living habits of humans and cattle** represent the situation in Sweden of today. How this will change in a far future (up to 10 000 years) can not be given for sure but in a reasonable time period we think that the values chosen will be relevant.

Radiological data, finally, are the only parameter category for which constant values are used. The dose coefficients are surely connected with both variations and uncertainties but in the literature no such information is available. As it is beyond the knowledge of the authors to set up relevant ranges deterministic values were used. The half-lives of radionuclides are known physical facts and those are the parameters which are connected with least uncertainties of the parameters used in this study.

11.2 Comparison with other studies

The two former studies to which comparison is done are those presenting the models used in the former safety analysis for SFR /Bergström and Puigdomenech, 1987/ and in the safety assessment SR 97 /Bergström et al, 1999/. In the former safety assessment for SFR two different models were used; initially for the coast and later for an inland area, created by landrise. The coast model was divided into local coast, regional coast, Baltic Sea and global Sea just like the coastal model in this study. The inland model included a lake and also a well. In SR 97 models for lakes, running waters, agricultural land, peat bogs, coastal areas and wells were used. These (except for the running waters model) have been the base for the models of this study (the peat bog model is here called mire model).

One way to compare results from models is to calculate conversion factors expressed in Sv/Bq at equilibrium for constant releases of the radionuclides. For the models used in the former safety assessment for SFR these factors were simply called conversion factors whereas for the models used in SR 97 they were called ecosystem specific dose conversion factors, EDFs. This term will be used hereafter irrespective of former names. EDFs for the coastal model of this study are presented in Appendix B.

When comparing the EDFs for Co-60 and Cs-135 for a coastal release in the former safety assessment of SFR (deterministically calculated) the EDFs in this study are somewhat lower (about two times for Cs-135 and eight times for Co-60) but about 1 000 times lower than those in the SR 97 study. This large difference is due to that the parameter values used in SR 97 described Äspö near Oskarshamn on the Swedish east-coast, where the water volumes are smaller and the water retention times longer compared to the conditions at SFR. The somewhat lower values, compared to the former safety assessment of SFR are due to differences in used water volumes and water retention times. The higher discrepancy for Co-60 than that for Cs-137 can be explained by the use of an askew parameter range for the bioaccumulation factor for fish in this study (see Table A-14).

As mentioned earlier the inland model of the former safety assessment for SFR included a lake as well as a well which makes comparisons difficult as these recipients are treated separately in this study. A comparison between the EDFs for the **lake models** of this study and the models of SR 97 (which includes irrigation of an agricultural field) shows no one-sided results. For Co-60, Pu-239 and Pu-240 the EDFs of this study are about one order of magnitude higher than those of the SR 97 model. For Cs-137 the EDF for this study is about ten times lower. The water volume passing through the lake modelled in SR 97 is about four times larger than for the lake modelled in this study. The main difference between the two models, though, is how sedimentation and resuspension processes are described, see section 4.2 and /Bergström et al, 1999/. Another difference is that irrigation of an agricultural field is not modelled in this study which was the case in the earlier one.

An **ecosystem based model** for the transport and fate of C-14 in the Model area has been developed /Kumblad, 2001/. Three different pathways for release of C-14 into the recipient are simulated; direct release into the water (called simulation A), direct uptake of benthic primary producers (simulation B) or a combination of these two, i.e. half of the total amount of C-14 is released into the water and the other half is directly taken up in biota (simulation C). The EDF gained for simulation A ($3 \cdot 10^{-19}$) is of the same order of magnitude as that of this study ($1 \cdot 10^{-18}$) when it has been modified to represent about the same consumption rate of fish (Kumblad L, Department of Systems Ecology, Stockholm University, personal communication 2001-09-27). The EDF for simulation C is about 400 times higher than that for simulation A which is about the same as that for simulation B. The large difference in exposure between simulation A and B/C shows the importance of the rapid water exchange in the area. The bottoms in the Model area are to a large extent covered with a layer of benthic algae /Kautsky et al, 1999; Snoeijs, 1985, 1986/ and elements in the groundwater passing this film may be taken up by these organisms. The extent of this uptake is, anyway, not known and therefore it is not possible to say which of the simulations is the most reasonable one.

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Appendix A

Element specific data used in the models are presented in this Appendix for the following parameters:

1) root uptake factors for different crops and pasturage,
2) translocation factors from vegetation surfaces to edible parts of cereals and root crops,
3) distribution coefficients for elements between solid and dissolved fractions in agricultural soil and in peat bogs,
4) distribution coefficients for elements between solution and suspended particles in lakes and the Baltic Sea,
5) distribution factors to milk and meat,
6) bioaccumulation factors for fish in fresh and brackish water as well as for marine water plants and fresh water crustacean.

All data are given as best estimates (B.E). Maximum and minimum of ranges and type of distribution used in the calculations are given in each table (LT = Logtriangular distribution, T = Triangular distribution). The reference for each value can also be seen.

Half-life and dose coefficients for ingestion, inhalation and external exposure are presented in Table A-1. For inhalation dose coefficients for different speeds of the passage through the lungs and intestine are tabulated /EU, 1996/. The highest value of those has been used.

Table A-1. Half-life /Firestone et al, 1999/ and dose coefficients for treated nuclides (ingestion and inhalation from /EU, 1996/ and external exposure from /Svensson, 1979/).

Nuclide	Type of dominating decay	Half-life Year	External exposure $(Sv/h)/(Bq/m^3)$	Ingestion Sv/Bq	Inhalation Sv/Bq
H-3	β	12	0	1.8E-11	2.6E-10
C-14	β	5 730	0	5.8E-10	5.8E-9
Cl-36	β	301 000	0	9.3E-10	7.3E-9
Co-60	β, γ	5	2.8E-13	3.4E-9	3.1E-8
Ni-59	β	76 000	0	6.3E-11	4.4E-10
Ni-63	β	100	0	1.5E-10	1.3E-9
Se-79	β	1 130 000	0	2.9E-9	6.8E-9
Sr-90	β	29	0	2.8E-8	1.6E-7
Zr-93	β	1 530 000	0	1.1E-9	2.5E-8
Nb-93m	γ	16	5.5E-17	1.2E-10	1.8E-9
Nb-94	β, γ	20 000	1.6E-13	1.7E-9	4.9E-8
Mo-93	EC	4 000	0	3.1E-9	2.3E-9
Tc-99	β	211 000	0	6.4E-10	1.3E-8
Pd-107	β	6 500 000	0	3.7E-11	5.9E-10
Ag-108m	γ	418	1.6E-13	2.3E-9	3.7E-8
Cd-113m	β	14	0	2.3E-8	1.1E-7
Sn-126	β, γ	100 000	3.0E-15	4.7E-9	2.8E-8
I-129	β	15 700 000	3.4E-16	1.1E-7	3.6E-8
Cs-135	β	2 300 000	0	2.0E-9	8.6E-9
Cs-137	β, γ	30	5.6E-14	1.3E-8	3.9E-8
Sm-151	β	90	4.6E-18	9.8E-11	4.0E-9
Eu-152	β, EC	14	1.5E-13	1.4E-9	4.2E-8
Eu-154	β, EC	9	1.3E-13	2.0E-9	5.3E-8
Eu-155	β	5	3.9E-15	3.2E-10	6.9E-9
Ho-166m	γ	1 200	1.6E-13	2.0E-9	1.2E-7
U-232	α	69	1.6E-17	3.3E-7	3.7E-5
Pu-238	α	88	1.3E-17	2.3E-7	1.1E-4
Pu-239	α	24 110	6.6E-18	2.5E-7	1.2E-4
Pu-240	α	6 563	0	2.5E-7	1.2E-4
Pu-242	α	372 300	0	2.4E-7	1.1E-4
Am-241	α	432	1.1E-15	2.0E-7	9.6E-5
Cm-243	α, EC	29	1.0E-14	1.5E-7	6.9E-5

Table A-2. Element specific root uptake factors for pasturage (which represents both grass and fodder)([Bq/kg d.w. grass]/[Bq/kg d.w. soil]).

