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Models for dose assessments

Modules for various biosphere types

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Abstract

The main objective of this study was to provide a basis for illustrations of yearly dose rates to the most exposed individual from hypothetical leakages of radionuclides from a deep bedrock repository for spent nuclear fuel and other radioactive waste. The results of this study will be used in the safety assessment SR 97 and in a study on the design and long-term safety for a repository planned to contain long-lived low and intermediate level waste.

The repositories will be designed to isolate the radionuclides for several hundreds thousands of years. In the SR 97 study, however, hypothetical scenarios for leakage are postulated. Radionuclides are hence assumed to be transported in the geosphere by groundwater, and probably discharge into the biosphere. This may occur in several types of ecosystems. A number of categories of such ecosystems were identified, and turnover of radionuclides was modelled separately for each ecosystem. Previous studies had focused on generic models for wells, lakes and coastal areas. These models were, in this study, developed further to use site-specific data. In addition, flows of groundwater, containing radionuclides, to agricultural land and peat bogs were considered. All these categories are referred to as modules in this report. The forest ecosystems were not included, due to a general lack of knowledge of biospheric processes in connection with discharge of groundwater in forested areas.

Examples of each type of module were run with the assumption of a continuous annual release into the biosphere of 1 Bq for each radionuclide during 10 000 years. The results are presented as ecosystem specific dose conversion factors (EDFs) for each nuclide at the year 10 000, assuming stationary ecosystems and prevailing living conditions and habits. All calculations were performed with uncertainty analyses included. Simplifications and assumptions in the modelling of biospheric processes are discussed.

The calculations showed that it is not necessarily exposure via a well which gives the highest EDFs. The highest values were in general obtained for peat bogs and agricultural land. The biological parameters contributed in general considerably to the uncertainties in calculated EDF-values. The interval of EDF-values varied due to the ecosystems studied where the intervals were widest for agricultural land and peat bogs.

The use of modules may be seen as a step towards applying site specific data to biosphere model calculations.There is, however, a need for further development of the module concept for use in biosphere safety assessments.

Sammanfattning

Huvudsyftet med denna studie är att utarbeta en metod för att illustrera den årliga dosraten till den mest exponerade individen, från hypotetiska läckage av radionuklider från ett djupförvar för använt kärnbränsle och annat radioaktivt avfall. Resultaten från studien kommer att användas i säkerhetsanalysen SR 97 och i en studie om utformning och långsiktig säkerhet för ett planerat förvar för långlivat låg- och medelaktivt avfall.

Förvaren kommer att utformas så att de isolerar radionuklderna för hundra tusentals år. I SR 97 antas emellertid hypotetiska scenarier för läckage av radionuklider. Dessa antas därvid transporteras med grundvatten genom geosfären och antagligen nå biosfären. Detta kan ske i olika typer av ekosystem. Ett antal kategorier av ekosystem till vilka radionuklider från geosfären kan utmynna har identifierats och omsättningen av radionuklider modellerades separat för varje nuklid. Tidigare studier har fokuserats på generiska modeller för brunnar, sjöar och kustområden. Dessa har utvecklats vidare för hantering av platsspecifika data. Denna studie inkluderar dessutom ett kontaminerat grundvattenflöde till jordbruksmarker och torvmossar. Alla dessa kategorier benämns moduler i denna rapport. Skogsekosystemet ingår inte i denna studie, eftersom kunskaper om processer i samband med utflöde av grundvatten till skogsmark saknas.

Beräkningsexempel har gjorts för varje typ av modul med antagande av ett årligt utsläpp av 1 Bq av varje radionuklid. Beräkningarna omfattar en tidsperiod på 10 000 år. Resultaten presenteras som ekosystem specifika dosomvandlingsfaktorer (EDFar).för varje nuklid vid 10 000 år. Ekosystemen antas ha konstanta förhållanden under de 10 000 åren och dagens levnadsförhållanden och vanor antas gälla. Alla beräkningar har utförts med osäkerhetsanalyser. Förenklingar och antaganden i bl a processbeskrivningar diskuteras i rapporten.

Beräkningarna visade att det inte alltid är exponering via en brunn som ger den högsta exponeringen. De högsta EDF-värdena erhölls i allmänhet för utsläpp till torvmossar eller underifrån till jordbruksmark. De biologiska parametrarna bidrog i stor utsträckning till osäkerheterna i beräknade EDFar. Intervallet för EDF-värdena varierade avsevärt beroende på vilken typ av ekosystem som behandlades och de var vidast för utsläpp till torvmossar och jordbruksmark.

Användande av modulerna kan ses som ett steg mot användande av platspecifika data i biosfärmodeller. Det finns emellertid behov av att utveckla modulkonceptet vidare för att användas i biosfärsberäkningar.

Executive summary

Safety analyses for disposed radioactive waste include several consecutive steps. The last step is to illustrate the radiation doses to humans and biota from calculated hypothetical leakages of radionuclides from a repository. A useful tool for this is presented and discussed in this report.

The main objective of this study was to provide a basis for illustrations of yearly dose rates to the most exposed individual from hypothetical leakages of radionuclides from a deep bedrock repository for spent nuclear fuel and other radioactive waste. The results of this study will be used, in order to be able to consider dispersion in the biosphere, in the safety assessment SR 97, and in a study on the design and long-term safety for the part of the repository planned to contain long-lived low and intermediate level waste.

A schematic illustration of the method to use modules in illustration of doses to man from leakages of radionuclides by groundwater to a specific site is shown in Figure 1.

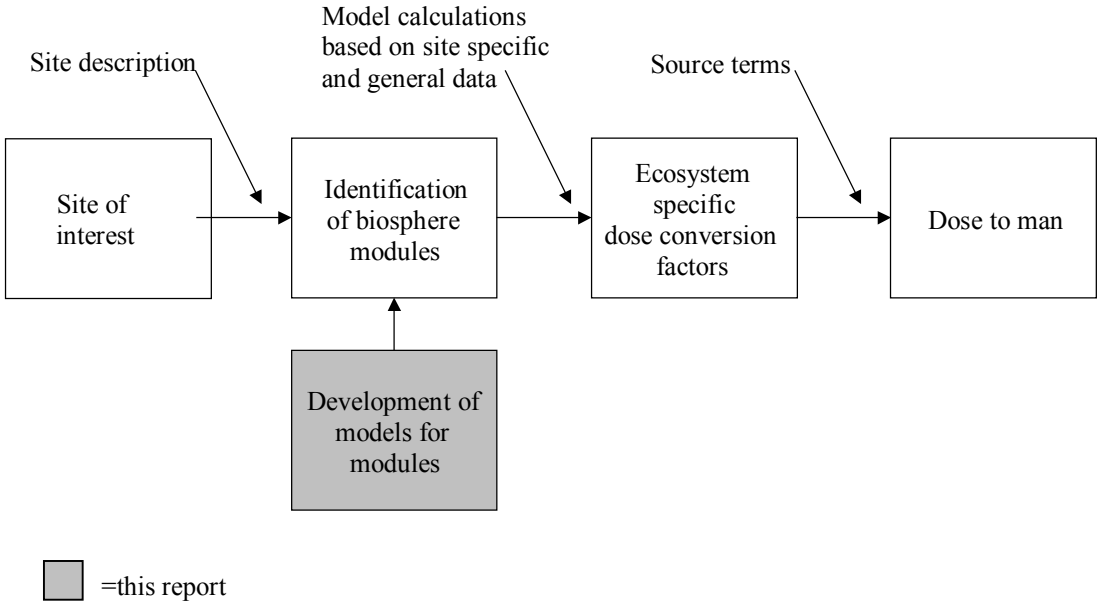

Figure 1 *The different steps in the process which result in estimates of dose to the most exposed individual. The shadowed box represents the part treated in this report.*

Conceptual models were designed for six modules: well, lake, running waters, coastal area, agricultural land and peat bog. In addition, irrigation was defined as a sub-module to be included in the other modules when appropriate. The forest ecosystem was not included, but it should be considered in future model development.

Complex biospheres were subdivided into modules, assumed to be homogeneous regarding type of ecosystem, land use and critical groups. Exchange of material and radionuclides between the biospheric components of the modules were described by rate constants expressed as turnover per year, while the exposure to humans were obtained from ”steady-state” factors. It was beyond the scope of this report to deal in detail with all processes. Several complicated interactions were therefore simplified because of lack of knowledge to be able to consider them explicitly. The effect of many processes was included implicitly by using empirical data for the transfer of radionuclides within the biosphere, however.

Examples of each type of module were run with the assumption of a continuous annual release into the biosphere of 1 Bq each for 44 radionuclides, leading to ecosystem specific dose conversion factors (EDFs). The calculations were performed for a time period of 10 000 years during which constant ecosystems conditions were assumed with prevailing living habits and practices. The time for which maximum exposure was reached during the 10 000 years varied depending on module and nuclide properties. For poorly sorbing elements and nuclides with relatively short half-lives, this time was less than 100 years, while it takes longer for the elements that are most affected by retardation in the geosphere, and for nuclides with long half-lives. All calculations were performed with uncertainty analyses, in such a way that all input parameter values, except for dose conversion factors, were given specific statistical distributions. This yielded results with a range of variation, and also gave possibilities to identify those processes and parameters that dominated the uncertainty of the calculated doses. The results show that modules may be a useful tool to rank discharge areas according to ensuing doses. The calculations show that it is not necessarily releases to a well that gives the largest contribution to the total dose. This may instead be the case for areas with high potential accumulation of radionuclides such as peat bogs.

The best way of studying model confidence is to compare model results against independent data sets. However, this is not possible for this type of study. The uncertainty is due to e.g. simplifications and lack of knowledge of natural phenomena, model simplification, incomplete description of the ecosystem, uncertainties of measured data etc. Uncertainties in the specification of the scenario comprise the whole methodology applied for the assessments. A basic concept in all models set up for the modules is that the biosphere was divided into compartments between which there was assumed an annual transfer of water and thereby material. The compartments were assumed to be physical areas with the same properties, for example chemistry and that elements were homogeneously distributed within them. This is never the case in reality, which leads to uncertainties in the conceptual model. Typical such are the use of annual average values for processes. Simplifications and assumptions in process descriptions are discussed in the report. Uncertainites due to computational errors are of minor importance, due to the use of properly verified codes. The major sources of uncertainties in results were in connection with conceptual modelling and due to the selection of parameter values. Especially the biological parameter showed up to be important.

The use of modules may be seen as an improvement towards applying site specific data to biosphere model calculations. There is, however, a need for a further development of the module approach to biosphere safety assessments. The present models do not consider the evolution with time of each module, and they are not linked to one another. A module for forest ecosystems is required as forests are the most common ecosystem in Sweden. Interactive studies of specific sites for model calculations can give input on how a real biosphere may be divided into the modules specified here, and on how site specific data may be obtained and applied.

The identification of modules at some selected sites, called Aberg, Beberg and Ceberg, are described by Nordlinder et al (1999). EDFs for the three sites are also presented in that report.

The EDFs are based on a continuous release of 1 Bq/year during 10 000 years for each radionuclide. The results may therefore be used to present doses to humans at the time of 10 000 years from the start of continuous releases of radionuclides from a repository. Should the results be used to calculate doses at earlier times, e.g. after 1 000 years, this may lead to an overestimation of the doses for most nuclides. This is due to a build-up of concentrations of some nuclides in the biosphere during long times, and because steady-state conditions are not prevailing during the time periods mentioned above.

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1 Introduction

Safety analyses for disposed radioactive waste include several consecutive steps of which the last aims to illustrate the radiation doses to humans and environment from calculated leakages of radionuclides from a repository. The Swedish concept for disposal of spent nuclear fuel is a deep bedrock repository with multiple barriers, designed to protect the environment during hundreds of thousands of years (KBS-3, 1983). Despite of this, the safety analysis should show consequences of failures in the barriers that may lead to releases of radionuclides into the biosphere. Turnover and accumulation of radionuclides in the biosphere from the point of discharge to dose to humans should therefore be calculated (SSI, 1998). This is performed by applying radio-ecological models of varying complexity for the ecosystem to be studied (Bergström, 1983). Description of such models can be found in Davis et al, (1993), Klos et al, (1996) and BIOMOVS II, (1996b).

The biosphere is divided into a number of categories of ecosystems into which a release of radionuclides from a repository may discharge. Given the information about the points and time of the release of radionuclides and combining it with information about the current biosphere, we can make an estimate of what recipient and ecosystem should be used when performing dose calculations.

Previous studies emphasised generic models for wells, lakes and coastal areas (Bergström & Nordlinder, 1990a and Bergström & Nordlinder, 1990b). These models were further developed for use of site-specific data. In this study the flow of groundwater to agricultural land and peat bogs is considered as well. All these categories are designated as modules in this report.

The main objective with this study is to provide a basis for illustrations of yearly dose rates to the most exposed individual, from hypothetical leakages of radionuclides from a repository for spent nuclear fuel and other radioactive waste. Site-specific dose conversion factors (EDFs) for each module and radionuclide of potential interest were calculated for unit releases during 10 000 years. The result from this study will be used in the safety assessment SR 97 (SKB, 1999a) and in a study about the design and long-term safety for the repository for long-lived low and intermediate level waste (SKB, 1999b).

The following steps were taken:

- Identification of potential discharge areas for groundwater to the biosphere.
- Design of conceptual models for ecosystems at points of groundwater discharge.
- Computation of nuclide specific factors for conversion of annual releases of 1 Bq to doses to the most exposed individual in the different ecosystems after 10 000 years.
- Calculation and describtion of the uncertainties in results due to variations in parameter values.

The system of modules is developed in such a way that site-specific conditions are simple to consider with site-specific parameter values. For applying the dose rates from unit releases obtained from the modules in an appropriate way it is necessary to identify the modules that should be applicable for the area to which the contaminated groundwater may enter. The identification of modules at some selected sites, called Aberg, Beberg and Ceberg, is described by Nordlinder et al (1999).

Basic data such as half-lives and dose conversion factors for the nuclides are given in Chapter 2. A description of models for various typical ecosystems is given in Chapter 3. Results with discussions are given in Chapter 4. Various aspects of uncertainties with emphasis on uncertainties in processes and data are discussed in Chapter 5, while finally Chapter 6 gives general conclusions and recommendations regarding the use of the modules. Biological parameters and other element specific values are shown in Appendix A. Appendix A also includes a brief literature survey of root-uptake factors. The details for making dose calculations are presented in Appendix B.

2 Nuclides

The nuclides considered in this study were selected from the most dominant in spent nuclear fuel and long-lived low and intermediate level waste. A common property is that they all decay slowly enough (see Table 2-1) to reach the biosphere in case of accidental leakages from a repository. The radionuclides cover a spectrum of element specific properties.

Generation of daughter radionuclides in the biosphere was not considered in this study. This was justified from earlier calculations performed for long periods of times, up to several millions of years, showing that daughter nuclides to actinides gave minor contribution to the total dose (Bergström & Nordlinder, 1989). The 10 000 years perspective considered in this study also leads to insignificant generation of other daughter nuclides due to the long half-lives of their precursors.

EDFs at year 10 000 were calculated for each one of the nuclides considered. During the first 10 000 years after closure C-14, Cl-36, Ni-59, Mo-93, I-129, Cs-135 and Np-237 will give the highest doses from a failure of the fuel encapsulation (Lindström & Lindgren, 1999 and SR 95, 1996). Therefore the results from these radionuclides will be treated in more detail.