Pasturage (dry veg/dry soil)

Element	B.E	Distr	Min	Max	Reference
H	5E+1	LT	2E+1	8E+1	1)
C	–	–	–	–	No root uptake considered
Cl	3E+1	T	1E+1	1E+2	2)
Co	1E-1	LT	1E-2	1E+0	3)
Ni	2E-1	LT	2E-2	2E+0	4)
Se	2E+1	LT	1E+0	3E+1	5)
Sr	1E+0	LT	4E-1	3E+0	4)
Zr	1E-3	LT	1E-4	1E-2	4)
Nb	5E-3	LT	5E-4	5E-2	6)
Mo	8E-1	LT	8E-2	8E+0	4)
Tc	8E+0	LT	8E-1	8E+1	4)
Pd	2E-1	LT	2E-2	2E+0	7)
Ag	5E-1	LT	5E-2	4E+0	8)
Cd	5E+0	LT	5E-1	5E+1	9)
Sn	1E-1	LT	1E-2	2E+0	10)
I	6E-1	LT	6E-2	6E+0	11)
Cs	2E-1	LT	2E-2	2E+0	4)
Sm	1E-2	LT	1E-3	1E-1	12)
Eu	1E-2	LT	1E-3	1E-1	12)
Ho	1E-3	LT	1E-4	1E-2	13)
U	2E-2	LT	2E-3	2E-1	4)
Pu	4E-4	LT	5E-5	7E-1	4)
Am	1E-3	LT	5E-4	2E-1	4)
Cm	1E-3	LT	1E-4	4E-3	4)

Comments to Table A-2

1 Calculated considering the same concentration of tritium in soil water as in water in plant.

2 /Coughtrey et al, 1985/ recommend a best estimate between 25 and 50 for “natural vegetation, pasturage herbage, edible vegetables and cereal grain”. A value of 30 has been used for pasturage and this value has been converted into wet weight for cereals, root crops and vegetables assuming water contents of 10, 80 and 90 %, respectively.

3 The number of studies on cobalt is limited. Data given in /IAEA, 1994/ and /Ng et al, 1982/ is of the same magnitude. The data used is extracted from these publications.

4 /IAEA, 1994/.

5 In /Coughtrey et al, 1985/ a range of ~2 → 66 is presented for “pasturage herbage”. A best estimate of 20 has been used and this value has been converted into wet weight for cereals, root crops and vegetables, assuming water contents of 10, 80 and 90 %, respectively.

6 /Davis et al, 1993/.

7 Due to lack of data the same values as for nickel has been used, as recommended in /Bergström et al, 1985/ referring to /Grogan, 1985b/. Pt is considered to be a better analogue for Pd /Moody, 1982/ but no data were available for this element either.

8 The best estimates and ranges are defined in /Bergström et al, 1991/. Silver is found in about the same concentrations in vegetation as in soil /Coughtrey et al, 1985; Bowen, 1979/. Very high concentrations were found in flour though. In /Coughtrey et al, 1985/ a general value of 1.0 kg/kg is recommended which is said to be a conservative value. This value is also recommended by /IAEA, 1982/. Considering this it is recommended in /Bergström et al, 1991/ to use a value of 0.5 kg/kg as best estimate for pasturage. For other crops this value is converted to wet weight, taking into account their different water content (10 % in cereals, 80 % in root crops and 90 % in vegetables). As a value of 1 is said to be conservative a maximum value four times that value is used. The minimum value used is ten times lower than the best estimate.

9 /NCRP, 1996/.

10 The root uptake factors for tin were updated in /Bergström et al, 1985/ as justified by data in /Coughtrey et al, 1983/. In the latter a value of 1 kg/kg is recommended but a value of 0.1 was used. This mistake has not been corrected so a too low value has been used.

11 /Deitermann et al, 1989/.

12 The best estimates are taken from /Aggeryd and Bergström, 1990/ who refer to /Baker et al, 1976/, /Miller et al, 1980/ and /Coughtrey and Thorne, 1983a/.

13 The best estimates are according to /Baker et al, 1976/ and /Coughtrey and Thorne, 1983a/. The value for pasturage was given for lanthanides in general.

Table A-3. Element specific root uptake factors for cereals ([Bq/kg w.w. cereals]/[Bq/kg d.w. soil]).

Cereals (grain/dry soil)

Element	B.E	Distr	Min	Max	Reference
H	5E+1	LT	2E+1	8E+1	1)
C	–	–	–	–	No root uptake considered
Cl	3E+1	T	9E+0	9E+1	2)
Co	1E-1	LT	1E-2	1E+0	3)
Ni	3E-2	LT	3E-3	3E-1	4)
Se	2E+1	LT	9E-1	3E+1	5)
Sr	2E-1	LT	2E-2	1E+0	4)
Zr	9E-4	LT	9E-5	9E-3	4)
Nb	4E-3	LT	4E-4	4E-2	6)
Mo	7E-1	LT	7E-2	7E+0	4)
Tc	6E-1	LT	6E-2	3E+0	4)
Pd	3E-2	LT	3E-3	3E-1	7)
Ag	4E-1	LT	4E-2	3E+0	8)
Cd	5E+0	LT	5E-1	5E+1	9)
Sn	4E-1	LT	1E-2	1E+0	10)
I	1E-1	LT	1E-2	1E+0	11)
Cs	2E-2	LT	2E-3	2E-1	4)
Sm	1E-4	LT	1E-5	1E-3	12)
Eu	2E-4	LT	2E-5	2E-3	12)
Ho	1E-4	LT	1E-5	1E-3	13)
U	1E-3	LT	1E-4	1E-2	4)
Pu	7E-6	LT	7E-7	7E-5	4)
Am	2E-5	LT	2E-6	2E-4	4)
Cm	2E-5	LT	1E-6	3E-4	4)

Comments to Table A-3

1 Calculated considering the same concentration of tritium in soil water as in water in plant.

2 /Coughtrey et al, 1985/ recommend a best estimate between 25 and 50 for “natural vegetation, pasturage herbage, edible vegetables and cereal grain”. A value of 30 has been used for pasturage and this value has been converted into wet weight for cereals, root crops and vegetables assuming water contents of 10, 80 and 90 %, respectively.

3 /Bergström et al, 1991/.

4 /IAEA, 1994/.

5 In /Coughtrey et al, 1985/ a range of ~2 → 66 is presented for “pasturage herbage”. A best estimate of 20 has been used and this value has been converted into wet weight for cereals, root crops and vegetables, assuming water contents of 10, 80 and 90 %, respectively.

6 /Davis et al, 1993/.

7 Due to lack of data the same values as for nickel has been used, as recommended in /Bergström et al, 1985/ referring to /Grogan, 1985b/. Pt is considered to be a better analogue for Pd /Moody, 1982/ but no data were available for this element either.

8 The best estimates and ranges are defined in /Bergström et al, 1991/. Silver is found in about the same concentrations in vegetation as in soil /Coughtrey et al, 1985; Bowen, 1979/. Very high concentrations have been found in flour though. In /Coughtrey et al, 1985/ a general value of 1.0 kg/kg is recommended which is said to be a conservative value. This value is also recommended by /IAEA, 1982/. Considering this it is recommended in /Bergström et al, 1991/ to use a value of 0.5 kg/kg as best estimate for pasturage. For other crops this value is converted to wet weight according to the water content of respective crop. As a value of 1 is said to be conservative a maximum value four times that value is used. The minimum value used is ten times lower than the best estimate.

9 /NCRP, 1996/.

10 The root uptake factors for tin were updated in /Bergström et al, 1985/ as justified by data in /Coughtrey et al, 1983/. In the latter a value of 1 kg/kg is recommended but a value of 0.1 was used. This mistake has not been corrected so a too low value has been used.

11 /Robens et al, 1988/.

12 The best estimates are taken from /Aggeryd and Bergström, 1990/ who refer to /Baker et al, 1976/, /Miller et al, 1980/ and /Coughtrey and Thorne, 1983a/.

13 The best estimates are according to /Baker et al, 1976/ and /Coughtrey and Thorne, 1983a/. The value for pasturage was given for lanthanides in general.

Table A-4. Element specific root uptake factors for root crops ([Bq/kg w.w. root crop]/[Bq/kg d.w. soil]).

Root crops (fresh veg/dry soil)

Element	B.E	Distr	Min	Max	Reference
H	1E+1	T	5E+0	2E+1	1)
C	–	–	–	–	No root uptake considered
Cl	6E+0	T	2E+0	2E+1	2)
Co	1E-2	LT	1E-3	1E-1	3)
Ni	4E-2	LT	4E-3	4E-1	4)
Se	4E+0	LT	2E-1	6E+0	5)
Sr	6E-2	LT	1E-2	3E-1	4)
Zr	2E-4	LT	2E-5	2E-3	4)
Nb	1E-3	LT	1E-4	1E-2	6)
Mo	2E-1	LT	2E-2	2E+0	4)
Tc	5E-2	LT	5E-3	5E-1	4)
Pd	4E-2	LT	4E-3	4E-1	7)
Ag	2E-1	LT	2E-2	1E+0	8)
Cd	1E+0	LT	1E-1	1E+1	9)
Sn	6E-2	LT	1E-2	1E+0	10)
I	1E-2	LT	1E-3	1E+0	11)
Cs	2E-2	LT	2E-3	2E-1	4)
Sm	4E-5	LT	4E-6	4E-4	12)
Eu	6E-5	LT	6E-6	6E-4	12)
Ho	9E-5	LT	9E-6	9E-4	13)
U	3E-3	LT	3E-4	3E-2	4)
Pu	3E-5	LT	3E-6	3E-4	4)
Am	4E-5	LT	4E-6	4E-4	4)
Cm	3E-5	LT	2E-6	5E-4	4)

Comments to Table A-4

1 Calculated considering the same concentration of tritium in soil water as in water in plant.

2 /Coughtrey et al, 1985/ recommend a best estimate between 25 and 50 for “natural vegetation, pasturage herbage, edible vegetables and cereal grain”. A value of 30 has been used for pasturage and this value has been converted into wet weight for cereals, root crops and vegetables assuming water contents of 10, 80 and 90 %, respectively.

3 /Bergström et al, 1991/.

4 /IAEA, 1994/.

5 In /Coughtrey et al, 1985/ a range of ~2 → 66 is presented for “pasturage herbage”. A best estimate of 20 has been used and this value has been converted into wet weight for cereals, root crops and vegetables, assuming water contents of 10, 80 and 90 %, respectively.

6 /Davis et al, 1993/.

7 Due to lack of data the same values as for nickel were used, as recommended in /Bergström et al, 1985/ referring to /Grogan, 1985b/. Pt is considered to be a better analogue for Pd /Moody, 1982/ but no data were available for this element either.

8 The best estimates and ranges are defined in /Bergström et al, 1991/. Silver is found in about the same concentrations in vegetation as in soil /Coughtrey et al, 1985; Bowen, 1979/. Very high concentrations have been found in flour though. In /Coughtrey et al, 1985/ a general value of 1.0 kg/kg is recommended which is said to be a conservative value. This value is also recommended by /IAEA, 1982/. Considering this it is recommended in /Bergström et al, 1991/ to use a value of 0.5 kg/kg as best estimate for pasturage. For other crops this value is converted to wet weight according to the water content of respective crop (10 % for cereals, 80 % for root crops and 90 % for vegetables). As a value of 1 is said to be conservative a maximum value four times that value is used. The minimum value used is ten times lower than the best estimate.

9 /NCRP, 1996/.

10 The root uptake factors for tin were updated in /Bergström et al, 1985/ as justified by data in /Coughtrey et al, 1983/. In the latter a value of 1 kg/kg is recommended but a value of 0.1 was used. This mistake has not been corrected so a too low value has been used.

11 /Robens et al, 1988/.

12 The best estimates are taken from /Aggeryd and Bergström, 1990/ who refer to /Baker et al, 1976/, /Miller et al, 1980/ and /Coughtrey and Thorne, 1983a/.

13 The best estimates are according to /Baker et al, 1976/ and /Coughtrey and Thorne, 1983a/. The value for pasturage was given for lanthanides in general. The minimum and maximum values of the ranges are ten times lower and higher respectively of the best estimate.

Table A-5. Element specific root uptake factors for vegetables ([Bq/kg w.w. vegetable]/[Bq/kg d.w. soil]).

Vegetables (fresh veg/dry soil)

Element	B.E	Distr	Min	Max	Reference
H	1E+1	T	5E+0	2E+1	1)
C	–	–	–	–	No root uptake considered
Cl	3E+0	T	1E+0	1E+1	2)
Co	1E-2	LT	1E-3	1E-1	3)
Ni	2E-2	LT	2E-3	2E-1	4)
Se	2E+0	LT	1E-1	3E+0	5)
Sr	3E-1	LT	3E-2	3E+0	4)
Zr	1E-4	LT	1E-5	1E-3	4)
Nb	5E-4	LT	5E-5	5E-3	6)
Mo	8E-2	LT	8E-3	8E-1	4)
Tc	2E+1	LT	1E-1	8E+1	4)
Pd	2E-2	LT	2E-3	2E-1	7)
Ag	1E-1	LT	1E-2	8E-1	8)
Cd	5E-1	LT	5E-2	5E+0	9)
Sn	5E-2	LT	1E-2	1E+0	10)
I	3E-2	LT	3E-3	3E-1	11)
Cs	2E-2	LT	2E-3	2E-1	4)
Sm	3E-3	LT	3E-4	3E-2	12)
Eu	3E-3	LT	3E-4	3E-2	12)
Ho	3E-3	LT	3E-4	3E-2	13)
U	1E-3	LT	1E-4	1E-2	4)
Pu	2E-5	LT	2E-6	2E-4	4)
Am	7E-5	LT	7E-6	7E-4	4)
Cm	8E-5	LT	8E-6	8E-4	4)

Comments to Table A-5

1 Calculated considering the same concentration of tritium in soil water as in water in plant.

2 /Coughtrey et al, 1985/ recommend a best estimate between 25 and 50 for “natural vegetation, pasturage herbage, edible vegetables and cereal grain”. A value of 30 has been used for pasturage and this value has been converted into wet weight for cereals, root crops and vegetables assuming water contents of 10, 80 and 90 %, respectively.

3 /Bergström et al, 1991/.

4 /IAEA, 1994/.

5 In /Coughtrey et al, 1985/ a range of ~2 → 66 is presented for “pasturage herbage”. A best estimate of 20 has been used and this value has been converted into wet weight for cereals, root crops and vegetables, assuming water contents of 10, 80 and 90 %, respectively.

6 /Davis et al, 1993/.

7 Due to lack of data the same values as for nickel have been used, as recommended in /Bergström et al, 1985/ referring to /Grogan, 1985b/. Pt is considered to be a better analogue for Pd /Moody, 1982/ but no data were available for this element either.