2.1 Dose conversion factors

Human can be exposed to radioactivity by external irradiation and internally by ingestion or inhalation of radionuclides. The internal radiation doses are calculated as a product of the intake rates and the nuclide-specific dose conversion factors, expressed as Sv per ingested or inhaled Bq. The International Commission of Radiation Protection (ICRP) has used dosimetric models and biokinetic data for obtaining these dose conversion factors (ICRP 23, 1974 and ICRP 42, 1984). Dose conversion factors for inhalation, type M, were selected as recommended in ICRP 71 (1995). The external dose conversion factors are usually given as Sv per hour per Bq per m^2 or per kg soil. These were transformed to Sv per hour per Bq per m^3 in the values given in Table 2-1.

Table 2-1 Half-lives and dose conversion factors for treated nuclides (Ingestion and Inhalation from, ICRP 1996, Vol 72 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	4.5E-11
Be-10	β	1 600 000	0	1.1E-9	9.6E-9
C-14	β	5 730	0	5.8E-10	2.0E-9
Cl-36	β	301 000	0	9.3E-10	7.3E-9
Co-60	β, γ	5.3	2.8E-13	3.4E-9	1.0E-8
Ni-59	β	75 000	0	6.3E-11	1.3E-10
Ni-63	β	96	0	1.5E-10	4.8E-10
Se-79	β	1 130 000	0	2.9E-9	2.6E-9
Sr-90	β	29	0	2.8E-8	3.6E-8
Zr-93	β	1 530 000	0	1.1E-9	1.0E-8
Nb-94	β, γ	20 300	1.6E-13	1.7E-9	1.5E-9
Mo-93	EC	3 500	0	3.1E-9	5.9E-10
Tc-99	β	213 000	0	6.4E-10	4.0E-9
Pd-107	β	6 500 000	0	3.7E-11	8.5E-11
Ag-108m	γ	127	1.6E-13	2.3E-9	7.4E-9
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	1.5E-8
Cs-135	β	2 300 000	0	2.0E-9	3.1E-9
Cs-137	β, γ	30	5.6E-14	1.3E-8	9.7E-9
Sm-151	β	90	4.6E-18	9.8E-11	4.0E-9
Ho-166m	γ	1 200	1.6E-13	2.0E-9	1.2E-7
Pb-210	β	22	7.2E-17	6.9E-7	1.1E-6
Ra-226	α	1 600	6.0E-16	2.8E-7	3.5E-6
Ac-227	α	22	0	1.1E-6	2.2E-4
Th-229	α	7 340	2.0E-15	4.9E-7	1.1E-4
Th-230	α	77 000	3.5E-17	2.1E-7	4.3E-5
Th-232	α	14 050 000 000	1.5E-17	2.3E-7	4.5E-5
Pa-231	α	32 760	1.8E-15	7.1E-7	1.4E-4
U-233	α	158 500	5.9E-17	5.1E-8	3.6E-6
U-234	α	244 500	3.1E-17	4.9E-8	3.5E-6
U-235	α	703 800 000	1.1E-14	4.7E-8	3.1E-6
U-236	α	23 415 000	0	4.7E-8	3.2E-6
U-238	α	4 468 000 000	0	4.5E-8	2.9E-6
Np-237	α	2 140 000	1.8E-15	1.1E-7	2.3E-5
Pu-238	α	88	1.3E-17	2.3E-7	4.6E-5
Pu-239	α	24 065	6.6E-18	2.5E-7	5.0E-5
Pu-240	α	6 537	0	2.5E-7	5.0E-5
Pu-242	α	376 300	0	2.4E-7	4.8E-5
Am-241	α	432	1.1E-15	2.0E-7	4.2E-5
Am-242m	α	152	1.5E-17	1.9E-7	3.7E-5
Am-243	α	7 380	2.9E-15	2.0E-7	4.1E-5
Cm-244	α	18	0	1.2E-7	2.7E-5
Cm-245	α	8 500	3.2E-15	2.1E-7	4.2E-5
Cm-246	α	4 730	0	2.1E-7	4.2E-5

3 Modelling methods for modules

The inflow areas for groundwater carrying radionuclides, treated in this report, are lakes, running water, coastal areas, agricultural land and peat bogs. The forest ecosystem is excluded due to lack of information for other radionuclides than for cesium. However, it should be considered in future assessments. Models were designed for the modules mentioned above and for a well for calculation of exposure of radionuclides to humans. Earlier experience and current knowledge of dominant exposure pathways for humans were used. Irrigation was defined as a sub-module to be included in the other modules. Several complicated interactions were simplified due to incomplete knowledge for considering them explicitly. However, many processes were included implicitly, due to the use of empirical data for the transfer of radionuclides within the biosphere. This is further discussed in Section 5.2.

Information relevant for all modules is presented in Section 3.1. In Sections 3.2 - 3.8 each one of the modules is described regarding nuclide turnover and exposure pathways specific for that module. Common data such as distribution coefficients, biological uptake factors and consumption data are all given in Appendices A and B. A discussion of uncertainties in the description of processes and in data is included in Chapter 5.

3.1 Common features

The surface ecosystem was classified into the modules, lake, running water, agricultural land, peat bogs, coastal areas and wells. The method for classification of modules is described further in Nordlinder et al (1999). Each square (250x250 m) was assumed to be a homogenous ecosystem. The exchanges of material and radionuclides between the components within each module were described by rate constants expressed as turnover per year while the exposure part was obtained from ”steady-state” factors. This lead to a number of differential equations which were solved numerically with ACTIVI from the computer code BIOPATH (Bergström et al, 1982 and Bergström et al, 1995). The PRISM-system (Gardner et al, 1983) was used to generate randomly drawn values according to the statistical distribution, given for each parameter. However, dose factors were not varied. The processes and parameters that dominate the uncertainty in the calculated doses were identified.

3.1.1 Discharge points

Typical outflow regions where groundwater discharges are coastal areas, lakes, running waters and wet lands as peat bogs. The source term to these primary recipients will hence be the same as the output from the stream-tubes (SR 95, 1996 and SKB, 1999a). There is a strong seasonal variation coupled to the flow of groundwater in similarity to surface water (Eriksson, 1985). In this study, however, mean annual values and conditions were applied. This may be further improved as low basin discharges can be

expected to have higher concentrations of substances that are important from a chemical viewpoint (Eriksson, 1985).

General properties of discharge areas, such as good conditions for vegetation and thereby high turnover rates of organic matter in aerated groundwater, are important factors for the future fate of radionuclides reaching such areas.

3.1.2 Most exposed individual

All potential releases of radionuclides from a deep bedrock repository will occur in a distant future. The residence and habits of individuals that may be exposed are unknown. Estimates of future doses may therefore be seen as tentative. According to current Swedish regulations (SSI, 1998), a risk of 10^{-6} with a range of 100, is allowed for a member of a group of people, large enough that it is possible that they may be exposed from 10 repositories. On the other hand, it is possible to show the risk for the most exposed individual, which is accordingly a value ten times higher. Such an approach was taken in these calculations. All EDFs were calculated by assuming that all foodstuffs was produced locally in areas where one might expect the highest concentrations of radionuclides, i.e. at the entrance points to the biosphere from the geosphere. The individuals were also assumed to live in such inflow areas. The supposition of self-sustaining households may be seen as a conservative assumption compared to the situation in the society of today. The same consumption data were used as in a previous generic study of "dose conversion factors" (Bergström & Nordlinder, 1990a and Bergström & Nordlinder, 1990b)

3.1.3 Exposure pathways

Humans were the main target in these assessments. They can be exposed internally as well as externally from radiation, see Figure 3-1.

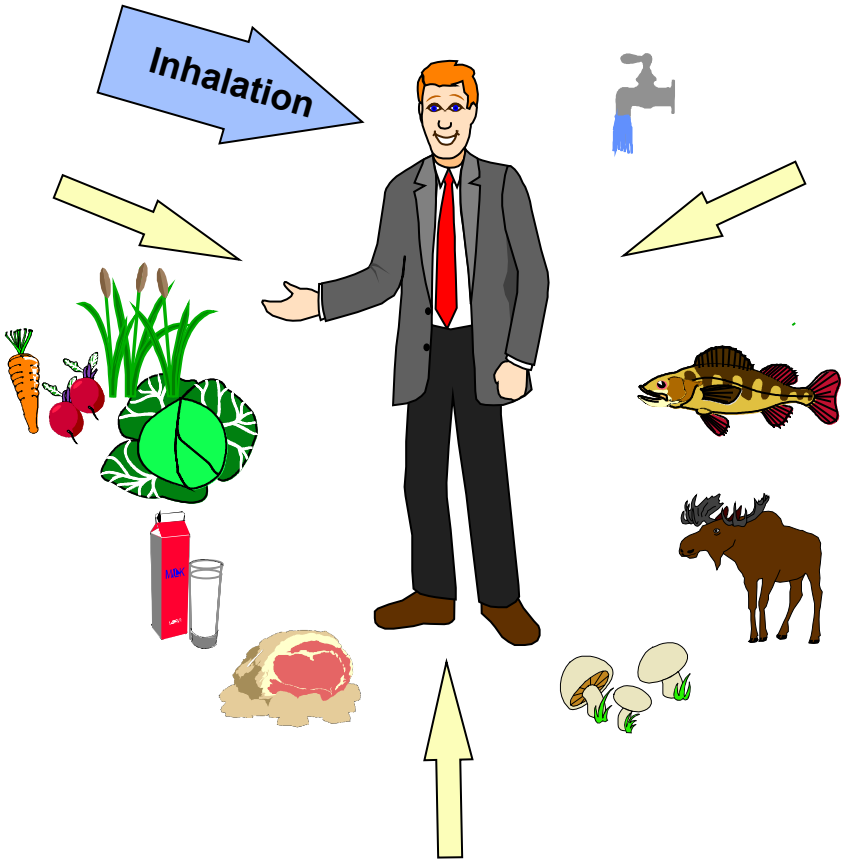

Figure 3-1 *Potential exposure pathways to man; internal by inhalation and consumption of water and foodstuff; external (white arrows) from radionuclides in air and on the ground.*

External exposure may occur from ground deposition 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 and Bergström, 1983). Hence this was, the only external exposure pathway considered in the modules.

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 moved along the foodchains 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 see Figure 3-2. Contamination my also occur from soil to vegetation due to rain-splash. The surface deposited radionuclides can by translocation be transferred to edible parts. In these assesssments vegetation was considered to be contaminated by root-uptake and retention of irrigation water on the surfaces, leading to transfer to edible parts. Most of the root uptake occurs via water in soil. However, roots are also efficient in taking up elements from soil minerals by lowering the pH in their closest surroundings (Wiklander, 1976).

Root uptake factors (also called transfer factors) were obtained from empirical data on concentrations of elements in different species and in the corresponding dry soil. Data for different plants are available for most elements, though the values vary considerably for each element (IUR, 1984).

Values of root-uptake factors are affected by the species, type of soil, climate and element specific properties (IAEA, 1983). In addition, cultivation of soil such as ploughing, fertilisation and irrigation influence the root uptake (IAEA, 1994). A literature review is presented in Appendix A.

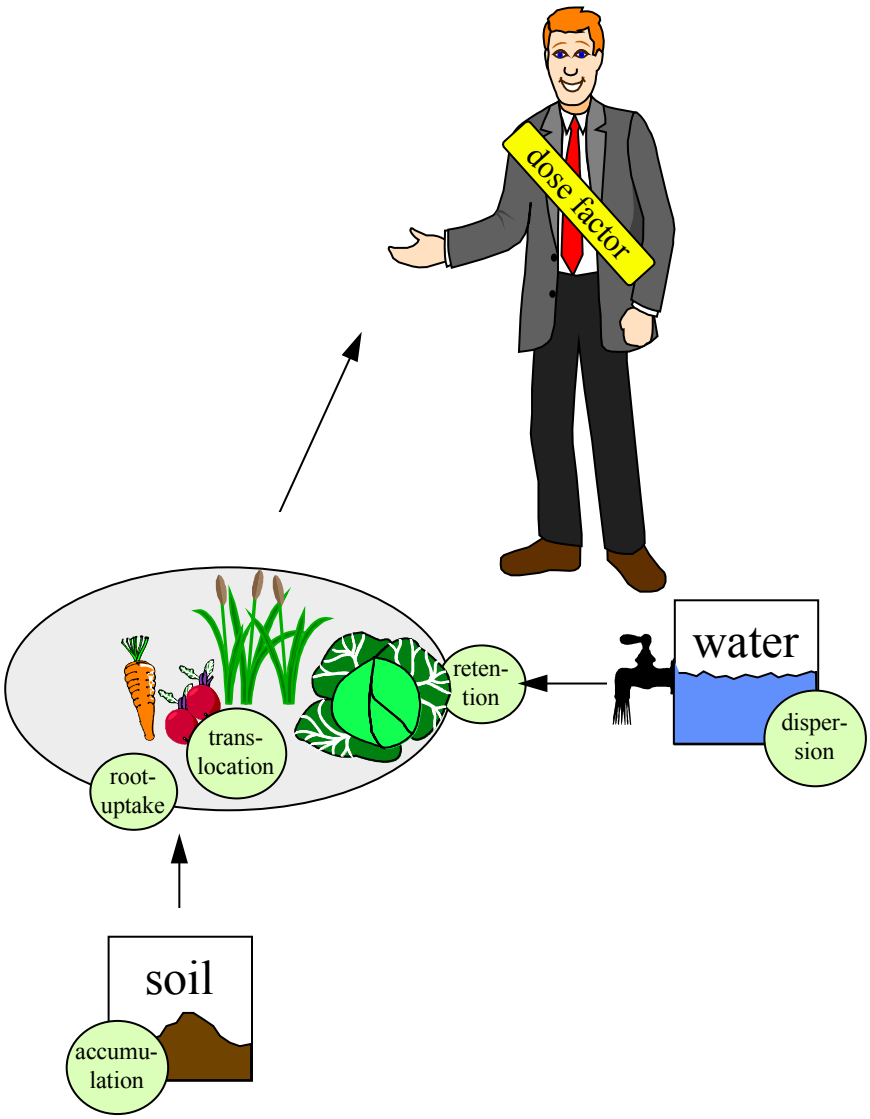

Figure 3-2 Pathways of radionuclide transfer from contaminated soil and water by crops to humans.

In the following assessments crops were 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). Fruits and berries were excluded due to lack of data for most nuclides.

The main pathways for transfer of radionuclides to animals and further on to humans are shown in Figure 3-3. When cows consume fodder and water containing radionuclides a fraction of the radionuclides is transferred to milk, entrails and meat. They may also ingest contaminated soil when grazing. The transfer to milk and meat was decribed by element specific distribution factors, giving the relation between daily intake of a nuclide and its concentration in milk or meat, respectively, expressed as day per litre or day per kilogram.