8 The best estimates and ranges are defined in /Bergström et al, 1991/. Silver is found in about the same concentrations in vegetation as in soil /Coughtrey et al, 1985; Bowen, 1979/. Very high concentrations has been found in flour though. In /Coughtrey et al, 1985/ is a general value of 1.0 kg/kg recommended which is said to be a conservative value. This value is also recommended by /IAEA, 1982/. Considering this it is recommended in /Bergström et al, 1991/ to use a value of 0.5 kg/kg as best estimate for pasturage. For other crops this value is converted to wet weight, according to the water content of respective crop. As a value of 1 is said to be conservative a maximum value four times that value is used. The minimum value used is ten times lower than the best estimate.

9 /NCRP, 1996/.

10 The root uptake factors for tin were updated in /Bergström et al, 1985/ as justified by data in /Coughtrey et al, 1983/. In the latter a value of 1 kg/kg is recommended but a value of 0.1 was used. This mistake has not been corrected so a too low value has been used.

11 /Robens et al, 1988/.

12 The best estimates are taken from /Aggeryd and Bergström, 1990/ who refer to /Baker et al, 1976/, /Miller et al, 1980/ and /Coughtrey and Thorne, 1983a/.

13 The best estimates are according to /Baker et al, 1976/ and /Coughtrey and Thorne, 1983a/. The value for pasturage was given for lanthanides in general. The minimum and maximum values of the ranges are ten times lower and higher respectively of the best estimate.

Table A-6. Element specific translocation factors from surface to edible part of cereals and root crops ([Bq/kg w.w.]/[Bq/m^2]), triangularly distributed.

Translocation factors (m^2/kg)

Element	Mean	Min	Max	Reference
H	1E-1	1E-2	3E-1	1)
C	1E-1	1E-2	3E-1	1)
Cl	1E-1	1E-2	3E-1	1)
Co	2E-1	1E-1	3E-1	2)
Ni	1E-2	5E-3	4E-2	3)
Se	1E-1	1E-2	3E-1	2)
Sr	4E-1	1E-1	7E-1	2)
Zr	1E-1	1E-2	3E-1	1)
Nb	2E-1	1E-1	3E-1	2)
Mo	1E-1	1E-2	3E-1	2)
Tc	5E-1	4E-1	6E-1	2)
Pd	1E-1	1E-2	3E-1	1)
Ag	1E-1	1E-2	3E-1	1)
Cd	2E-2	6E-5	5E-2	4)
Sn	1E-1	1E-2	3E-1	1)
I	1E-1	5E-2	2E-1	5)
Cs	2E-1	1E-1	3E-1	4)
Sm	1E-1	1E-2	3E-1	1)
Eu	2E-2	1E-2	3E-2	6)
Ho	1E-1	1E-2	3E-1	1)
U	1E-1	1E-2	3E-1	1)
Pu	2E-2	1E-2	3E-2	2)
Am	1E-2	5E-3	2E-2	2)
Cm	2E-2	1E-2	3E-2	7)

Comments to Table A-6

1 Since data are lacking a best estimate of 10 % has been used as for many other elements for which other data have not been found. A rather wide range has been used as the uncertainty is high.

2 /Coughtrey et al, 1985/.

3 Data is scarce but according to /Coughtrey et al, 1985/ it is assumed that very little nickel is transferred to edible parts. Therefore a low value has been used.

4 /IAEA, 1994/.

5 /Coughtrey et al, 1983/.

6 /Bergström et al, 1991/.

7 In lack of data the same value as for plutonium is used.

Table A-7. Element specific distribution coefficients (K_d) for soil, concentration in solid matter/concentration in solution ([Bq/kg d.w.]/[Bq/m^3]).

K_d Soil (m^3/kg)

Element	B.E	Distr	Min	Max	Reference
H	–	–	–	–	1)
C	1E-3	LT	4E-4	1E-2	2)
Cl	1E-3	LT	1E-4	1E-2	3)
Co	1E+0	LT	1E-2	2E+1	4)
Ni	5E-1	LT	5E-2	5E+0	4)
Se	1E-2	LT	1E-3	1E-1	5)
Sr	1E-2	LT	1E-3	1E-1	4)
Zr	1E+0	LT	1E-1	1E+1	4)
Nb	5E-1	LT	5E-3	5E+0	4)
Mo	1E-1	LT	1E-2	1E+0	4)
Tc	5E-3	LT	1E-3	1E-2	6)
Pd	2E-1	LT	2E-2	2E+0	4)
Ag	1E-1	LT	1E-2	1E+0	4)
Cd	1E-1	LT	2E-3	3E+0	4)
Sn	1E-1	LT	5E-2	5E-1	7)
I	3E-1	LT	1E-1	1E+0	8)
Cs	1E+0	LT	1E-1	1E+1	9)
Sm	1E+0	LT	1E-1	1E+1	4)
Eu	2E+2	LT	2E+1	2E+3	9)
Ho	1E+0	LT	1E-1	1E+1	4)
U	1E-1	LT	1E-2	1E+0	10)
Pu	5E+0	LT	1E-1	1E+1	11)
Am	2E+0	LT	2E-1	2E+1	4)
Cm	1E+1	LT	1E+0	1E+2	4)

Comments to Table A-7

1 Hydrogen does not sorb to particle matter to a large extent and therefore no effort has been put on finding K_d-values.

2 The best estimate and range were defined in /Bergström and Nordlinder, 1990b/ but the minimum value given in the reference has been increased somewhat (before $1 \cdot 10^{-4}$). The best estimate is orginally from /Andersson et al, 1982/.

3 Estimated from /McKinley and Scholtis, 1992/.

4 /IAEA, 1994/.

5 Extracted from /Coughtrey et al, 1985/.

6 Estimated from /Hoffman and Baes, 1979/.

7 Tin is assumed to be analogous with Pb for which data orginally was taken from /Jiskra, 1985/. This study refers to /Wuschke et al, 1981/.

8 The best estimate has been calculated from a transfer rate used in /Bergström and Wilkens, 1983/. The transfer rate was extracted from /Kantelo et al, 1981/.

9 /NCRP, 1996/.

10 In /Grogan, 1985a/ this value was used with reference to /Parker and Grant, 1979/ and /Dahlman et al, 1976/.

11 /Coughtrey et al, 1985/.

Table A-8. Element specific distribution coefficients (K_d) for peat (organic soil), concentration in solid matter/concentration in solution ([Bq/kg d.w.]/[Bq/m^3]).

K_d Peat (m^3/kg)

Element	B.E	Distr	Min	Max	Reference
H	–	–	–	–	1)
C	7E-2	LT	7E-3	7E-1	2)
Cl	1E-2	LT	1E-3	1E-1	3)
Co	1E+0	LT	5E-2	2E+0	4)
Ni	1E+0	LT	2E-1	7E+0	4)
Se	2E+0	LT	2E-1	2E+1	4)
Sr	2E-1	LT	4E-3	6E+0	4)
Zr	7E+0	LT	7E-1	7E+1	4)
Nb	2E+0	LT	2E-1	2E+1	4)
Mo	3E-2	LT	3E-3	3E-1	4)
Tc	2E-3	LT	4E-5	6E-2	4)
Pd	7E-1	LT	7E-2	7E+0	4)
Ag	2E+1	LT	2E+0	9E+1	4)
Cd	8E-1	LT	8E-3	8E+1	4)
Sn	2E+0	LT	2E-1	2E+1	4)
I	3E-2	LT	3E-3	3E-1	4)
Cs	3E-1	LT	1E-1	3E+0	4)
Sm	3E+0	LT	3E-1	3E+1	4)
Eu	1E+0	LT	5E-2	2E+1	5)
Ho	3E+0	LT	3E-1	3E+1	4)
U	4E-1	LT	3E-3	4E+0	4)
Pu	2E+0	LT	2E-1	2E+1	4)
Am	1E+2	LT	1E+1	1E+3	4)
Cm	1E+1	LT	1E+0	1E+2	4)

Comments to Table A-8

1 Hydrogen does not sorb to particle matter to a large extent and therefore no effort has been put on finding K_d-values.

2 /Davis et al, 1993/.

3 No value was found in /IAEA, 1994/ or elswhere so an estimation was performed. Assuming that chlorine behaves like iodine in reducing environments (personal communication P-O Aronsson, Ringhals NPP, 2001-10-22) a K_d-value close to that used for iodine has been used.

4 /IAEA, 1994/.

5 Data are lacking for this uncommon element and as Eu-152, Eu-154 and Eu-155 have short half-lives it has not been of interest in any of the safety assessments performed by the authors and therefore no effort has been placed on finding data. The values for other lantanindes vary. The highest value (for Ce) is 2 000 m^3/kg. As the nuclides have short half-lives they are important for dose to humans in a short time perspective, which makes accumulation in sediments an unimportant process. To be conservative a relatively low K_d-value has been used so that a larger fraction is available for exposure to man.

Table A-9. Element specific distribution coefficients (K_d) for suspended matter in lakes, concentration in solid matter/concentration in solution ([Bq/kg d.w.]/[Bq/m^3]).

K_d Lake sediment (m^3/kg)

Element	B.E	Distr	Min	Max	Reference
H	–	–	–	–	1)
C	1E-3	LT	1E-4	1E-2	2)
Cl	1E+0	LT	1E-1	1E+1	3)
Co	5E+0	LT	1E+0	7E+1	4)
Ni	1E+1	LT	1E+0	1E+2	3)
Se	5E+0	LT	1E+0	1E+1	5)
Sr	1E+0	LT	1E-1	1E+1	4)
Zr	1E+0	LT	1E-1	1E+1	4)
Nb	1E+1	LT	1E+0	1E+2	3)
Mo	1E-3	LT	1E-4	1E-2	6)
Tc	1E-1	LT	1E-2	1E+0	5)
Pd	2E+0	LT	2E-1	2E+1	3)
Ag	2E+0	LT	2E-1	2E+1	3)
Cd	1E-1	LT	1E-2	1E+0	7)
Sn	5E+1	LT	1E+1	1E+2	2)
I	3E-1	LT	1E-1	1E+0	5)
Cs	1E+1	LT	1E+0	1E+2	5)
Sm	5E+0	LT	5E-1	5E+1	3)
Eu	5E-1	LT	5E-2	5E+0	4)
Ho	3E-1	LT	3E-2	3E+0	8)
U	1E+1	LT	1E+0	1E+2	9)
Pu	1E+2	LT	1E+1	1E+3	5)
Am	5E+0	LT	5E-1	5E+1	4)
Cm	5E+0	LT	1E-1	7E+1	4)

Comments to Table A-9

1 Hydrogen does not sorb to particle matter to a large extent and therefore no effort has been put on finding K_d-values since they are missing in e.g. /IAEA, 1994/.

2 /Bergström and Nordlinder, 1990b/.

3 /McKinley and Scholtis, 1992/.

4 /IAEA, 1994/.

5 /Coughtrey et al, 1985/.

6 The best estimate is taken from /Aggeryd and Bergström, 1990/. In that report the same K_d-values were used for sediments as for soil. The value is orginally from /Jiskra, 1985/ who refers to /Inoue and Morisawa, 1974/. The same value has been used for freswater and brackish water.

7 /NCRP, 1996/.

8 These values were presented in /Bergström et al, 1999/ with reference to /McKinley and Scholtis, 1992/. In the latter no values are presented for Ho, instead the values from the row below (values for I) have been read by mistake. Comparing these values with those for e.g. Sm (another lanthanide) they are very low. Uptake in biota is low which also indicates rather strong sorption properties. For the results in SR 97 /Bergström et al, 1999/ and SAFE /Karlsson et al, 2001/ this mistake makes the calculations of exposure to man more conservative as a larger fraction of the released amounts is available for uptake in aquatic biota. Because of the relatively short half-life, accumulation of Ho in sediments is of no importance in the SAFE study.

9 According to /Bergström and Wilkens, 1983/ a distribution coefficient for lakes in northern Sweden of 43 m^3/kg (mean value) was reported by /Holm, 1981/ and a range of 0.2–250 m^3/kg was also given in /Bergström and Wilkens, 1983/. From this information a value of 10 m^3/kg has been extracted.

Table A-10. Element specific distribution coefficients (K_d) for suspended matter in brackish waters, concentration in solid matter/concentration in solution ([Bq/kg d.w.]/[Bq/m^3]).

K_d Baltic Sea sediment (m^3/kg)					
Element	**B.E**	**Distr**	**Min**	**Max**	**Reference**
H	1E-3	LT	5E-5	1E-2	1)
C	1E-3	LT	1E-4	1E-2	2)
Cl	1E-3	LT	1E-4	1E-2	1)
Co	1E+2	LT	1E+0	2E+2	3)
Ni	1E+1	LT	1E+0	1E+2	2)
Se	5E+0	LT	1E+0	1E+1	4)
Sr	1E-1	LT	1E-2	1E+0	5)
Zr	5E+1	LT	5E+0	5E+2	4)
Nb	1E+1	LT	1E+0	1E+2	4)
Mo	1E-3	LT	1E-4	1E-2	6)
Tc	1E-1	LT	1E-2	1E+0	4)
Pd	1E+1	LT	1E+0	1E+2	3)
Ag	1E+0	LT	1E-1	1E+1	7)
Cd	5E+0	LT	1E+0	1E+2	1)
Sn	5E+1	LT	1E+1	1E+2	8)
I	3E-1	LT	1E-1	1E+0	4)
Cs	1E+1	LT	1E+0	1E+2	4)
Sm	1E+2	LT	1E+1	1E+3	2)
Eu	1E+1	LT	1E+0	1E+2	9)
Ho	1E-1	LT	1E-2	1E+0	10)
U	1E+1	LT	1E+0	1E+2	11)
Pu	1E+2	LT	1E+1	1E+3	4)
Am	1E+1	LT	1E+0	1E+2	4)
Cm	1E+3	LT	1E+1	2E+3	12)

Comments to Table A-10

1 /IAEA, 1985/.

2 /McKinley and Scholtis, 1992/.

3 /Bergström and Nordlinder, 1993/.

4 /Coughtrey et al, 1985/.

5 The best estimate is derived from /Bergström and Nordlinder, 1990a/ who treat lake water. It is most possible that this study has used the Kd-value for fresh water used in /Bergström and Puigdomenech, 1987/ (0.1 m^3/kg). This value is orginally extracted from /Kenna, 1980/.

6 The best estimate is taken from /Aggeryd and Bergström, 1990/. In that report the same Kd-values were used for sediments as for soil. The value is orginally from /Jiskra, 1985/ who refers to /Inoue and Morisawa, 1974/. The same value has been used for freswater and brackish water.

7 The best estimate and range are defined in /Bergström and Nordlinder, 1993/. The Kd-values in that study were extracted from /Coughtrey et al, 1985/, /Puigdomenech and Bergström, 1995/ and /McKinley and Scholtis, 1992/.

8 /Bergström and Nordlinder, 1990b/.

9 Data are lacking for this uncommon element and the values for other lanthanides vary. The highest value (for Ce) is 2 000 m^3/kg. As the nuclides have short half-lives they are important for dose to humans in a short time perspective which makes accumulation in sediments an unimportant process. To be conservative a relatively low K_d-value has been used so that a larger fraction is available for exposure to man.