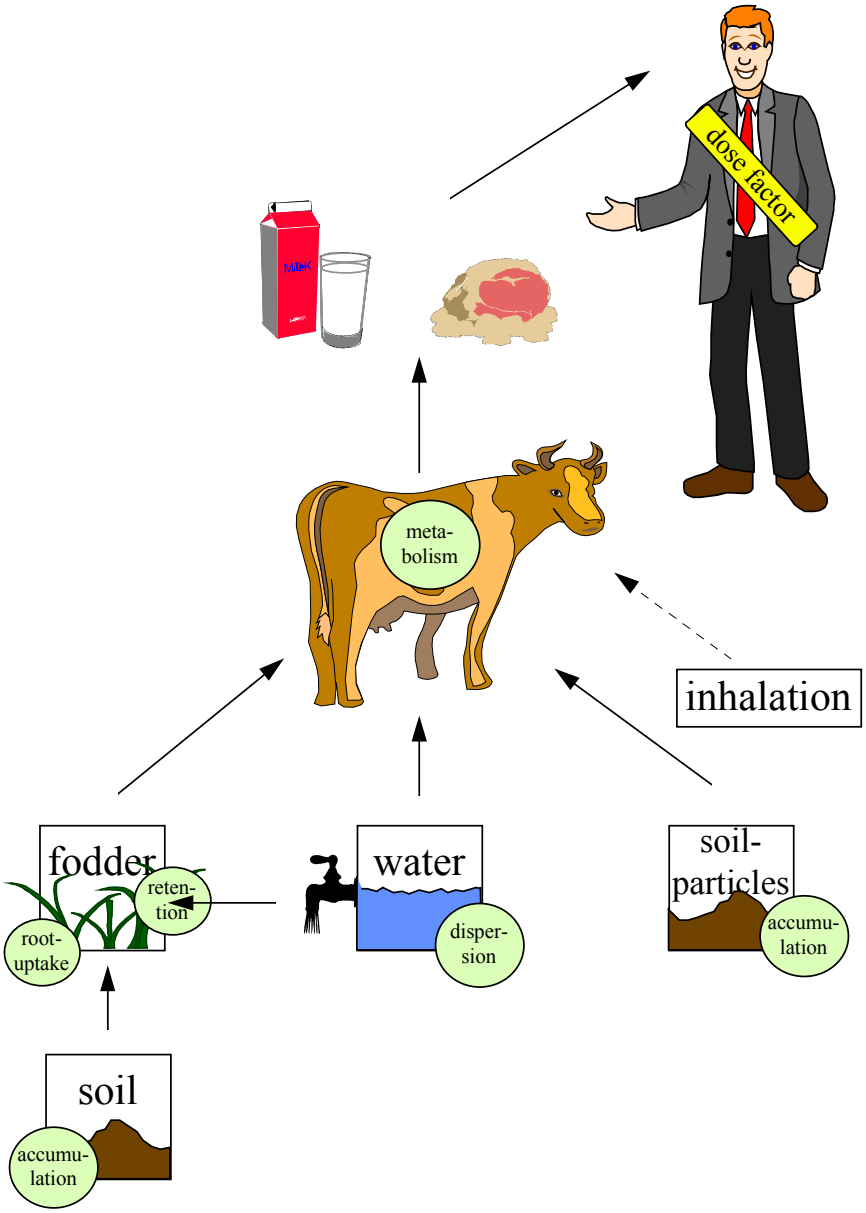

Figure 3-3 Pathways of radionuclide transfer from contaminated soil, water and air by milk and meat to humans.

We have here assumed that beef represents the total annual amount of meat consumed by humans. Consumption of entrails as kidney and liver were excluded in the study. Only milk from cows was considered as cows produce most of the milk for consumption and thereby most cheese is produced from cow milk.

Radionuclides released to surface waters may expose humans due to consumption of aquatic products such as fish, crustacean and algae, in addition to direct consumption of water, see Figure 3-4 below. According to their element specific properties radionuclides may be more or less concentrated along the aquatic foodchains. The uptake in aquatic biota was described by element specific bioaccumulation factors, relating to the ratio between concentration of the element in biota relative to the water, assuming steady-state conditions.

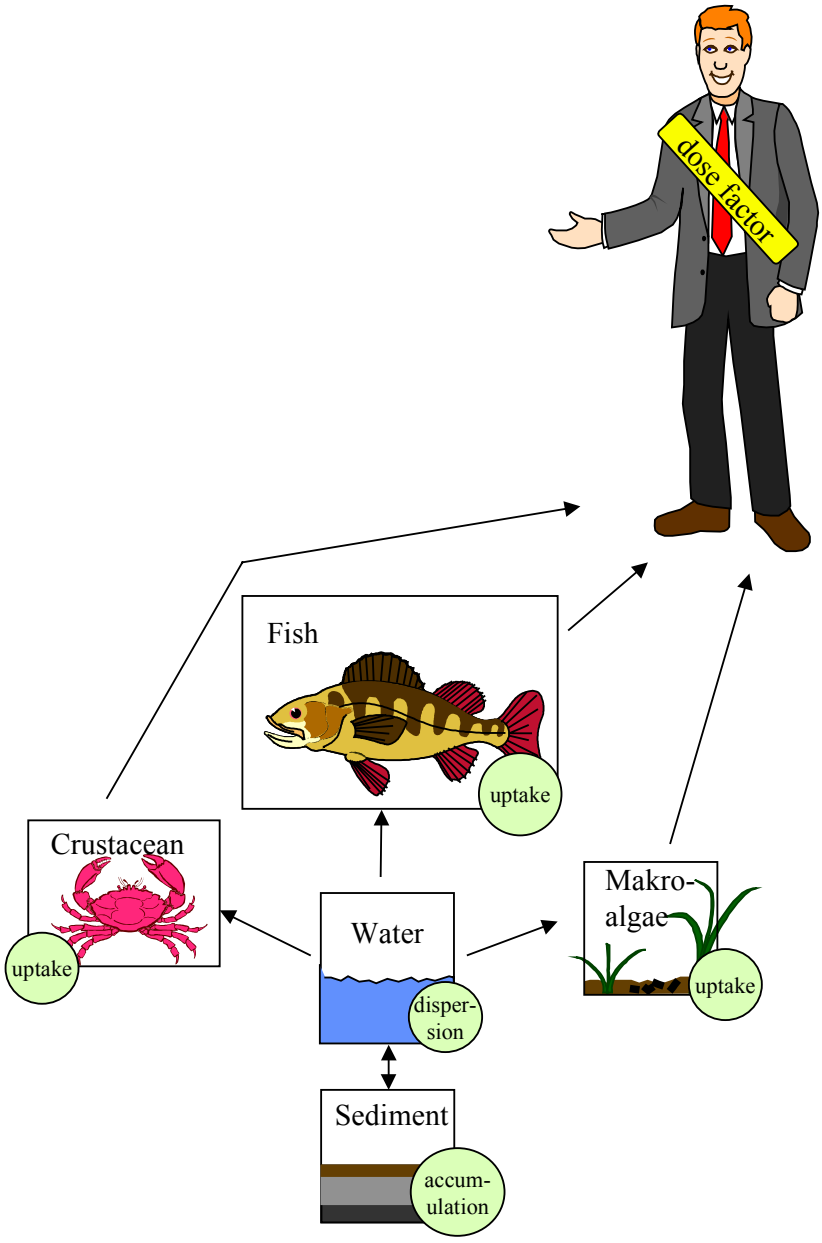

Figure 3-4 Pathways of radionuclide transfer from contaminated water by aquatic products to humans.

3.1.4 General assumptions

Some major assumptions were used in all the modules. They are summarised and briefly commented on below. Some of them are discussed further in Section 5.2.

- The processes affecting the transport of nuclides from the out-flow of stream-tubes to the module of interest are neglected.

The groundwater passes the saturated/unsaturated part of soil before it discharges. During this passage (Freeze & Cherry, 1979) several chemical and biochemical reactions can alter contaminant concentration in groundwater. This retention was neglected in this study.

- Immediate homogeneous mixing of the nuclides in the compartment of release.

According to present practice and in similarity to other studies (Davis et al, 1993) the homogeneous mixing concept was used. Radionuclides entering water compartments were therefore mixed in the totally available volumes. This was justified by the purpose of showing mean doses from a continuous leakage of elements during several years and by the assumption that water and fish from the entire lake were used.

- Homogeneous soil- and peat compartments.

The assumption of homogenous mixture in soil was a simplification. For the top soil of farming land this can be considered valid at a time scale of a decade or more due to ploughing. In undisturbed forests there will be a gradient in the vertical profile. However, such a module was not included in the present system. For peat, the assumption of homogeneity is not valid. However, when peat is used, either as fuel or as farming-land, some mixing due to human activities occurs. The approach of homogeneity is however practical due to the use of data representing average concentrations of respective element in soils.

- Annual mean values for water turnover, precipitation, sedimentation rates etc.

The main targets for the assessment were average annual exposure. Therefore, we used annual average values, though there is a strong seasonal variation in most processes, e.g. runoff. This is justified if the average values used give an annual transfer that corresponds to a time-varying transport in a correct way. This assumption needs further analysis.

- Water balance described by a mean average runoff.

There was a water balance assumed in the modules leading to that the annual water turnover was described by an average runoff (precipitation-evaporation). This volume corresponds to the annual amounts percolating through the soil-layers to groundwater. This was because only a minor amount of precipitation is drained by surface runoff (Grip & Rodhe, 1985).

- No reduction of radionuclides in the compartment due to biological uptake (e.g. in soil due to harvest of crop).

The neglection of loss of radionuclides due to root-uptake and harvesting in agricultural fields was a conservative assumption. Usually, in assessments of this type (BIOMOVS II, 1996b and Davis et al, 1993) the transfer along the foodchains, aquatic as well as terrestrial, is described by factors for steady-state conditions implying that this biological transport does not influence the redistribution of radionuclides when calculating the doses in the models, see further discussion in Chapter 5.

- The transfer of nuclides in foodchains was described by ”steady-state” factors.

The steady-state conditions in the exposure pathway calculations can be assumed when the growth cycle for organisms is shorter than one year and annual doses are to be considered.

- No changes of the biosphere, current conditions prevailing.

IAEA as well as the Swedish Radiation Protection Institute recommend that current biosphere conditions and man’s behaviour should be used to show if consequences would be acceptable. Future changes such as land rise, glaciations and permafrost conditions can be considered by applying different scenarios. In SR 97 (SKB, 1999a) the effect of landrise on future ecosystems is considered.

- Independent on pathway of intake of radionuclides the same fraction of radionuclides was distributed to milk and meat, respectively.

The same value was used for describing the concentration of the elements to milk and meat independent on pathway of animal intake as no data are available for different types of intake (e.g. water, fodder or soil).

- No loss of radionuclides due to food preparation and water treatment.

Food preperation such as peeling, washing and warming will reduce the amount of radionuclides for consumption. Such losses were not considered.

3.1.5 Ranges used in the variation analyses

Uncertainty analyses were included in all calculations. All parameters were assigned statistical distributions and ranges from which random values were generated. Usually information on type of distribution is lacking due to few data for a specific parameter. In those cases the parameters were given triangular or logtriangular distributions, as is recommended by IAEA (1995). The results from uncertainty analyses are less sensitive to distribution type than to the used ranges, which justifies this approach, especially when comparing triangular to normal distributions and logtriangular to lognormal distributions.

The basis for selection of ranges is summarised below for most parameters.

Runoff ranges (e.g. see Tables 3-1 and 3-5) were taken from 30-year statistics of variation in precipitation according to statistics from the Swedish Hydrological and Meteorological Institute.

The variation in lake **morphymetry** for the lake module was estimated to be about 10 %. This uncertainty depends solely on measurement errors and is comparatively low. Many other data vary over two orders of magnitude. **Sedimentation rates** and **concentrations of suspended matter** were given much wider ranges due to lack of site specific data for those parameters. The ranges (Evans, 1986) represent variability for eutrophic and oligotrophic lakes, respectively.

For **soil conditions** such as plough layer, porosity and density general data on variability in various soil types were used due to lack of site specific information.

The variations for **irrigation** rates and **retention** of irrigation waters on vegetation surfaces were subjectively determined. Extreme values of irrigation are not plausible for Swedish conditions due to the climate. The maximum values for retention was, owever, conservatively biased as even the mean value is based on the maximum interception capacity.

The factors used to describe the transfer of radionuclides to biota, **bioaccumulation and distribution factors**, were element specific and the used ranges varied due to availability of information, see Tables A-1 – A-9. When literature data on ranges were lacking, they were usually set to a factor of ten up and down from the mean values. The ranges of crop yield values corresponded to averages for central Sweden and were therefore quite narrow.

The **K_d-values** describing the element distribution between solid and dissolved fractions in water, soil and peat, had in general wide ranges, mostly within two orders of magnitude, showing the lack of confidence in data.

Erosion rates and **bioturbation** rates were varied subjectively in order to symbolise mean values over larger areas. Erosion, in particular, may vary considerably from one field to another due to location. However, extreme conditions were not considered.

Normal distribution was assumed for **consumption data** for man and cattle with standard deviations corresponding to 10 % of the mean values. The group was in this study defined as individuals with similar consumption habits. Narrow ranges were therefore adopted. The ranges for cattle's consumption of water and foodstuffs were also narrow (T. Morén, Lövsta Gård, personal communication).

3.2 Sub-module irrigation

Irrigation is a sub-module to the well, running waters and lake modules. This sub-module is included in those modules if giving the area and quantity of water per unit area, used for irrigation per year. The main feature for this module is soil. Flora and fauna are also parts of the module but only as exposure pathways to man. All types of

gardening and farming are included, but for practical reasons some selected crops see Section 3.1.3 are considered to represent all types of crops used for consumption.

Irrigation results in a transfer of nuclides (dissolved or sorbed on particles) from the water body to plants and soil, see Figure 3-5.

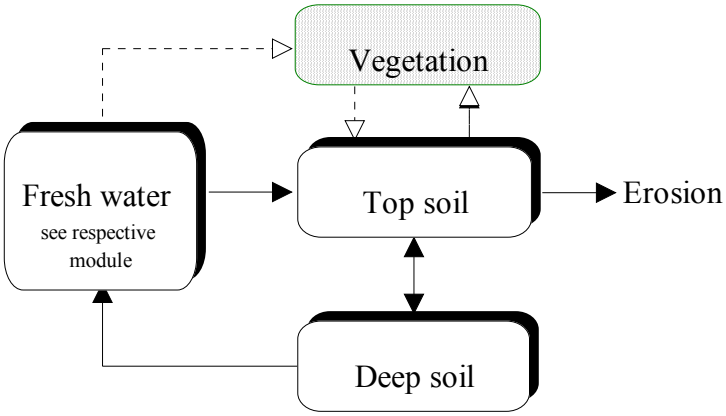

Figure 3-5 Compartment system for the sub-module irrigation.

3.2.1 Irrigation

The rate constant describing transfer of radionuclides from surface water (sw) to soil (s) due to irrigation is obtained from the following expression

$$TC_{sw,s} = \frac{IRR}{VW} \cdot A$$

where

IRR = Annual irrigation rate [$m^3/(m^2 \cdot year)$]
VW = Volume of water from which irrigation water is taken [m^3]
A = Area of irrigated soil [m^2]

The out-flow of radionuclides from soil is assumed to be to the well or to the surface water from which the irrigation water is taken. The consequence of this feedback is that no radionuclides are lost from the system due to irrigation.

3.2.2 Initial retention

Retained irrigation water with its contents of radionuclides contaminates the surfaces of vegetation. It was handled as a non-continuous process where about 3 mm of water was 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) 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 the weathering half-life (IAEA, 1994). A new irrigation occasion causes an additional retention while there is an exponential decrease during the time

passing between the irrigation occasions. The amounts of radionuclides on the surfaces of vegetation (veg) are therefore a function of frequencies of irrigation and time for harvest, see the expression below.

$$Y_{veg}(t) = Y_{veg}(0) \cdot \exp\left(- t \cdot \frac{\ln 2}{T_{½w}} \right)$$

where

$Y_{veg}(0)$ =Initial radionuclide content on a surface after irrigation [Bq/m^2]

t =Time after irrigation [days]

$T_{½w}$ =Empirically found weathering half-life (15 days during growing season) (IAEA, 1994)

For further description of how the integrations are performed see Appendix B.

Surface deposited radionuclides can be translocated from surfaces of vegetation to the edible parts. This is described by an element and species specific translocation factor. 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 is included for root crops and cereals in this module. A fraction of the radionuclide content was modelled to be transferred to the edible parts at each irrigation occasion (see Appendix A for data). For green vegetables and pasturage the retained radionuclides on the vegetative surface are consumed by man or cattle.