10 These values were presented in /Bergström et al, 1999/ with reference to /McKinley and Scholtis, 1992/. In the latter no values are presented for Ho, instead the values from the row below (values for I) have been read by mistake. Comparing these values with those for e.g. Sm (another lanthanide) they are very low. Uptake in biota is low which also indicates rather strong sorption properties. For the results in SR 97 /Bergström et al, 1999/ and SAFE /Karlsson et al, 2001/ this mistake makes the calculations of exposure to man more conservative as a larger fraction of the released amounts is available for uptake in aquatic biota. Because of the relatively short half-life, accumulation of Ho in sediments is of no importance in the SAFE study.

11 The values for fresh waters has been used for the Baltic sea as well.

12 Best estimate and range are defined in /Bergström and Nordlinder, 1993/. The Kd-values in that study were extracted from /Coughtrey et al, 1985/, /Puigdomenech and Bergström, 1995/ and /McKinley and Scholtis, 1992/.

Table A-11. Element specific transfer coefficients to cow milk from daily intake ([Bq/l]/[Bq intake/day]).

F-milk (day/l)

Element	B.E	Distr	Min	Max	Reference
H	2E-2	LT	1E-2	3E-2	1)
C	1E-2	LT	5E-3	2E-2	2)
Cl	1.7E-2	T	1.5E-2	2E-2	3)
Co	3E-4	LT	6E-5	1E-2	3)
Ni	2E-2	LT	2E-3	5E-2	3)
Se	4E-3	LT	4E-4	4E-2	4)
Sr	2.8E-3	LT	1E-3	3E-3	3)
Zr	6E-7	LT	6E-8	6E-6	3)
Nb	4E-7	LT	1E-7	4E-6	3)
Mo	2E-3	LT	2E-4	2E-2	3)
Tc	2E-5	LT	1E-5	1E-3	3)
Pd	1E-3	LT	1E-4	1E-2	5)
Ag	5E-5	LT	5E-6	5E-4	3)
Cd	1E-4	LT	1E-5	1E-3	6)
Sn	1E-3	LT	1E-4	1E-2	7)
I	1E-2	LT	1E-3	4E-2	3)
Cs	8E-3	LT	1E-3	3E-2	3)
Sm	2E-5	LT	2E-6	2E-4	8)
Eu	2E-5	LT	2E-6	2E-4	8)
Ho	2.5E-6	LT	3E-7	3E-5	9)
U	4E-4	LT	7E-5	6E-4	3)
Pu	1E-6	LT	3E-9	3E-6	3)
Am	2E-6	LT	4E-7	2E-5	3)
Cm	2E-5	LT	2E-6	2E-4	10)

Comments to Table A-11

1 /Van den Hoek et al, 1979/ have studied the metabolism of tritium in cattle. The studies showed that secretion of H-3 gave a transfer coefficient of 0.016 (average value). This value has been rounded off and a narrow range has been used since tritium is a constituent of water and therefore the variation is considered not to be so large

2 The best estimates are taken from /Bergström and Puigdomenech, 1987/ with reference to unpublished results by Bergström and Hoffman. The results have not been published but the value used is in accordance with that used by /Davis et al, 1993/; $1.5 \cdot 10^{-2}$ day/l for milk and $6.4 \cdot 10^{-2}$ day/kg for meat. Values for carbon are based on the amounts of the stable element in different medium and organisms and as that content do not vary very much for cattle and grass respectively a rather narrow range has been used.

3 /IAEA, 1994/.

4 /Davis et al, 1993/.

5 The amount of data concerning the metabolism of palladium in cattle is not large, e.g. no values are found in /IAEA, 1994/. The values given in /NCRP, 1996/ and those used in /Davis et al, 1993/ differ very much. For milk a value of $1 \cdot 10^{-4}$ day/l is presented by NCRP and $1 \cdot 10^{-2}$ day/l is used by Davis et al. The values for meat are $2 \cdot 10^{-4}$ day/kg (NCRP) and $4 \cdot 10^{-3}$ day/kg (Davis et al). The values used here are set to include these ranges.

6 An equilibrium forage-to-milk transfer coefficient of $1 \cdot 10^{-4}$ day/l is suggested in /Coughtrey et al, 1985/. It is also stated that this value is larger than suggested by available experimental results and may require modification. To use a conservative approach this value is used.

7 /NCRP, 1996/.

8 /Miller et al, 1980/.

9 /Baker et al, 1976/.

10 /IAEA, 1982/.

Table A-12. Element specific transfer coefficients to cow meat from daily intake ([Bq/kg d.w.]/[Bq intake/day]).

F-meat (day/kg)

Element	B.E	Distr	Min	Max	Reference
H	1E-2	LT	5E-3	2E-2	1)
C	3E-2	LT	1.5E-2	6E-2	2)
Cl	2E-2	LT	1E-2	4E-2	3)
Co	1E-2	LT	4E-5	7E-2	3)
Ni	5E-3	LT	5E-4	5E-2	3)
Se	1.5E-2	LT	1E-4	2E-2	4)
Sr	8E-3	LT	3E-4	1E-2	3)
Zr	1E-6	LT	1E-7	1E-5	3)
Nb	3E-7	LT	3E-8	1E-2	3)
Mo	1E-3	LT	1E-4	1E-2	3)
Tc	1E-4	LT	1E-5	1E-3	3)
Pd	1E-3	LT	1E-4	1E-2	5)
Ag	3E-3	LT	2E-3	6E-3	3)
Cd	4E-4	LT	4E-5	4E-3	3)
Sn	1E-2	LT	1E-3	1E-1	6)
I	4E-2	LT	7E-3	5E-2	3)
Cs	5E-2	LT	1E-2	6E-2	3)
Sm	5E-3	LT	5E-4	5E-2	7)
Eu	6E-3	LT	6E-4	6E-2	8)
Ho	5E-3	LT	5E-4	5E-2	7)
U	3E-4	LT	3E-5	3E-3	3)
Pu	1E-5	LT	2E-7	2E-4	3)
Am	4E-5	LT	4E-6	1E-4	3)
Cm	2E-5	LT	2E-6	2E-4	9)

Comments to Table A-12

1 In /Bergström et al, 1991/ a special model for calculation of dose from tritium was set up. The transfer coefficients used were taken from /Neil, 1991/. A transfer coefficient from vegetation to meat of 0.2 kg/kg grass was used. In SR 97 and SAFE another model approach has been used and instead of this kind of transfer coefficient a transfer coefficient to meat related to the cow's consumption of cereals and grass has been used. With a consumption rate of about 20 kg/day (5 kg grass and 12 kg cereals per day in SR 97 and 8.5 kg grass and 11 kg cereals per day in SAFE) and a transfer coefficient of 0.01 day/kg this matches the transfer coefficient from vegetation to meat given by /Neil, 1991/. In accordance to what is said about the transfer coefficient to milk a narrow range has been used since tritium is a constituent of water and therefore the variation is considered not to be so large.

2 The best estimates are taken from /Bergström and Puigdomenech, 1987/ with reference to unpublished results by Bergström and Hoffman. The results have not been published but the value used is in accordance with that used by /Davis et al, 1993/; $1.5\cdot10^{-2}$ day/l for milk and $6.4\cdot10^{-2}$ day/kg for meat. Values for carbon are based on the amounts of the stable element in different medium and organisms and as that content do not vary very much for cattle and grass respectively a rather narrow range has been used.

3 /IAEA, 1994/.

4 /Davis et al, 1993/.

5 The amount of data concerning the metabolism of palladium in cattle is not large, e.g. no values are found in /IAEA, 1994/. The values given in /NCRP, 1996/ and those used in /Davis et al, 1993/ differ very much. For milk a value of $1 \cdot 10^{-4}$ day/l is presented by NCRP and $1 \cdot 10^{-2}$ day/l is used by Davis et al. The values for meat are $2 \cdot 10^{-4}$ day/kg (NCRP) and $4 \cdot 10^{-3}$ day/kg (Davis et al). The values used here are set to include these ranges.

6 /NCRP, 1996/.