3.2.3 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 (t) to the deep (d) soil compartment is described by a transfer coefficient considering vertical transport due to percolation of water and transport of soil by (earth)worms.

$$TC_{u,d} = \frac{R}{\varepsilon \cdot h_s} \cdot Ret + \frac{BioT}{h_s (1-\varepsilon) \cdot \rho_p}$$

where

$$Ret = \frac{1}{\left[1 + K_d \cdot \rho_p \cdot (1-\varepsilon)/\varepsilon\right]}$$

and

R =Runoff (Precipitation–Evapotranspiration) [m^3/(m^2·year)]

BioT =Bio-transport due to bioturbation [kg/(m^2·year)]

h_s =Depth of top soil plough layer [m]

K_d =Distribution factor, concentration of the radionuclide on solids relative to dissolved [m^3/kg]

ρ_p =Density of soil particles [kg/m^3]

ε =Porosity of plough layer [m^3/m^3]

In the expression above all the pores in the unsaturated zone were assumed to be able to hold water. This simplification may underestimate the migration rates.

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, see Table 3-1, as it is also considered as a reverse transport from deep soil to top soil.

Radionuclides in deep soil are assumed to leak back to groundwater (well), running waters or lake water. This is described by the same expression as above, but with depths and porosities according to deep soil. In the well module this leakage is assumed to be back to the water in the well.

The loss (out) due to erosion (e) is described by m soil lost per year, see the expression below

$$TC_{e,out} = \frac{Eros}{h_s}$$

where

Eros = Erosion rate [m/year]

Radionuclides are transferred by erosion to the surface water module in question. There is no information about erosion rates, thus the values were estimated, see Chapter 5.

For a garden plot loss of soil (out) due to weeding (wed) is assumed

$$TC_{wed,out} = \frac{Rem \cdot A}{h_s (1-\varepsilon) \cdot \rho_p \cdot A}$$

where

Rem = Removal of soil [kg/(m^2,year)]
A = Irrigated area [m^2]

3.2.4 Root-uptake

Nuclides in soil are partly transferred to plants via root uptake. This uptake is described as empirically found transfer factors, which are obtained from measurements of nuclide content in plant and in soil, see Appendix A. They are inversely correlated to the K_d-values in the calculations as high sorption reduces the bioavailability.

3.2.5 Exposure pathways

Human exposure pathways are:

- Consumption of milk and meat from cattle consuming pasturage and soil from irrigated soil
- Consumption of various crops grown on irrigated soil
- Inhalation of dust from irrigated soil
- Intake of soil via for example insufficiently washed vegetables

3.2.6 Data

The Tables 3-1 and 3-2 contain generic values for some physical-chemical parameters when site-specific data are not available. These values were used in the calculations in Chapter 4. In lack of precise data most parameters were assumed to be triangularly distributed, as recommended by IAEA (1989). The ranges are, however, often subjectively determined, which is discussed in Section 5.4. The irrigated farming land area is selected to be large enough for production of the foodstuff. It is also somewhat larger than the squares obtained when dividing an area into modules (Nordlinder et al, 1999).

Table 3-1 General data used when the irrigation sub-module is applied to agricultural soils.

	Abbr	Unit	Mean	Min Triangularly dist	Max Triangularly dist	References
Irrigation	IRR	$m^3/(m^2 \cdot year)$	0.15	0.10	0.20	Linner, 1997*
Irrigation events	n	number of	5	3	7	Assumed values
Irrigation period		days	90	90	90	Assumed values
Irrigation area	A	m^2	100000	98000	110000	Assumed values
Retention of irrigation water	I	m^3/m^2	0.003	0.001	0.005	Persson, 1997
Runoff	R	$m^3/(m^2 \cdot year)$	0.24	0.20	0.28	Lindborg & Schüldt, 1998
Depth of top layer	h_s	m	0.30	0.25	0.35	Haak, 1983
Depth of deeper soil		m	1.0	0.9	1.1	Assumed values
Soil particles density	ρ_p	kg/m^3	2400	2000	2800	Hillel, 1980
Soil porosity, top deeper	ε	m^3/m^3 m^3/m^3	0.4 0.3	0.3 0.2	0.5 0.4	Hillel, 1980
Weathering half-life	$T_{½w}$	day	15	10	20	Hoffman & Baes, 1979
Bioturbation	BioT	$kg/(m^2 \cdot year)$	2	1	3	Müller-Leman and van Dorp, 1996
Erosion	EROS	m/year	0.00010	0.00003	0.00030	Assumed values

Table 3-2 General data used when the irrigation sub-module is applied to a garden plot.

	Abbr	Unit	Mean	Min Triangularly dist	Max Triangularly dist	Reference
Irrigation	IRR	m^3/year	0.15	0.10	0.20	Linner, 1997*
Irrigation events	n	number of	5	3	7	Assumed values
Irrigation period		days	90	90	90	Assumed values
Irrigation area	A	m^2	1000	900	1100	Assumed values
Retention of irrigation water	I	m^3/m^2	0.0003	0.0001	0.0005	Persson, 1997
Runoff	R	$m^3/(m^2 \cdot year)$	0.24	0.20	0.28	Lindborg & Schüldt, 1998
Depth of top layer	h_s	m	0.30	0.25	0.35	Haak, 1983
Depth of deeper soil		m	1.0	0.9	1.1	Assumed values
Soil particle density	ρ_p	kg/m^3	2400	2000	2800	Hillel, 1980
Soil porosity, top deeper	ε	m^3/m^3 m^3/m^3	0.4 0.3	0.3 0.2	0.6 0.4	Hillel, 1980
Weathering half-life	$T_{½w}$	day	15	10	20	IAEA, 1994
Bioturbation	BioT	$kg/(m^2 \cdot year)$	2	1	3	Müller-Leman and van Dorp, 1996
Soil removal	Rem	$kg/(m^2 \cdot year)$	0.4	0.3	0.5	Assumed values

* Harry Linner, SLU, Sweden (pers. comm).

3.2.7 A summary of processes and parameters

Processes and parameters are summarised in Table 3-3 below. The abbreviations in capital letters are used in Chapter 5 treating uncertainty analyses. Comments are also given on how parameter values were obtained for the calculations.

Table 3-3 Processes and parameters used in the calculations for the irrigation sub-module.

Processes and parameters	Comments	Table
Human activities		
Occasions of irrigation per year	General data	3-1, 3-2
Annual amount of water for irrigation	General data	3-1
Consumption rates of milk (CONMI), meat (CONME), cereals (CONCE), vegetables (CONVEG), root-crops (CONROOT), inhalation rates, exposure time	General data	B-2
Physical processes and properties		
Soil porosities, density, depth	General data, extracted from Hillel, 1980	3-1
Runoff (precipitation, evaporation)	Site-specific data	3-1
Retention of irrigation water on plant surfaces	General data	3-1
Soil erosion (EROS)	General data	3-1
Weathering	General data	3-1
Dust concentration in air	General data	B-3
Chemical processes and properties		
Sorption of nuclides on soil particles	Element specific distribution factors, general for soils	A-6
Biological processes and parameters		
Yield-values of pasturage (YPA) and leafy vegetables (YVE)	General data	B-3
Cattle's consumption of pasturage (COCON), cereals and soil	General data	B-1
Cattle's metabolism, transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Root-uptake factors for cereals (BFC), leafy vegetables (BFV), root crops (BFR) and pasturage (BFP)	Element specific data, general for soils and vegetation types	A-1 – A-4
Translocation to edible parts	Element specific data/estimations	A-5
Vertical transport of soil due to earthworms	General data	3-1
Length of growing period (GRPER)	Region specific data from climatic conditions	3-1
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

3.3 Groundwater (well)

Well (groundwater) is the simplest of all modules. It consists simply of one compartment for water. Water from the well is used as drinking water for humans and cattle in the module. The sub-module irrigation is included in this module. In the results below a garden plot, where the annual consumption of vegetables is grown, is assumed to be irrigated with 0.15 $m^3/(m^2 \cdot year)$ ranging between 0.1 and 0.2 $m^3/(m^2 \cdot year)$. For description of how irrigation is handled, see section 3.2 above.

3.3.1 Exposure pathways

Human exposure pathways considered are

- Consumption of water
- Consumption of milk and meat from cattle drinking well water

Additional exposure pathways via the irrigation module are

- Consumption of green vegetables
- Consumption of root-crops
- Consumption of soil
- Inhalation of resuspended particles from garden plots
- External exposure from garden plots

3.3.2 Data

The annual volumes for mixing are assumed to be the same as annual capacities. These annual volumes are obtained by assuming that reported capacities (Swedish archive) per hour should be valid for the whole year.

The only specific parameter for the well module is the dilution volume which is set to 2 000 m^3/year in the calculations shown in Chapter 4. The value is somewhat higher than 0.2 m^3/h.

3.3.3 A summary of processes and parameters

Processes and parameters are summarised in Table 3-4. The abbreviations in capital letters are used in Chapter 5, variation analyses. Comments are also given on how parameter values were obtained for the calculations.

Table 3-4 Processes and parameters used in the calculations for the well module.

Processes and parameters	Comments	Table
Human activities		
Irrigation of garden plot, see sub-module irrigation	General data	3-2
Consumption rates of water (CONWA), milk (CONMI), meat (CONME), vegetables (CONVEG) and root crops(CONROOT)	General data	B-2
Physical processes and properties		
Well capacities	Swedish Well Archive	
Biological processes and parameters		
Cattle's consumption (COCON) of water	General data	B-1
Cattle's metabolism; transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

3.4 Lake

The lake module consists of compartments for water and sediments. The source term for the module is assumed to be via inflow of contaminated water directly to the water compartment, see Figure 3-6. Most nuclides released to this surface water are partly attached to particulate matter causing a transport to the bottom sediments. The turnover of water causes a transport out of the lake for a fraction of the nuclides. Processes involving sediments were modelled by three compartments; one for erosion and transportation bottoms, one for accumulation sediment and one for deep sediments.

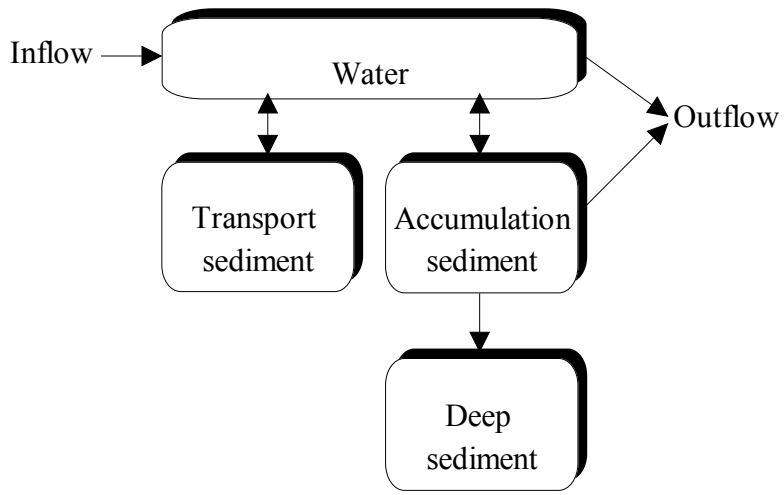

Figure 3-6 Compartment system for the lake module.

3.4.1 Water turnover

For calculations of radionuclide concentrations in water the inflow/outflow or residence time of water and lake volumes are needed. Data for the lake, such as volume, surface area and catchment area are therefore needed, see the equations below. Depending on availability of data several options are possible to use in the module. The methods available in the model are given below according to the order of priority in the module. The rate constants (TC_{out}) symbolising the outflow of water are therefor obtained from the following equations

$$TC_{out} = 1/RT$$

where

RT = Mean residence time for the water in the lake [years]

or

$$TC_{out} = Q_w / V$$

where

Q_w = Outflow from the lake [m^3/year]
V = Volume of the lake [m^3]

or

$$TC_{out} = A_c \cdot R/V$$

where

A_c = Catchment area for the lake [m^2]
R = Runoff from the catchment area of the lake [$m^3/(m^2 \cdot year)$]

3.4.2 Sediments

Sediments near lake shorelines are erosion bottoms with no or very little net sedimentation, while in deeper parts there are accumulation bottoms where net sedimentation occurs (Håkanson & Jansson, 1983). Therefore, the surface sediments were divided into two compartments. The percentage of bottoms that are erosion and transportation bottoms (P_{et}) were calculated according to an empirical expression (Håkanson & Jansson, 1983)

$$P_{et} = 25 \cdot \frac{\sqrt{a \cdot 10^{-6}}}{D} \cdot e^{\left(0.23 \cdot \frac{D}{\sqrt{a \cdot 10^{-6}}}\right)}$$

where

a = Area of the lake [m^2]
D = Mean depth of the lake [m]

The transport of organic and inorganic particulate material, from the water body (w) to sediment (sed), was in the model assumed to take place via gravitational settling. The transfer of elements in water (w) to sediment (sed) was obtained in the model by using the expression according to Hill and Lawson (1980). The transfer coefficient or rate constant becomes

$$TC_{w,sed} = \frac{K_d \cdot SR}{D \cdot (1 + K_d \cdot Susp)}$$

where

SR = Gross sediment rate [kg/(m^2·year)]
Susp = Concentration of suspended matter in water [kg/m^3]
D = Mean depth of the lake [m]
K_d = Distribution coefficient, concentration on solids form relative to dissolved [m^3/kg]

The transfer rates were assumed to be distributed between the transportation and accumulation bottoms according to size.

Resuspension from erosion bottom sediments to the water body was set to 90 % (TC = 0.9/year) while the transfer coefficient describing resuspension from accumulation sediments was given by the following expression which was empirically found to give good agreement with observed concentration of Cs-137 (Nordlinder et al, 1997).

$$TC_{sed,w} = e^{\left(-k \cdot \frac{D_m}{\sqrt{a \cdot 10^{-6}}}\right)}$$

where

k = Empirically obtained factor, varied uniformly between 1 and 2
D_m = Maximum depth of the lake [m]
a = Area of the lake [m^2]

If the maximum depth is not given in the input, this depth is set to three times the mean depth (Håkanson & Jansson, 1983).

Sediments on accumulation bottoms act as a sink for radionuclides if no evolution of the lake is considered. The volume of top sediment is assumed constant while an addition of new top-sediments in a compartment model can be described by a corresponding net transport of particles and associated nuclides from top sediments to deeper sediments. The volume of the top sediment is assumed to be constant. The creation of new top

sediment thus transfers the bottom of the accumulation sediment to deep sediment. This sink may, however, be seen in a long-term perspective as a secondary source to exposure if sediments are transferred or transformed to soil.

According to a review by Evans (1986) the fraction of primary production reaching the sediment is 10 to 50 %. The organic fraction was here assumed to be completely decomposed and was therefore not considered to contribute to the sediment net growth in this model.

Considering these factors and the mass balance for the accumulating sediment, the transfer rate describing the transport to deeper (d) from upper sediments becomes

$$TC_{sed,d} = \frac{(SR - PP \cdot f_{sed}) \cdot \left(\frac{P_a}{100}\right)}{M_{sed}} - TC_{sed,w}$$

where

SR = Gross sedimentation rate [kg/(m^2·year)]

PP = Primary production [kg/(m^2·year)]

f_{sed} = Fraction of primary production reaching the sediment [-],

P_a = Percent accumulation bottoms ($100 - P_{et}$)

M_{sed} = Mass of top sediment per area in accumulation bottom [kg/m^2]

3.4.3 Exposure pathways

Human exposure pathways considered are:

- Consumption of water
- Consumption of fish
- Consumption of crustacean
- Consumption of milk
- Consumption of meat

It is assumed that cattle graze along shores of the lakes where they consume aquatic plants as well as water. Irrigation with lake water is also assumed, see Chapter 4. This leads to an additional contamination of milk and meat by the cows' consumption of terrestrial plants and soil for the remaining part of the year, when they are not grazing along the shores. The additional pathways due to irrigation are then:

- Consumption of cereals
- Consumption of root crops
- Consumption of vegetables
- Consumption of soil
- Inhalation of resuspended particles from the soil
- External exposure from contaminated ground

3.4.4 Data

Site specific data are shown in Table 3-5. Table 3-6 below shows suggestions of some general data in absence of site specific information. Data used for the irrigation part are shown in Tables in Section 3.2 — Irrigation module. Data for oligotrophic lakes are used in the calculations.