7 /Baker et al, 1976/.

8 /Miller et al, 1980/.

9 /IAEA, 1982/.

Table A-13. Element specific bioaccumulation factors to fish in freshwater ([Bq/kg w.w.]/[Bq/l]).

BAF freshwater fish (l/kg w.w.)

Element	B.E	Distr	Min	Max	Reference
H	1E+0	LT	5E-1	2E+0	1)
C	5E+4	LT	5E+3	5.1E+4	2)
Cl	5E+1	LT	1E+1	1E+2	3)
Co	3E+2	LT	1E+1	4E+2	1)
Ni	1E+2	LT	1E+1	1E+3	1)
Se	2E+3	LT	5E+2	5E+3	4)
Sr	6E+1	LT	1E+0	1E+3	1)
Zr	2E+2	LT	3E+0	3E+2	1)
Nb	3E+2	LT	1E+2	3E+4	1)
Mo	1E+1	LT	1E+0	1E+2	1)
Tc	2E+1	LT	2E+0	8E+1	1)
Pd	1E+2	LT	1E+1	1E+3	5)
Ag	5E+0	LT	2E-1	1E+1	1)
Cd	2E+1	LT	2E+0	2E+2	6)
Sn	3E+3	LT	3E+2	3E+4	1)
I	2E+2	LT	1E+1	5E+2	7)
Cs	1E+4	LT	5E+3	2E+4	8)
Sm	3E+1	LT	3E+0	3E+2	9)
Eu	5E+1	LT	1E+1	2E+2	1)
Ho	3E+1	LT	3E+0	3E+2	9)
U	1E+1	LT	2E+0	5E+1	1)
Pu	3E+1	LT	4E+0	3E+2	1)
Am	3E+1	LT	1E+1	3E+2	1)
Cm	3E+1	LT	1E+1	3E+2	1)

Comments to Table A-13

1 /IAEA, 1994/.

2 /Bergström and Nordlinder, 1990b/.

3 /Coughtrey et al, 1985/.

4 Extracted from /Coughtrey et al, 1985/.

5 Due to lack of data the same values as for nickel were used, as recommended in /Bergström et al, 1985/ referring to /Grogan, 1985b/. That study concerns root uptake but a similar behaviour between palladium and nickel can also be expected in aquatic environments. Pt is considered to be a better analogue for Pd /Moody, 1982/ but no data were available for this element either.

6 /Coughtrey and Thorne, 1983b/

7 /Poston and Klopfer, 1986/.

8 Value valid for oligotrophic freshwater. In /Bergström and Nordlinder, 1990b/ a value of 5 000 l/kg is used based on a number of studies. /Kohlemainen et al, 1968/ gives bioaccumulation factors in the range of 200–2 000 l/kg for piscivorous fish in eutrophic (nutrient rich) lake systems and /Neumann, 1985/ gives a generic value of 200 l/kg. /Vanderploegh et al, 1975/ give the transfer at steady state as a function of the potassium concentration in water; 15 000/[K] for piscivorous fish. Studies of the Chernobyl fallout in the same type of lakes give a value of about 5 000 l/kg for species used as food /Bergström and Nordlinder, 1989/. In /Bergström et al, 1999/ a best estimate of 10 000 l/kg was used for oligotrophic lakes (with low potassium content). The value is higher than for oligotrophic lakes in accordance to the function set up by /Vanderploeg et al, 1975/.

9 /USNRC, 1977/.

Table A-14. Element specific bioaccumulation factors for fish to be used in the coastal model ([Bq/kg w.w.]/[Bq/l]).

BAF brackish water fish (l/kg w.w.)

Element	B.E	Distr	Min	Max	Reference
H	1E+0	LT	5E-1	2E+0	1)
C	2E+3	LT	1.8E+3	3E+3	2)
Cl	1E+0	LT	1E-1	1E+1	3)
Co	3E+2	LT	3E+1	5E+2	4)
Ni	3E+2	LT	3E+1	5E+2	5)
Se	4E+3	LT	2E+3	8E+3	6)
Sr	3E+1	LT	1E+0	1E+2	7)
Zr	1E+2	LT	1E+1	2E+2	8)
Nb	1E+2	LT	1E+1	5E+2	9)
Mo	1E+1	LT	1E+0	5E+1	10)
Tc	3E+1	LT	1E+0	1E+2	11)
Pd	1E+1	LT	1E+0	1E+2	12)
Ag	5E+2	LT	1E+2	1E+3	13)
Cd	2E+2	LT	2E+1	2E+3	6)
Sn	1E+3	LT	1E+2	1E+4	14)
I	3E+1	LT	1E+1	1E+2	15)
Cs	2E+2	LT	1E+2	5E+2	16)
Sm	3E+1	LT	3E+0	3E+2	17)
Eu	1E+2	LT	1E+1	1E+3	12)
Ho	3E+1	LT	3E+0	3E+2	17)
U	5E+1	LT	1E+1	1E+2	18)
Pu	3E+1	LT	5E+0	5E+1	19)
Am	1E+2	LT	1E+1	2E+2	20)
Cm	5E+1	LT	1E+1	3E+2	21)

Comments to Table A-14

1 A best estimate of 1 l/kg was recommended in /Bergström and Nordlinder, 1992/ based on /Strand et al, 1976/ and /Blaylock and Frank, 1979/.

2 Based on the carbon content in fresh fish and the rates in surrounding water a bioaccumulation factor of about 2 400 l/kg is gained for the Baltic Sea (calculated from carbon amounts reported in /Hesböl et al, 1990/). The factor gets about 1 800 l/kg for the Swedish west coast. A best estimate of 2 000 l/kg has therefore been used.

3 /Coughtrey et al, 1985/. The value for marine fish has been used since the behaviour of chlorine in that kind of environments is more similar to that in brackish water than the behaviour in freshwater environments.

4 In /Bergström and Nordlinder, 1992/ a best estimate of 300 l/kg and a range of 20–400 l/kg are recommended for brackish water fish. In /Coughtrey and Thorne, 1983b/ a value of 3 l/kg is recommended for fish flesh whereas /Poston and Klopfer, 1986/ recommends much higher values (100 l/kg for marine environments and 330 l/kg for freshwater fish when water quality is unknown). Because of this uncertainty the range has been altered somewhat, i.e. minimum and maximum values have both been increased.

5 The best estimate and range are recommended in /Bergström and Nordlinder, 1992/. The accumulation of nickel in muscles of marine animals is limited /Coughtrey and Thorne, 1983b/. Due to the small amount of data these authors recommend data for cobalt to be used instead. The low uptake has also been confirmed by /Tjaelve et al, 1988/.

6 Estimated from /Coughtrey et al, 1983b/.

7 The value is taken from /Bergström and Nordlinder, 1992/. This study recommends a best estimate of 30 l/kg for brackish waters based on recommendations in /Patzer, 1976/ of bioaccumulation factors of 3–10 l/kg for fresh water fish and 0.9–90 L/kg for marine fish. The range given here includes these two ranges.

8 The best estimate and range are taken from /Bergström and Nordlinder, 1992/ who refer to /Coughtrey and Thorne, 1983a/ for long-time exposure. A factor of 10 is recommended for muscle concerning short-time exposure.

9 The best estimate and range are taken from /Bergström and Nordlinder, 1992/. The bioaccumulation factor presented in /Thompson et al, 1972/ is 30 000 l/kg which is considerably higher than what has been found by others /ScottRussel, 1966/ and /Ancellin et al, 1979/. /Ancellin et al, 1979/ has measured factors between 1 and 30 in in-situ studies and between 110 and 260 in experiments. /Freeke, 1967/ used a factor of 100 in his calculations and this value has also been used here.