The sedimentation rates given below cover a spectrum of lakes. In the literature primary production is given as gram carbon per m^2 and year. This was transferred to dry mass by assuming a 50 % carbon content in dry primary production

Table 3-5 Site specific data used in the lake module calculations.

	Abbr	Unit	Mean	Min* Triangularly dist	Max*	Reference
Area	a_l	km^2	4.3	3.9	4.7	Wallsten & Blomqvist, 1982
Max depth	D_m	m	4.1	3.7	4.5	Carlsson & Gidlund, 1983
Mean depth	D	m	2.0	1.2	2.3	Sundblad & Bergström, 1983
Catchment area	A_c	km^2	117	105	129	Wallsten & Blomqvist, 1982
Runoff	R	$m^3/(m^2 \cdot year)$	0.24	0.20	0.28	Lindborg & Schüldt, 1998

* The min and max values were set to 90 and 110 % of the mean value.

Table 3-6 General data used in the lake module calculations.

	Abbr	Unit	Mean	Min	Max	Reference
				Triangularly dist		
Gross sedimentation rate	SR	kg d.w./(m^2·year)				
Eutrophic			2.0	0.5	4.0	Evans, 1986
Oligotrophic			1.0	0.2	2.0	Evans, 1986
Suspended matter	Susp	kg d.w./m^3				
Eutrophic			6E-3	3E-3	1E-2	Evans, 1986
Oligotrophic			1E-3	5E-4	1E-2	Evans, 1986
Primary production	PP	kg d.w./(m^2·year)				
Eutrophic			0.24	0.08	0.40	Håkanson & Peters, 1995
Oligotrophic			0.010	0.005	0.060	Evans, 1986, Håkanson & Peters, 1995
Fraction of primary production reaching sediment	f_{sed}		0.35	0.10	0.50	Evans, 1986
Number of days for grazing at shore	ND	day	90	75	105	Assumed values
Yield values	Y_i	kg/m^2	0.5	0.4	0.6	Assumed values
Plant transpiration	TR	g/(m^2·h)	100	50	300	Jörgensen et al, 1991, Table 1-392
Resuspended fraction	Resusp		0.9	0.8	0.95	Assumed values
Mass of top sediment	M_{sed}	kg/m^2	10	5	15	Assumed values

3.4.5 A summary of processes and parameters

Processes and parameters are summarised in Table 3-7 below. The abbreviations in capital letters are used in Chapter 5, variation analyses. Comments are also given on how parameters were obtained for the calculations.

Table 3-7 Processes and parameters used in the calculations for the lake module.

Processes and parameters	Comments	Table
Human activities		
Days cows grazing at shore (ND)	General data	B-1
Consumption rates of water (CONWA), fish (CONF), crustacean (CONCRU), milk (CONMI), meat (CONME), cereals (CONCE), vegetables (CONVEG) and root crops (CONROOT)	General data	B-2
Exposure times (time on contaminated soil)	General data	B-2
Irrigation of agricultural areas, see sub-module irrigation section 3.2		
Physical processes and properties		
Lake volume, drainage areas, precipitation, evaporation, lakes' and sediments' depths, water turnover rates	Site specific data	3-5
Suspended matter, sedimentation rates in lakes	General data	3-6
Resuspension rate from sediment	Calculated from site specific data	
Fraction of resuspension and accumulation bottoms	Calculated from site specific data	
Mass of top sediment	General data	3-6
Primary production	General data	3-6
Fraction of primary production transferred to sediment	General data	3-6
Chemical processes and properties		
Adsorption of nuclides on suspended matter in water	Element specific data	A-8
Biological processes and parameters		
Uptake to fish (BFF)	Element specific data general for fish	A-12
Uptake to crustacean (BFCR)	Element specific data general for crustacean	1-17
Transpiration (BFAQ)	General data	3-6
Cattle's consumption (COCON) of water and water plants	General data	B-1
Yield values of water plants	General data	3-6
Cattle's metabolism; transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Decomposition of organic material to sediment	100 % annual decomposition in sediments (general data)	
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

3.5 Running waters

Similar to the well module, the module running waters is simple, only encompassing one compartment for water. Exchange of elements between sediments and running waters was neglected, see Chapter 5.

3.5.1 Water flow rates

The concentration of a radionuclide (C_r) in water when only mixing is considered and for a unit leakage of 1 Bq/year of respective radionuclide becomes

$$C_r = 1/Q_w \qquad [Bq/m^3]$$

where

Q_w = The flow of water [m^3/year].

If the flow is not known, it is calculated from data on the size of the catchment area and runoff

$$Q_w = A_c \cdot R$$

where

A_c = Catchment area [m^2]
R = Runoff [$m^3/(m^2 \cdot year)$]

3.5.2 Exposure pathways

The exposure pathways considered in the running water module are:

- Consumption of water
- Consumption of fish
- Consumption of crustacean
- Consumption of milk
- Consumption of meat

In similarity to the lake module it is assumed that cattle graze along shores where they consume aquatic plants as well as water. Their annual consumption of water is taken from the running water. Irrigation using water from the stream is also assumed in the calculations, see Chapter 4. This leads to an additional contamination of milk and meat by the cows' consumption of soil and terrestrial plants for the time not spent on the shore. The additional pathways due to irrigation are then:

- Consumption of cereals
- Consumption of root crops
- Consumption of green vegetables
- Consumption of soil
- Inhalation of resuspended particles from soil
- External exposure from contaminated ground

3.5.3 Data

Table 3-8 below shows site specific data used in the calculations, see Chapter 4. The catchment area is set to approximately the same size as the study sites in Nordlinder et al, (1999). General data concerning uptake of elements in water plants are shown in Table 3-9.

Table 3-8 Site specific data used in the running waters module calculations .

	Abbr	Unit	Mean	Min	Max	Reference
				Triangularly dist		
Catchment area	A_c	km^2	20	18	22	Assumed value (see text above)
Runoff	R	$m^3/(m^2 \cdot year)$	0.24	0.20	0.28	Lindborg & Schüldt, 1998

Table 3-9 General data used in the running waters module calculations.

	Abbr	Unit	Mean	Min	Max	Reference
				Triangularly dist		
Number of days for grazing at shore	ND	Day	90	75	100	Assumed values
Yield values	Y_i	kg/m^2	0.5	0.4	0.6	Haak, 1983
Plant transpiration	TR	$g/(m^2 \cdot h)$	100	50	300	Jörgensen et al, 1991, Table 1-392

3.5.4 A summary of processes and parameters

Processes and parameters are summarised in Table 3-10 below. The abbreviations in capital letters are used in Chapter 5, variation analyses. Comments are also given on how parameter values were obtained for the calculations.

Table 3-10 Processes and parameters used in the calculations for the running waters module

Processes and parameters	Comments	Table
Human activities		
Keeping cattle at shore of the running waters (ND)	General data	B-1
Consumption rates of water (CONWA), fish (CONF), crustacean (CONCRU), milk (CONMI), meat (CONME), cereals (CONCE), vegetables (CONVEG) and root crops(CONROOT)	General data from Swedish foodstuff statistics	B-2
Irrigation of agricultural areas	See sub-module irrigation	
Physical processes and properties		
Flow rate or drainage areas and run-off	Site specific data	3-8
Biological processes and parameters		
Uptake to fish (BFF)	Element specific data, general for fish	A-12
Uptake to crustacean (BFCR)	Element specific data, general for crustacean	A-14
Transpiration (BFAQ)	General data	3-9
Cattle's consumption (COCON) of water and water plants	General data	B-1
Yield values of water plants (YWA)	General data	3-9
Cattle's metabolism; transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

3.6 Coast

The coast module represents the Swedish Baltic Sea coast. This means that the module is possible to use for archipelagos as well as for open coasts. The main reservoirs for this module are surface water and sediment. Fauna is also a part of the module, but only as an exposure pathway to humans. The module consists of two parts, a bay and open coast. An exchange of water occurs between these two parts. Depending on the local situation the source of radionuclides to the module can be either via an inflow to a bay in the archipelago or to the open coast, see Figure 3-7.

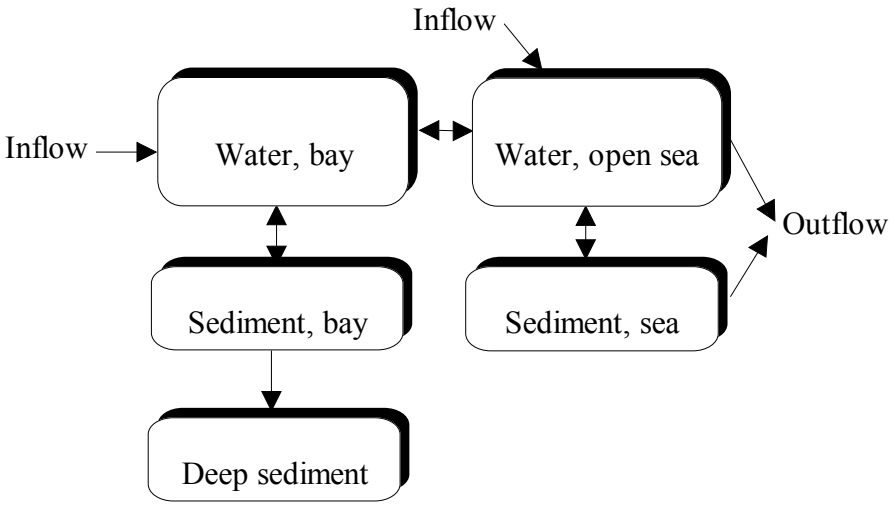

Figure 3-7 Compartment system for the coast module. The inflow of radionuclides may occur to water in the bay of an archipelago as well as to water, open sea.

3.6.1 Water turnover

The turnover of water within a bay and the section of the open coast of interest can be expressed from the mean residence time of water for each compartment. An alternative is to give the flow rate in and out from the bay and the flow rate out from the open coast compartment. The transfer coefficient describing the water turnover in the bay becomes

$$TC_{b,O} = \frac{365}{RETTIME}$$

where

365 =Days/year
RETTIME =Mean residence time of water in the bay (day)

The rate constant describing the inflow from open sea to the bay becomes

$$TC_{O,b} = \frac{365}{RETTIME} * \frac{V_b}{V_o}$$

where

V_b =Volume of the bay
Vo =Volume of open seabox

Simple generic methods to estimate the turnover of water in a bay and open coast, which could easily be implemented in this module, are not generally available.

3.6.2 Sediments

The transport of organic and inorganic particulate material from the water body to sediment was in the model assumed to take place via gravitational settling. The transfer of elements from the water (w) compartment to sediment (sed) was obtained by using the following expression according to Hill & Lawson (1980)

$$TC_{w,sed} = \frac{K_d \cdot SR}{D \cdot (1 + K_d \cdot Susp)}$$

where

SR = Gross sediment rate [kg/(m^2·year)]

Susp = Concentration of suspended matter in water [kg/m^3]

D = Mean depth of the bay [m]

K_d = Distribution coefficient, concentration of the element on particles in suspended matter relative to dissolved form [m^3/kg]

Resuspension is an important process for the turnover of particulate matters in coastal areas (Blomqvist & Larsson, 1994). The transfer coefficient describing resuspension from sediments in the bay was given by the following expression which was obtained from studies of Cs-137 fallout from the Chernobyl accident, see section 3.4 (Nordlinder et al, 1997).

$$TC_{sed,w} = e^{\left(-k \cdot \frac{D_m}{\sqrt{a \cdot 10^{-6}}}\right)}$$

where

k = Empirical factor, varied uniformly between 1 and 2

D_m = Maximum depth of the bay [m]

a = Area of the bay [m^2]

Sediments on accumulation bottoms may act as a sink for radionuclides. This sink may, however, be seen in a long-term perspective as a secondary source to exposure if sediments may be transferred to soil. The volume of the accumulating sediment is assumed to be constant. The new sedimentation thus transfers the bottom of the accumulating sediments (sed) to deeper sediments (d). The transfer coefficient describing the transport to deeper sediments then becomes

$$TC_{sed,d} = \frac{SR}{M_{sed}} - TC_{sed,w}$$

where

SR = Gross sedimentation rate [kg/(m^2·year)]

M_{sed} = Mass of top sediment per area [kg/m^2]

Few quantifications are available for the resuspension rate from sediment to water for open coast. In the calculations a transfer rate was therefore used which has been used in studies of routine discharges from nuclear power plants (Nordlinder & Bergström, 1992).

3.6.3 Exposure pathways

Human exposure pathways considered in this module are:

- Consumption of fish
- Consumption of milk and meat from cattle drinking the water and eating aquatic plants
- Consumption of water plants (algae)

3.6.4 Data

Tables 3-11 and 3-12 below give the values used in the calculations shown in Chapter 4. They consist of site-specific physical parameters as well as suggestions for general data in absence of precise information.

Table 3-11 Site specific data used in the coast module calculations.

	Abbr	Unit	Mean	Min	Max Triangularly dist	Reference
Area of surface bay	a_b	m^2	1.4E6	1.3E6	1.5E6	Sundblad & Mathiasson, 1994
Mean depth, bay	D	m	2.3	2.1	2.5	Sundblad & Mathiasson, 1994
Max depth, bay	D_m	m	8.0	7.5	8.5	Sundblad & Mathiasson, 1994
Mean residence time of water in bay	RET-TIME	days	45	42	48	Engqvist 1997
Volume open seacoast box	V_o	m^3	1.7E8	1.4E8	2E8	Nordlinder & Bergström, 1992
Mean depth open sea		m	7	6	8	Nordlinder & Bergström, 1992
Water turnover, open sea		$year^{-1}$	44	30	57	Nordlinder & Bergström, 1992

Table 3-12 General data used in the coast module calculations.

	Abbr	Unit	Mean	Min	Max	References
				Triangularly dist		
Gross sedimentation rate	SR	kg d.w./(m^2·year)				
Bay			2.0	0.5	5.0	Meili et al, 1998
Open coast			0.20	0.05	0.40	Meili et al, 1998
Suspended matter	SUSP	kg d.w./m^3				
Bay			0.0010	0.0005	0.0020	*
Open coast			0.0010	0.0005	0.0020	Assumed values
Mass of top sediment per area	M_s	kg/m^2	10	5	15	Assumed values
Resuspension rate from sea sediment		$year^{-1}$	0.2	0.1	0.3	Nordlinder & Bergström, 1992
Number of days for grazing at shore	ND	days	90	75	105	Assumed values
Yield values	Y_i	kg/m^2	0.5	0.3	0.4	Assumed values
Plant transpiration	TR	g/(m^2·h)	100	50	300	Jörgensen et al, 1991, Table 1-392

* Due to lack of data the same values were applied as in an earlier modelling of long-lived radionuclides (Bergström & Nordlinder, 1991c).

3.6.5 A summary of processes and parameters

Processes and parameters are summarised in Table 3-13 below. The abbreviations in capital letters are used in Chapter 5 treating uncertainty analyses. Comments are also given on how parameter values were obtained for the calculations.

Table 3-13 Processes and parameters used in the calculations for the coast module.