10 The best estimate and range are taken from /Bergström and Nordlinder, 1992/ who refer to /Short et al, 1971/.

11 The best estimate and range are taken from /Bergström and Nordlinder, 1992/ based on the following information; /Blaylock and Frank, 1979/ measured bioaccumulation factors from 11 to 121 for different fish species in a small pond. Laboratory experiments with marine fish show lower factors, about 2 l/kg /Masson et al, 1989/. /Pentreath, 1981/ measured an average value of 11 l/kg whereas /Verthé et al, 1986/ measured a bioaccumulation factor of 0.2 l/kg in marine environment for Serranus cabrilla. As is said in /Beasley and Lorz, 1986/ the uptake of technetium is considered to be low in fish.

12 /NCRP, 1996/.

13 The best estimate and range are taken from /Bergström and Nordlinder, 1992/ who refer to /Coughtrey and Thorne, 1983b/. The value for freshwater in /Coughtrey and Thorne, 1983b/ has been used.

14 A value of 100 l/kg is recommended in /Bergström and Nordlinder, 1992/ which is said to be in accordance to the recommendation given in /Coughtrey et al, 1983/. A value of 1 000 l/kg is given in /Coughtrey et al, 1983/ anyway and therefore this value has been used in SAFE whereas a value of 100 was used in SR 97 /Bergström et al, 1999/.

15 The best estimate and range are taken from /Bergström and Nordlinder, 1992/. A best estimate of 30 is given for freshwater fish in /Coughtrey et al, 1983/.

16 The best estimate and range are taken from /Bergström and Nordlinder, 1992/. This study refers to /Evans, 1985, 1991/ and /Grimås, 1991/.

17 /USNRC, 1977/.

18 The best estimate and range are defined in /Bergström and Nordlinder, 1990b/. The best estimate is the value recommended for freshwater fish muscle in /Poston and Klopfer, 1986/. In /Hoffman, 1988/ factors mostly less than one are obtained but the higher value was chosen in order to be conservative.

19 The range and best estimate are defined in /Bergström and Nordlinder, 1992/. Bioaccumulation factors between 0.9 and 550 l/kg have been recorded from the Marshall Islands /Noskin et al, 1981/ whereas factors for freshwater fish have been reported from i.a. /Vanderborght, 1985/. Also here the variation was large; from 0.04 l/kg up to above 200. According to /Eyman and Trabalka, 1980/ the factor decreases with increasing position in the food web. They recommend a factor of 250 for bottom dwelling fishes, a factor of 25 for plankton feeding species and a factor of 5 for piscivorous species. It is mainly piscivorous species which are consumed by humans, but to be conservative a best estimate of 30 has been used.

20 The range and best estimate are defined in /Bergström and Nordlinder, 1992/. The available amount of data concerning americium is scarce. /Poston and Klopfer, 1986/ recommend a value of 100 l/kg for freshwater fish. In the same reference, values of bioaccumulation factors depending on the trophic level of the fish are also presented; 2 500 l/kg for bottom dwelling fish, 250 for plankton feeding species and 50 for piscivorous species. It is mainly piscivorous species which are consumed by humans, and in order not to underestimate the uptake in fish a best estimate of 100 has been used.

21 The best estimate and range are defined in /Bergström and Nordlinder, 1992/. The amount of data concerning bioaccumulation of curium is scarce. /Eyman and Trabalka, 1980/ recommend bioaccumulation factors related to the throphic level of the fish species; 50 l/kg for piscivorous fishes and 250 for plankton feeding species. As it is mainly piscivorous species which are consumed by humans a best estimate of 50 has been used.

Table A-15. Element specific bioaccumulation factors for freshwater crustaceans and for marine water plants ([Bq/kg w.w.]/[Bq/l]) /Thompson et al, 1972/.

	Freshwater crustaceans (l/kg w.w.)				Marine water plants (l/kg w.w.)			
Element	**B.E**	**Distr**	**Min**	**Max**	**B.E**	**Distr**	**Min**	**Max**
H	1E+0	LT	5E-1	2E+0	1E+0	LT	5E-1	2E+0
C	9E+3	LT	9E+2	1E+4	2E+3	LT	2E+2	1E+4
Cl	1E+2	LT	1E+1	1E+3	1E-1	LT	1E-2	1E+0
Co	2E+2	LT	2E+1	2E+3	1E+3	LT	1E+2	1E+4
Ni	1E+2	LT	1E+1	1E+3	3E+2	LT	3E+1	3E+3
Se	2E+2	LT	2E+1	2E+3	1E+3	LT	1E+2	1E+4
Sr	1E+2	LT	1E+1	1E+3	1E+1	LT	1E+0	1E+2
Zr	7E+0	LT	7E-1	7E+1	2E+3	LT	2E+2	1E+4
Nb	1E+2	LT	1E+1	1E+3	1E+3	LT	1E+2	1E+4
Mo	1E+1	LT	1E+0	1E+2	1E+1	LT	1E+0	1E+2
Tc	5E+0	LT	5E-1	5E+1	4E+3	LT	4E+2	1E+4
Pd	3E+2	LT	3E+1	3E+3	2E+3	LT	2E+2	1E+4
Ag	8E+2	LT	8E+1	8E+3	2E+2	LT	2E+1	2E+3
Cd	2E+3	LT	2E+2	2E+4	1E+3	LT	1E+2	1E+4
Sn	1E+3	LT	1E+2	1E+4	1E+2	LT	1E+1	1E+3
I	5E+0	LT	5E-1	5E+1	1E+3	LT	1E+2	1E+4
Cs	1E+2	LT	1E+1	1E+3	5E+1	LT	5E+0	5E+2
Sm	1E+3	LT	1E+2	1E+4	5E+3	LT	1E+2	1E+4
Eu	1E+3	LT	1E+2	1E+4	5E+3	LT	5E+2	5E+4
Ho	1E+3	LT	1E+2	1E+4	5E+3	LT	1E+2	1E+4
U	1E+2	LT	1E+1	1E+3	7E+1	LT	7E+0	7E+2
Pu	1E+2	LT	1E+1	1E+3	3E+2	LT	3E+1	3E+3
Am	1E+3	LT	1E+2	1E+4	5E+3	LT	1E+2	1E+4
Cm	1E+3	LT	1E+2	1E+4	5E+3	LT	1E+2	1E+4

Appendix B

In this section ecosystem specific dose conversion factors, EDFs (Sv/Bq) for the coastal model used for the present conditions at the SFR area are presented for selected radionuclides. These factors are useful for comparisons with earlier studies /Bergström et al, 1999/ and were calculated by running the model with constant unit releases during 1 000 years. Equilibrium is established after about 1–3 years, depending on radionuclide.

Table B-1. Ecosystem specific dose conversion factors, EDFs (Sv/Bq) for the coastal model used for the present conditions at the SFR area.

Nuclide	EDF (Sv/Bq) at equilibrium
H-3	2.7E-21
C-14 inorg	1.3E-18
C-14 org	1.3E-18
Cl-36	1.4E-19
Co-60	5.9E-19
Ni-59	1.7E-20
Ni-63	4.1E-20
Se-79	1.0E-17
Sr-90	1.1E-18
Zr-93	6.2E-20
Nb-93m	1.1E-20
Nb-94	1.5E-19
Mo-93	1.0E-19
Tc-99	1.2E-20
Pd-107	9.3E-22
Ag-108m	8.4E-19
Cd-113m	5.8E-18
Sn-126	5.9E-18
I-129	1.5E-17
Cs-135	5.8E-19
Cs-137	3.7E-18
Sm-151	4.6E-21
Eu-152	1.9E-19
Eu-154	2.8E-19
Ho-166m	9.5E-20
Pu-238	3.9E-18
Pu-239	4.3E-18
Pu-240	4.3E-18
Pu-242	4.1E-18
Am-241	1.2E-17

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