Processes and parameters	Comments	Table
Human activities		
Days cows are grazing at shore	General data	B-1
Consumption rates of, fish (CONF), algae (CONALG), milk (CONMI) and meat (CONME)	General data from Swedish foodstuff statistics	B-2
Exposure times	General data	B-2
Physical processes and properties		
Volumes, water depths, water turnover rates	Site specific data	3-11
Suspended matter, sedimentation rates	General data/Estimates	3-12
Resuspension rate	General data/Estimates	3-12
Mass of upper sediment	General data	3-12
Chemical processes and properties		
Adsorption on suspended matter	Element specific data	A-9
Biological processes and parameters		
Uptake to fish (BFF)	Element specific data, general for fish	A-13
Uptake to algae (BFCR)	Element specific data, general for algae	A-14
Transpiration (BFAQ)	General data	3-12
Cattle's consumption (COCON) of water and water plants	General data	B-1
Yield values of water plants	General data	3-12
Cattle's metabolism; transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

3.7 Agricultural land

The agricultural land module describes a saturated zone with a near horizontal flow of groundwater where the water table is about one meter below the surface. The radionuclides are assumed to enter the system via an inflow to this groundwater (Figure 3-8). The ecosystem represents agricultural fields, which are ditched or have draining-tiles. Due to this the groundwater level can be assumed to be relatively constant. Nuclides are being transported from the saturated zone up to the soil layers via processes as capillary rise, root-uptake and diffusion. The saturated zone is divided into two compartments one representing the water content and one solid material. The unsaturated zone consists of a surface layer and a deep layer. The nuclides are assumed to enter as soluble but attach directly to soil particles.

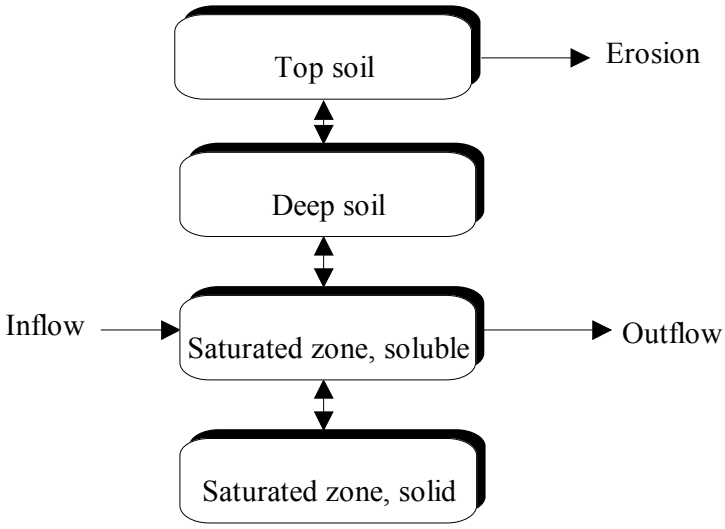

Figure 3-8 Compartment system for the agricultural land module. Nuclides enter in soluble form, interacts with the soil. A fraction of the radionuclides migrates upwards while the remaining fraction in soluble form leaves the system.

3.7.1 Transfer between solid and soluble fractions in saturated zone

The exchange of elements between dissolved and solid fractions in the saturated zone is exposed in one parameter although it involves chemical, biological and physical processes. If the reaction velocity is known, the transfer coefficients from dissolved (s) to solid matter (p) is given by the expression

$$TC_{s,p} = \frac{K_d \cdot \ln(2)}{Tk} \cdot \frac{(1-\varepsilon_{sa}) D_{sa} \cdot A \cdot \rho_{sa}}{\varepsilon_{sa} \cdot D_{sa} \cdot A} = \frac{K_d \cdot \ln(2)}{Tk} \cdot \frac{(1-\varepsilon_{sa}) \cdot \rho_{sa}}{\varepsilon_{sa}}$$

and from particulate to dissolved fraction by

$$TC_{p,s} = \frac{\ln(2)}{Tk}$$

where

K_d = Distribution coefficient, concentration of the element on solids relative to dissolved [m^3/kg].

Tk = Reaction half-time [year]

ε_{sa} = Porosity in saturated zone [m^3/m^3]

D_{sa} = Depth of saturated zone [m]

ρ_{sa} = Density of soil particles [kg/m^3]

A = Area of agricultural land [m^2]

The value of Tk was assumed to be considerably lower than the time steps used, these values are therefore insignificant for the model results.

3.7.2 Transfer from saturated zone

The transfer coefficient describing the horizontal flow of dissolved radionuclides in groundwater out from the saturated zone(sa), see Figure 3—8, is based on water balance and becomes

$$TC_{sa,out} = \frac{R \cdot A}{\varepsilon_{sa} \cdot D_{sa} \cdot A}$$

where

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

3.7.3 Transfer between saturated zone and deep soil

The direction of water flow between saturated and unsaturated zones varies with time (BIOMOVS II, 1996c). 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, $F_{sa,\,ds}$ was estimated to 200 mm per year, using results from lysimeter experiments in which the saturated level was held constant (BIOMOVS II, 1996c). The transfer coefficient from saturated to deep unsaturated soil (ds) considering the fraction of nuclides in soluble form becomes

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

where

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

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

$$TC_{ds,sa} = \frac{(R + F_{ds,sa}) \cdot A}{\varepsilon_{ds} \cdot D_{ds} \cdot A} \cdot \mathrm{Re\,t}$$

where

$F_{ds,sa}$ = Flow [$m^3/(m^2 \cdot year)$]
R = Runoff [$m^3/(m^2 \cdot year)$]
A = Area of agricultural land [m^2]

ε_{ds} = Porosity of deep soil [m^3/m^3]
D_{ds} = Depth of deep soil [m]
Ret = Retention, (see below) [-]

$$Ret = \frac{1}{1 + K_d \cdot \rho_p \cdot \frac{1-\varepsilon}{\varepsilon}}$$

where

K_d = Distribution coefficient for element i in soil [m^3/kg]
ρ_p = Density of particles [kg/m^3]
ε_i = Porosity of the compartment i [m^3/m^3]

3.7.4 Transfer between deep soil and top soil

The upward flow from deep soil to top soil was estimated from the same lysimeter experiment as mentioned above (BIOMOVS II, 1996c). An important process for transport between deep soil and top soil is bioturbation, mainly caused by earthworms (Müller-Lemans & van Dorp, 1996). The transfer coefficient from deep soil (ds) to top soil (ts), considering these factors, becomes

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

where

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

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

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

where

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

ρ_p = Density of soil porosity [kg/m^3]
A = Area of agricultural land [m^2]

3.7.5 Erosion

The loss of radionuclides (out) due to erosion (e) is estimated as

$$TC_{es,out} = \frac{Er \cdot A}{D_{ts} \cdot A}$$

where

Er = Erosion [m/year]
A = Area of agricultural land [m^2]
D_{ts} = Depth of top soil [m]

3.7.6 Exposure pathways

The soil is assumed to be used for agricultural purposes giving the following exposure pathways:

- Consumption of milk
- Consumption of meat
- Consumption of cereals
- Consumption of root crops
- Consumption of vegetables
- Consumption of soil
- Inhalation of resuspended particles
- External exposure from contaminated soil

3.7.7 Data

Table 3-14 below gives suggested values for some physical-chemical parameters when site-specific data are not available. These values were used in the calculations shown in Chapter 4.

The area of agricultural land is selected so it is large enough to support an individual with food, but still small enough to be totally covered by a square in the gridnet when identifying typical ecosystems at the study sites (Nordlinder et al, 1999).

Table 3-14 General data used in the agricultural land module calculations, triangularly distributed.

	Abbr	Unit	Mean	Min	Max	References
Runoff	R	$m^3/(m^2 \cdot year)$	0.24	0.20	0.28	Lindborg & Schüldt, 1998
Depth of top soil	D_{ts}	m	0.30	0.25	0.35	Assumed values
Depth of deep soil	D_{ds}	m	1.0	0.9	1.1	Assumed values
Depth of saturated zone	D_{sa}	m	5	2	8	Assumed values
Top soil porosity	ε_{ts}	m^3/m^3	0.4	0.3	0.5	Hillel, 1980
Deep soil porosity	ε_{ds}	m^3/m^3	0.3	0.2	0.4	Hillel, 1980
Saturated zone porosity	ε_{sa}	m^3/m^3	0.30	0.20	0.35	Assumed values
Bioturbation	BioT	$kg/(m^2 \cdot year)$	2	1	3	Müller-Leman & van Dorp, 1996
Water transport groundwater to deep soil	$F_{s,ds}$	$m^3/(m^2 \cdot year)$	0.2	0.1	0.3	BIOMOVS II, 1996c
Water transport deep soil to top soil	$F_{ds,ts}$	$m^3/(m^2 \cdot year)$	0.10	0.05	0.20	Assumed values
Area of agricultural land	A	m^2	1E4	2E3	5E4	Assumed values
Soil density	ρ_p	kg/m^3	2400	2000	2800	Hillel, 1980
Erosion	Er	m/year	0.00010	0.00003	0.00030	Assumed values

3.7.8 A summary of processes and parameters

Processes and parameters are summarised in Table 3-15 below. The abbreviations in capital letters are used later in Chapter 5 treating uncertainty analyses. Comments are also given on how parameter values were obtained for the calculations.

Table 3-15 Processes and parameters used in the calculations for the agricultural land module.

Processes and parameters	Comments	Table
Human activities		
Consumption rates of milk (CONMI), meat (CONME), cereals (CONCE), vegetables (CONVEG), root crops (CONROOT)	General data	B-2
Physical processes and properties		
Soil porosity, density, depths, area (FAR)	General data	3-14
Runoff	General data	3-14
Upward transport of water (UTW)	General data	3-14
Aquifer depth (AQDE)	General data	3-14
Soil erosion (EROS)	General data	3-14
Chemical processes and properties		
Sorption of nuclides on soil particles	Element specific distribution factors	A-6
Biological processes and parameters		
Cattle's consumption of pasturage, cereals and soil (COCON)	General data	B-1
Cattle's metabolism, transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Root-uptake factors for cereals (CBFC), leafy vegetables (BFV), root crops (BFR) and pasturage (BFP)	Element specific data	A-1 – A-4
Translocation to edible parts	Element specific data/estimations	A-5
Vertical transport in soil by earthworms	General data	3-14
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

3.8 Peat bog

Radionuclides enter the system via an outflow of ground water to the peat bog area. Peat accumulates many elements depending on chemical conditions (Statens energiverk, 1985). When burning peat, releases of radionuclides to air may occur, or an enrichment of them in ash (Nordlinder, 1989). The compartment system for the peat bog module is shown in Figure 3-9.

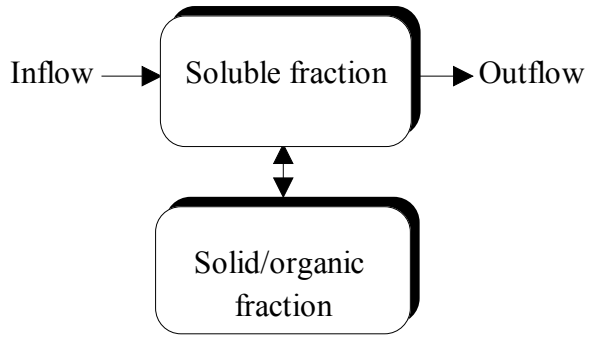

Figure 3-9 Compartment system for the peat bog module.

3.8.1 Transfer between solid and soluble fraction

The exchange of elements between dissolved and solid phases in peat is exposed in one parameter although it involves chemical, biological and physical processes. If the reaction velocity is known, the transfer coefficients from dissolved (s) to solid fraction (p) can be given by the expression

$$TC_{s,p} = \frac{K_d \cdot \ln(2)}{Tk} \cdot \frac{D_p \cdot A \cdot \rho_p}{\varepsilon_p \cdot D_p \cdot A} = \frac{K_d \cdot \ln(2)}{Tk} \cdot \frac{\rho_p}{\varepsilon_p}$$

and from particulate to dissolved fraction by

$$TC_{p,s} = \frac{\ln(2)}{Tk}$$

where

K_d = Distribution coefficient, concentration of the element on solids relative to dissolved [m^3/kg].

Tk = Reaction half-time [year]

ε_p = Porosity in peat [m^3/m^3]

D_p = Depth of peat bog [m]

ρ_p = Density of peat [kg/m^3]

A = Peat bog area [m^2]

The value of Tk was assumed to be considerably lower than the time steps used, these values are therefore insignificant for the model results.

3.8.2 Outflow of soluble fraction

The transfer coefficient for the horizontal flow (h) is based on water balance and becomes

$$TC_{h,out} = \frac{R \cdot A}{\varepsilon_p \cdot D_p \cdot A}$$

where

R = Runoff [$m^3/(m^2 \cdot year)$]
A = Area of the peat bog [m^2]
ε_p = Porosity of peat bog [m^3/m^3]
D_p = Depth of peat bog [m]

3.8.3 Exposure pathways

Crops for cattle and humans consumption are assumed to be grown on peat. Inhalation of off-gases from combustion without air filters was also assumed in the calculations (see Chapter 4).

Human exposure pathways considered are:

- Consumption of milk
- Consumption of meat
- Consumption of cereals
- Consumption of root-crops
- Consumption of vegetables
- Inhalation of resuspended particles from the peat bog
- Inhalation of nuclides in the off-gases from fuel peat
- External exposure from the peat bog

3.8.4 Data

Table 3-16 below gives suggested values for some physical-chemical parameters when site-specific data are not available. These values are used in the calculations shown in Chapter 4. The data in Table 3-17 are used for calculation of the exposure through inhalation of radionuclides in offgases from combustion of peat, see Appendix B.2.

Table 3-16 General data used in the peat bog module calculations.

	Abbr	Unit	Distr*	Mean	Min	Max	References
Runoff	R	$m^3/(m^2 \cdot year)$	T	0.24	0.20	0.28	Lindborg & Schüldt, 1998
Density	ρ_p	kg/m^3	T	100	70	200	Steinmann & Shotyk, 1997
Porosity	ε_p	-	T	0.90	0.80	0.95	Sharma & Forster, 1993
Peat area	A	m^2	U	10000	2000	50000	Assumed values
Peat depth	D_p	m	T	0.5	0.3	2.0	Assumed values

* LT = Logtriangular distribution, T = Triangular distribution
U = Uniform distribution.

Table 3-17 Parameters used in the dose assessment for combustion of peat.

Parameter	Unit	Distr*	Mean	Min	Max	Reference
Fuel load	kg d.w./h	T	1.0	0.5	2.0	Assumed values**
Filter efficiency	-	C	1			Not considered
Relative concentration	s/m^3	LT	1E-5	3E-6	3E-5	Widemo & Gyllander, 1979
Exposure time	h/year	T	8000	6500	8760	Assumed values

* T = Triangular distribution, LN = Logtriangular distribution,
C = Constant

** Energy content in dry peat is 20 MJ/kg (Statens energiverk, 1985)

3.8.5 A summary of processes and parameters

Processes and parameters are summarised in Table 3-18 below. The abbreviations in capital letters are used later in Chapter 5, variation analyses. Comments are also given on how parameter values were obtained for the calculations.

Table 3-18 Processes and parameters used in the calculations for the peat bog module.

Processes and parameters	Comments	Table
Human activities		
Consumption rates of milk (CONMI), meat (CONME), cereals (CONCE), vegetables (CONVEG) and root crops (CONVEG)	General data	B-2
Fuel load	General data	3-17
Inhalation	General data	B-2
Physical processes and properties		
Peat porosity, density	General data	3-16
Filtration of gases from combustion	General data	3-17
Air dispersion	General data	3-17
Chemical processes and properties		
Sorption on particles	Element specific distribution factors	A-7
Biological processes and parameters		
Cattle's consumption of pasturage, cereals and soil (COCON)	General data	B-1
Cattle's metabolism, transfer to milk (DFMI) and meat (DFME)	Element specific distribution factors	A-10, A-11
Root-uptake factors for cereals (CBFC), leafy vegetables (BFV), root crops (BFR) and pasturage (BFP)	Element specific data	A-1 – A-4
Others		
Half-lives	Nuclide specific constants	2-1
Dose coefficients	Nuclide specific constants	2-1

4 Results

Each type of module was run for an annual release of 1 Bq per year of each radionuclide during 10 000 years i.e. for a total release of 10 000 Bq. Results for each module are shown in the sections below. They are presented as annual EDFs for each one of the nuclides at year 10 000. Note that they represent doses based on an integration time within the human body of 50 years for the nuclides (ICRP 42, 1984).

A comparison between results from the six modules shows that releases of radionuclides to peat bog give rise to the potential highest EDFs, followed by EDFs from a release to a well and agricultural land. One explanation to the high EDFs for the peat module is the assumption that radionuclides are strongly retained in peat. Furthermore, all radionuclides released from the geosphere were assumed to pass through the peat, in contrast to the soil module for which a fraction of the release was assumed to be unavailable for transport through the upper soil layers. The results for the well module are directly inversely proportional to the well capacity. This is low compared to the water volume in the other modules thus explaining the comparatively high EDFs for this module. The well module does, on the other hand, not include any final destination for the radionuclides (with the exception of the part remained in soil) in contrast to the other modules. Releases of radionuclides to agricultural land result in somewhat lower EDFs than those for the peat bog and well modules.

Releases to a coastal area give the lowest EDFs compared to the other alternatives. This is because the coast was assumed to have the largest volumes of water for mixing and that this water was not used for irrigation. The results for the surface water modules, lake and running waters, are higher than those for the coast module but lower than those for the peat bog, agricultural land and well modules. It should be noted that the EDFs for the surface water modules are inversely proportional to the mixing volumes, implying that releases to a larger river result in decreased EDF-values.

The time required to obtain the highest exposure during the 10 000 years of release is determined by the build-up of a radionuclide in compartments as peat or soil and by the radionuclide's half-life. Steady-state conditions are obtained within the 10 000 years for poorly sorbing elements and nuclides with short half-lives, but this is not the case for all nuclides. For all modules with soil, accumulation in soil causes a continued increase of the EDF-rates during the whole period for most nuclides. Exceptions to this are H-3, C-14, Cl-36, Co-60, Sr-90, Cs-137, Sm-151, Pb-210, Ac-227 and Cm-244 for which maximum EDF-values are obtained within 1 000 years. The calculated time for short-lived radionuclides to reach 99 % of the EDF-value at year 10 000 is longer in agricultural land than in peat. For long-lived radionuclides with high sorption, steady-state may not be reached within 10 000 years for the agricultural land or peat modules. For the coast module, which does not include any soil or peat compartment, the time to reach 99 % is a few years for all nuclides. This is due to the short residence time of water and that resuspension from sediments not contributes to the concentrations of radionuclides in water as long as the releases are maintained.

4.1 Well

The most exposed individual in the well module is assumed to take the water from the well. His annual consumption of milk and meat is from cattle drinking water from the well. All vegetables he consumes annually are grown on his own garden plot which is irrigated with water from the well. When consuming the vegetables he inadvertently ingests some soil. He obtains external exposure from the garden plot when working in it, and he obtains internal exposure when inhaling resuspended particles.

Mean annual EDFs at year 10 000 with standard deviations and ranges corresponding to 99 % confidence interval of the calculated distributions are shown for the well module in Table 4-1. Table 4-2 shows the time it takes to reach 50, 90 and 99 % of these EDFs. The percentage contributions to the total EDF from various exposure pathways are shown in Table 4-3.

Consumption of water is an important exposure pathway for most nuclides. For actinides and Zr-93, inhalation of dust gives a considerable contribution to the EDF. This is because the α-emitting actinides in general have two orders of magnitudes higher dose conversion factors for inhalation than for ingestion. Other exposure pathways give minor contribution due to low bioavailability of these nuclides, because the accumulation in soil is strong and transfer rates to milk and meat are low. For nuclides with high bioavailability such as H-3, Cl-36, Ni-59, Mo-93, Tc-99, I-129 and Cs-135, consumption of vegetables and root crops dominates the exposure. The short physical half-live restricts the accumulation in soil for Cs-137. The high contribution to the EDF from root crops is caused by high translocation rates to roots from surface deposited radionuclides due to irrigation. External exposure dominates for Nb-94 and Ho-166m due to relatively high γ-energies and accumulation in soil, in combination with low uptake in biota.

The ranges of uncertainties are the lowest for those radionuclides for which consumption of drinking water dominates the exposure, because consumption values and well water volumes are parameters with low ranges. For the nuclides with high bioavailability the calculated ranges are the widest about or above one order of magnitude because the values of uptake factors have considerably higher variation, mostly within two orders of magnitude, see Tables A-1 – A-4.

Table 4-1 EDFs to the most exposed individual (Sv/year) at year 10 000 from continuous releases of 1 Bq/year to a well. Minimum and maximum values correspond to 99 % confidence interval of calculated distributions.

Nuclide	Mean	Std	Min	Max
H-3	1.3E-14	1.8E-15	8.9E-15	1.8E-14
Be-10	6.7E-13	1.4E-13	4.0E-13	1.2E-12
C-14	3.2E-13	4.8E-14	2.3E-13	4.7E-13
Cl-36	9.8E-13	3.0E-13	5.6E-13	2.5E-12
Co-60	1.4E-12	2.1E-13	9.8E-13	2.1E-12
Ni-59	7.9E-14	4.2E-14	3.2E-14	2.7E-13
Ni-63	8.2E-14	1.6E-14	5.2E-14	1.3E-13
Se-79	3.6E-12	2.0E-12	1.4E-12	1.5E-11
Sr-90	1.7E-11	3.4E-12	1.0E-11	2.9E-11
Zr-93	2.0E-13	6.3E-14	1.1E-13	4.3E-13
Nb-94	4.6E-12	2.2E-12	1.2E-12	1.1E-11
Mo-93	2.8E-12	1.5E-12	1.2E-12	1.0E-11
Tc-99	7.4E-13	3.6E-13	3.0E-13	2.1E-12
Pd-107	3.1E-14	1.3E-14	1.5E-14	9.2E-14
Ag-108m	1.8E-12	4.4E-13	1.1E-12	3.5E-12
Sn-126	5.2E-12	2.5E-12	2.4E-12	1.6E-11
I-129	1.2E-10	7.7E-11	5.4E-11	5.0E-10
Cs-135	2.6E-12	1.2E-12	1.2E-12	8.4E-12
Cs-137	7.9E-12	1.2E-12	5.5E-12	1.2E-11
Sm-151	4.1E-14	6.1E-15	2.9E-14	6.1E-14
Ho-166m	2.7E-12	6.2E-13	1.4E-12	4.4E-12
Pb-210	2.5E-10	2.9E-11	1.8E-10	3.4E-10
Ra-226	1.6E-10	3.4E-11	1.0E-10	3.1E-10
Ac-227	4.7E-10	7.0E-11	3.2E-10	7.0E-10
Th-229	5.1E-10	1.9E-10	2.4E-10	1.2E-09
Th-230	2.6E-10	1.1E-10	1.1E-10	6.5E-10
Th-232	2.9E-10	1.1E-10	1.2E-10	7.1E-10
Pa-231	8.5E-10	3.2E-10	3.9E-10	2.1E-09
U-233	2.8E-11	7.4E-12	1.7E-11	6.2E-11
U-234	2.8E-11	7.3E-12	1.7E-11	6.1E-11
U-235	2.6E-11	6.7E-12	1.6E-11	5.7E-11
U-236	2.6E-11	6.8E-12	1.6E-11	5.8E-11
U-238	2.5E-11	6.4E-12	1.5E-11	5.4E-11
Np-237	6.1E-11	1.9E-11	3.7E-11	1.7E-10
Pu-238	8.7E-11	1.0E-11	6.4E-11	1.2E-10
Pu-239	2.8E-10	1.2E-10	1.2E-10	7.5E-10
Pu-240	2.3E-10	8.7E-11	1.1E-10	5.8E-10
Pu-242	2.9E-10	1.2E-10	1.2E-10	7.8E-10
Am-241	9.1E-11	1.4E-11	6.2E-11	1.4E-10
Am-242m	7.3E-11	9.0E-12	5.4E-11	1.0E-10
Am-243	1.7E-10	6.5E-11	8.1E-11	4.3E-10
Cm-244	4.1E-11	4.5E-12	3.0E-11	5.4E-11
Cm-245	2.1E-10	7.8E-11	9.5E-11	5.2E-10
Cm-246	1.8E-10	6.2E-11	8.8E-11	4.3E-10

Table 4-2 Calculated times (years) to reach 50, 90 and 99 % of the EDFs at 10 000 years for continuous release to a well

Nuclide	50 %	90 %	99 %
H-3	3	11	23
Be-10	19	5100	9100
C-14	1	4	110
Cl-36	20	74	160
Co-60	1	3	13
Ni-59	600	5400	8600
Ni-63	1	44	290
Se-79	85	390	900
Sr-90	1	18	93
Zr-93	280	5800	9200
Nb-94	1700	5600	8700
Mo-93	180	1800	4100
Tc-99	28	140	320
Pd-107	260	3400	6500
Ag-108m	17	210	600
Sn-126	230	2500	5600
I-129	250	3900	8200
Cs-135	600	6300	9400
Cs-137	1	4	58
Sm-151	1	3	92
Ho-166m	250	2600	6100
Pb-210	1	3	10
Ra-226	9	1100	3700
Ac-227	1	6	65
Th-229	800	7200	9700
Th-230	1600	8200	9900
Th-232	1700	8200	9900
Pa-231	1500	8000	9900
U-233	36	1700	4300
U-234	35	1700	4300
U-235	34	1700	4300
U-236	35	1700	4300
U-238	32	1700	4300
Np-237	57	1800	4400
Pu-238	1	12	220
Pu-239	1700	8100	9900
Pu-240	800	7100	9700
Pu-242	2000	8400	9900
Am-241	2	250	1700
Am-242m	1	35	430
Am-243	600	6400	9500
Cm-244	1	3	28
Cm-245	1000	7400	9800
Cm-246	530	6500	9600

Table 4-3 Contributions from various exposure pathways for release to a well (%).

	Consumption of					Inhalation	External exposure
Nuclide	**water**	**milk**	**meat**	**vegetables**	**root crops**	**dust**	**ground**
H-3	44	20	4	11	20	0	0
Be-10	52	0	1	12	22	13	0
C-14	58	14	8	7	13	0	0
Cl-36	31	12	4	13	40	0	0
Co-60	64	2	3	8	20	0	3
Ni-59	25	9	1	16	48	1	0
Ni-63	55	20	3	10	13	0	0
Se-79	26	4	1	16	54	0	0
Sr-90	51	2	1	12	34	0	0
Zr-93	44	0	0	5	9	41	0
Nb-94	12	0	0	2	4	2	80
Mo-93	29	1	1	16	53	0	0
Tc-99	28	0	0	49	23	0	0
Pd-107	39	1	0	14	44	2	0
Ag-108m	40	0	1	12	31	0	16
Sn-126	29	2	1	24	42	0	0
I-129	29	6	5	20	40	0	0
Cs-135	25	4	5	22	43	0	0
Cs-137	52	9	11	8	19	0	1
Sm-151	70	0	3	9	15	2	0
Ho-166m	23	0	1	4	5	4	64
Pb-210	83	1	0	10	7	0	0
Ra-226	55	1	1	25	13	5	0
Ac-227	69	0	0	9	15	8	0
Th-229	29	0	0	4	6	61	0
Th-230	24	0	0	3	5	67	0
Th-232	24	0	0	3	5	67	0
Pa-231	25	0	0	4	7	63	0
U-233	58	0	0	7	15	19	0
U-234	58	0	0	7	15	19	0
U-235	59	0	0	7	15	18	0
U-236	59	0	0	7	15	19	0
U-238	59	0	0	7	15	18	0
Np-237	58	0	0	8	12	21	0
Pu-238	79	0	0	10	3	9	0
Pu-239	27	0	0	3	1	69	0
Pu-240	32	0	0	4	1	62	0
Pu-242	25	0	0	3	1	71	0
Am-241	70	0	0	8	1	21	0
Am-242m	77	0	0	9	1	12	0
Am-243	36	0	0	4	1	58	0
Cm-244	84	0	0	10	3	3	0
Cm-245	31	0	0	4	1	64	0
Cm-246	35	0	0	5	1	59	0

4.2 Lake

The most exposed individual in the lake module is assumed to take the water, unfiltered, from the lake. The plant products are taken from fields irrigated with unfiltered water, from the lake. Some soil is inadvertently ingested with the vegetables. Cattle producing humans annual consumption of meat and milk are kept on irrigated fields during the grazing season and are also assumed to consume water plants part of this time. When stabled they are fed with food produced on irrigated fields. All the year round they consume contaminated water from the lake. Cattle's intake of radionuclides is therefore due to

- Consumption of pasturage, contaminated by root uptake and retention of radionuclides from irrigation water
- Consumption of water plants
- Annual consumption of contaminated water
- Intake of soil when grazing

It is further assumed that the most exposed individual is externally exposed from ground when performing agricultural practices and thereby he also inhales contaminated soil particles. He catches and consumes fish and crustacean from the lake for his annual consumption.

The EDFs at year 10 000 from the lake module are shown in Table 4-4 as average, standard deviation and ranges corresponding to 99 % confidence interval. Table 4-5 shows the time it takes to reach 50, 90 and 99 % of the EDFs at year 10 000.

The whole spectrum of exposure pathways is represented in this module giving varying contributions to EDF depending on element's properties, see Table 4-6. For most radionuclides, except for H-3, Cl-36, Mo-93 and Ag-108m, consumption of fish is an important exposure pathway. It totally dominates for C-14, Co-60, Ni-63, Sn-126, Cs-isotopes, Pb-210, Th-isotopes and Pu-isotopes due to a high uptake in fish muscle, see Table A-12. For carbon this can be expected, as it is a main constituent in fish tissues. Cesium is also effectively taken up in fish, due to its similarity to potassium. Several studies on the Chernobyl fallout have verified that consumption of fish is a main exposure pathway for Cs-137 transferred to fresh waters (IAEA, 1996). The modelled lake is oligotrophic leading to a higher uptake of cesium isotopes than for a eutrophic lake (Rowan & Rasmussen, 1994). Eutrophic lakes usually have higher potassium concentration than oligotrophic ones. The exposures for Cs-137 could therefore be expected to be about a factor of three lower for a eutrophic lake.

For many radionuclides consumption of crustacean (i. e. crayfish) gives high contribution to the EDF. Consumption of cray-fish totally dominates for Ag-108m, Sm-151, Ho-166m, and Ac-227 as well as for isotopes of americium and curium, because of their high uptake values for crustacean. Bioaccumulation factors for invertebrates were used to calculate this uptake in lack of data for crustacean. The factors are however from the beginning of the 70's and may be seen as screening values (Thompson et al, 1972).

When comparing with Canadian work (Davis et al, 1993) no exposure via consumption of shellfish from fresh water is considered.

Consumption of cereals, irrigated with lake water is the dominant exposure pathway for the radionuclides Mo-93 and contributes also significantly to the EDFs for Cl-36, Tc-99 and Ag-108m. Especially Cl-36 has extremely high uptake factors for crops, see Tables A-1 – A-4 and also high transfer to milk and meat which leads to that consumption of milk is the dominant exposure pathway for Cl-36. Also for H-3, Ni-59, Ni-63, Tc-99 and I-129 consumption of milk and meat gives considerable contributions to the EDFs.

Consumption of water gives varying contributions to EDFs for all radionuclides. For uranium isotopes the contributions are the highest, above 30 %, while for the other actinides they vary from about 10 up to 20 %. For Pa-231, Pu-239, Pu-240 and Pu-242 inhalation of resuspended particles from the irrigated soil gives substantial contributions in similarity to the well case and for the same reasons. These elements have a strong accumulation in soil, low bioavailability and high dose coefficients for exposure through inhalation, see Table 2-1.

External exposure gives contribution to the EDF for Ho-166m because of high γ-energies and low uptake in biota. Nb-94 has also high γ-energies but the high contribution from the fish pathway overwhelms the external exposure.

The variations in the results have a wider range for the lake than for the well. Results for cesium and uranium isotopes show the lowest ranges because the EDFs for these radionuclides are so dominated by the exposure pathways directly related to radionuclide concentrations in water. In addition there is a consensus about the uptake to fish for these radionuclides. The ranges in the results for Nb-94 are the widest though consumption of fish totally dominates the exposure. This is because the found uptake factors for fish show high uncertainty leading to wide ranges for that factor, see Table A-12. The ranges of variation increase if there are significant contributions to EDF from consumption of terrestrial products.

The accumulation in soils, described by the Kd-values, influences substantially the time to reach 90 and 99 % of the EDFs, see e.g. Pu-isotopes and see Tables 4-5 and 4-6.

Table 4-4 EDFs to the most exposed individual (Sv/year) at year 10 000 from continuous releases of 1 Bq/year to a lake ecosystem. Minimum and maximum values correspond to 99 % confidence interval of calculated distributions.

Nuclide	Mean	Std	Min	Max
H-3	5.3E-18	1.9E-18	2.3E-18	1.3E-17
Be-10	2.3E-16	1.8E-16	5.8E-17	1.0E-15
C-14	1.3E-14	9.8E-15	9.5E-16	4.0E-14
Cl-36	4.8E-16	1.4E-16	2.4E-16	1.0E-15
Co-60	6.5E-16	3.0E-16	1.8E-16	1.7E-15
Ni-59	2.8E-17	1.4E-17	8.0E-18	8.4E-17
Ni-63	5.3E-17	2.9E-17	1.4E-17	1.7E-16
Se-79	7.0E-15	3.0E-15	2.4E-15	1.6E-14
Sr-90	5.6E-15	4.2E-15	1.6E-15	2.6E-14
Zr-93	1.4E-16	8.6E-17	3.8E-17	4.0E-16
Nb-94	4.3E-15	6.4E-15	3.6E-16	3.9E-14
Mo-93	7.5E-16	3.2E-16	2.8E-16	2.2E-15
Tc-99	8.9E-17	4.5E-17	3.5E-17	3.0E-16
Pd-107	1.0E-17	7.2E-18	3.0E-18	4.2E-17
Ag-108m	4.3E-16	2.1E-16	1.7E-16	1.3E-15
Sn-126	2.4E-14	2.3E-14	2.6E-15	1.2E-13
I-129	5.7E-14	2.3E-14	1.8E-14	1.4E-13
Cs-135	2.2E-14	5.2E-15	1.2E-14	3.8E-14
Cs-137	1.3E-13	2.9E-14	7.2E-14	2.1E-13
Sm-151	2.0E-17	1.4E-17	4.9E-18	8.3E-17
Ho-166m	5.2E-16	3.0E-16	1.7E-16	1.7E-15
Pb-210	2.0E-13	6.0E-14	9.0E-14	3.6E-13
Ra-226	4.6E-14	2.0E-14	1.8E-14	1.2E-13
Ac-227	2.2E-13	1.3E-13	5.6E-14	7.5E-13
Th-229	1.7E-13	1.0E-13	4.6E-14	6.2E-13
Th-230	7.4E-14	4.5E-14	2.1E-14	2.7E-13
Th-232	8.1E-14	5.0E-14	2.3E-14	2.9E-13
Pa-231	7.4E-14	2.5E-14	3.4E-14	1.6E-13
U-233	3.4E-15	1.1E-15	1.7E-15	8.0E-15
U-234	3.3E-15	1.1E-15	1.6E-15	7.7E-15
U-235	3.1E-15	1.0E-15	1.6E-15	7.4E-15
U-236	3.1E-15	1.0E-15	1.6E-15	7.3E-15
U-238	3.0E-15	9.7E-16	1.5E-15	7.0E-15
Np-237	3.8E-14	4.7E-14	5.8E-15	3.0E-13
Pu-238	1.7E-14	9.9E-15	6.6E-15	5.8E-14
Pu-239	3.1E-14	1.6E-14	1.1E-14	9.6E-14
Pu-240	2.9E-14	1.5E-14	1.0E-14	9.4E-14
Pu-242	3.1E-14	1.5E-14	1.1E-14	9.3E-14
Am-241	4.1E-14	3.0E-14	9.7E-15	1.8E-13
Am-242m	3.8E-14	2.8E-14	8.8E-15	1.6E-13
Am-243	4.4E-14	3.1E-14	1.2E-14	1.8E-13
Cm-244	2.2E-14	1.6E-14	5.6E-15	9.3E-14
Cm-245	4.8E-14	3.3E-14	1.3E-14	2.0E-13
Cm-246	4.7E-14	3.3E-14	1.3E-14	2.0E-13

Table 4-5 Calculated times (years) to reach 50, 90 and 99 % of the EDFs at 10 000 years for a continuous release to a lake.

Nuclide	50 %	90 %	99 %
H-3	1	1	3
Be-10	1	115	5100
C-14	1	1	2
Cl-36	2	7	20
Co-60	1	5	12
Ni-59	5	1800	6800
Ni-63	2	14	140
Se-79	1	18	170
Sr-90	1	3	23
Zr-93	1	4	2800
Nb-94	4	900	4800
Mo-93	82	1200	3100
Tc-99	1	9	37
Pd-107	1	550	3900
Ag-108m	2	86	360
Sn-126	6	35	830
I-129	9	2000	6600
Cs-135	2	11	630
Cs-137	2	9	20
Sm-151	1	6	20
Ho-166m	2	770	3900
Pb-210	1	1	2
Ra-226	3	250	2200
Ac-227	2	8	20
Th-229	23	1700	8000
Th-230	27	3000	8800
Th-232	3	3000	8900
Pa-231	130	5700	9500
U-233	3	66	1600
U-234	3	67	1600
U-235	3	65	1600
U-236	3	65	1600
U-238	3	60	1600
Np-237	3	110	1400
Pu-238	13	50	160
Pu-239	80	4400	9200
Pu-240	42	3200	8700
Pu-242	110	4800	9400
Am-241	1	14	530
Am-242m	1	7	10
Am-243	3	1200	6800
Cm-244	1	5	14
Cm-245	2	1800	8000
Cm-246	2	1100	7200

Table 4-6 Contributions from various exposure pathways for release to a lake (%).

	Consumption of								Inhalation	External exposure
Nuclide	**water**	**fish**	**shell-fish**	**milk**	**meat**	**cereals**	**root crops**	**vege-tables**	**dust**	**ground**
H-3	7	0	0	64	14	10	3	2	0	0
Be-10	10	77	1	0	2	4	4	2	0	0
C-14	0	98	2	0	0	0	0	0	0	0
Cl-36	4	9	2	44	15	21	5	2	0	0
Co-60	9	61	9	3	7	4	3	1	0	0
Ni-59	5	37	2	38	5	5	5	3	0	0
Ni-63	6	45	3	38	4	1	1	1	0	0
Se-79	1	84	1	4	0	7	2	1	0	0
Sr-90	11	49	5	10	4	10	8	3	0	0
Zr-93	16	71	1	0	0	4	4	2	1	0
Nb-94	1	94	0	0	0	0	0	0	0	4
Mo-93	9	7	0	9	5	52	12	6	0	0
Tc-99	15	15	0	1	0	18	14	36	0	0
Pd-107	8	59	12	5	1	6	6	3	0	0
Ag-108m	11	2	46	0	5	18	8	4	0	5
Sn-126	0	95	2	1	0	1	0	0	0	0
I-129	4	30	0	26	20	12	4	3	0	0
Cs-135	0	97	0	1	1	0	0	0	0	0
Cs-137	0	98	0	1	1	0	0	0	0	0
Sm-151	10	21	52	0	9	3	2	1	0	0
Ho-166m	8	16	42	0	6	2	2	1	1	19
Pb-210	7	86	4	1	0	1	1	1	0	0
Ra-226	13	36	20	5	2	13	3	7	0	0
Ac-227	10	34	49	0	0	3	2	1	0	0
Th-229	6	60	16	0	0	6	2	1	9	0
Th-230	6	58	16	0	0	7	1	1	11	0
Th-232	6	58	16	0	0	7	1	1	10	0
Pa-231	21	16	10	0	0	11	6	3	33	0
U-233	32	20	17	3	1	9	9	5	3	0
U-234	32	20	17	3	1	9	9	5	3	0
U-235	32	20	17	3	1	9	9	5	3	0
U-236	32	20	17	3	1	9	9	5	3	0
U-238	32	20	17	3	1	9	9	5	3	0
Np-237	6	75	13	0	0	2	1	1	2	0
Pu-238	24	56	12	0	0	1	1	3	3	0
Pu-239	17	42	9	0	0	1	1	2	28	0
Pu-240	18	45	10	9	0	1	1	2	23	0
Pu-242	17	41	9	0	0	1	1	2	30	0
Am-241	11	31	54	0	0	0	0	1	3	0
Am-242m	11	31	55	0	0	0	0	1	2	0
Am-243	10	28	50	0	0	0	0	1	11	0
Cm-244	11	31	55	0	0	0	0	1	0	0
Cm-245	9	28	48	0	0	0	0	1	13	0
Cm-246	10	28	49	0	0	0	0	1	11	0

4.3 Running waters

The most exposed individual in the running waters module is assumed to take the water, unfiltered, from the running waters. The plant products are taken from fields irrigated with unfiltered water, from that source. Some soil is inadvertently ingested with the vegetables. Cattle producing humans annual consumption of meat and milk are kept on irrigated fields during the grazing season and are also assumed to consume water plants part of this time. When stabled they are fed with food produced on irrigated fields. All the year round they consume contaminated water from the running waters. Cattle's intake of radionuclides is therefore due to

- Consumption of pasturage, contaminated by root uptake and retention of radionuclides from irrigation water
- Consumption of cereals, contaminated by root uptake and translocation from surface retention of radionuclides from irrigation water
- Annual consumption of contaminated water
- Intake of soil when grazing

It is further assumed that the most exposed individual is externally exposed from ground when performing agricultural practices and thereby he also inhales contaminated soil particles. He catches and consumes fish and crustacean from the running waters for his annual consumption.

The EDFs at year 10 000 for the running water module are shown in Table 4-7 as average values, standard deviation and ranges corresponding to 99 % confidence interval. Table 4-8 shows the time it takes to reach 50, 90 and 99 % of the EDFs given in Table 4-7. In Table 4-9 the percentage contributions to total EDF from all exposure pathways are shown.

The dominant exposure pathways vary depending on nuclide properties. The EDFs are, however, higher than for the lake because of lower water volumes used for mixing. The relative contribution to EDF from the different exposure pathways is similar to the results for the lake module, see section 4.2. This is because the two modules are identical regarding irrigation rates and exposure pathways. The main differences to the lake module are the exclusion of transfer to sediments and the lower water volumes.

The ranges of variation are very similar to the ones for the lake module (Section 4.2), usually within one order of magnitude.

The lower water volume leads to that a higher fraction of radionuclides is used for irrigation than in the lake module. This leads further to that the feed-back of radionuclides from soil to water leads to increased concentrations of the radionuclides in the water with time for the mobile radionuclides such as H-3, Cl-36, Se-79, Sr-90 and Tc-99. The times for reaching maximum dose will therefore incrase somewhat compared to the corresponding times for the lake module, see Tables 4-5 and 4-8.

Table 4-7 EDFs to the most exposed individual (Sv/year) at year 10 000 from continuous releases of 1 Bq/year to running waters. Minimum and maximum values correspond to 99 % confidence interval of calculated distributions.

Nuclide	Mean	Std	Min	Max
H-3	3.0E-17	1.0E-17	1.3E-17	6.6E-17
Be-10	1.4E-15	1.1E-15	3.4E-16	6.1E-15
C-14	7.7E-14	5.7E-14	6.2E-15	2.3E-13
Cl-36	2.8E-15	8.2E-16	1.4E-15	5.6E-15
Co-60	4.4E-15	2.0E-15	1.2E-15	9.8E-15
Ni-59	1.7E-16	7.9E-17	5.2E-17	5.0E-16
Ni-63	3.2E-16	1.7E-16	7.7E-17	1.1E-15
Se-79	4.1E-14	1.7E-14	1.4E-14	9.1E-14
Sr-90	3.2E-14	2.3E-14	1.1E-14	1.4E-13
Zr-93	8.4E-16	5.0E-16	2.2E-16	2.3E-15
Nb-94	2.5E-14	3.8E-14	2.3E-15	2.3E-13
Mo-93	4.4E-15	1.8E-15	1.6E-15	1.2E-14
Tc-99	5.2E-16	2.5E-16	1.8E-16	1.6E-15
Pd-107	6.1E-17	3.7E-17	1.9E-17	2.2E-16
Ag-108m	2.5E-15	1.2E-15	9.9E-16	7.8E-15
Sn-126	1.4E-13	1.4E-13	1.8E-14	7.4E-13
I-129	3.4E-13	1.3E-13	1.1E-13	8.1E-13
Cs-135	1.3E-13	3.1E-14	6.3E-14	2.3E-13
Cs-137	8.2E-13	1.9E-13	4.0E-13	1.5E-12
Sm-151	1.2E-16	7.2E-17	3.4E-17	4.1E-16
Ho-166m	3.0E-15	1.4E-15	1.2E-15	9.0E-15
Pb-210	1.2E-12	3.2E-13	5.4E-13	2.0E-12
Ra-226	2.7E-13	9.3E-14	1.2E-13	5.9E-13
Ac-227	1.4E-12	7.5E-13	3.7E-13	4.3E-12
Th-229	9.8E-13	5.2E-13	3.1E-13	3.2E-12
Th-230	4.3E-13	2.2E-13	1.4E-13	1.4E-12
Th-232	4.7E-13	2.4E-13	1.6E-13	1.5E-12
Pa-231	4.3E-13	1.3E-13	2.0E-13	8.8E-13
U-233	2.0E-14	5.3E-15	1.1E-14	3.8E-14
U-234	1.9E-14	5.1E-15	1.1E-14	3.7E-14
U-235	1.8E-14	4.9E-15	1.0E-14	3.5E-14
U-236	1.8E-14	4.9E-15	1.0E-14	3.5E-14
U-238	1.7E-14	4.7E-15	9.6E-15	3.4E-14
Np-237	2.2E-13	2.6E-13	4.1E-14	1.7E-12
Pu-238	1.2E-13	6.9E-14	4.6E-14	4.3E-13
Pu-239	1.8E-13	8.3E-14	6.8E-14	5.1E-13
Pu-240	1.7E-13	8.0E-14	6.5E-14	5.0E-13
Pu-242	1.8E-13	8.1E-14	6.8E-14	4.9E-13
Am-241	2.4E-13	1.5E-13	7.0E-14	8.3E-13
Am-242m	2.2E-13	1.4E-13	6.4E-14	7.8E-13
Am-243	2.6E-13	1.5E-13	8.0E-14	8.6E-13
Cm-244	1.4E-13	8.3E-14	3.6E-14	4.8E-13
Cm-245	2.8E-13	1.5E-13	8.8E-14	9.1E-13
Cm-246	2.7E-13	1.5E-13	8.4E-14	9.1E-13

Table 4-8 Calculated times (years) to reach 50, 90 and 99 % of the EDFs at 10 000 years for a continuous release to running waters.

Nuclide	50 %	90 %	99 %
H-3	3	10	20
Be-10	1	210	5400
C-14	1	3	6
Cl-36	18	67	140
Co-60	1	3	7
Ni-59	3	2200	7100
Ni-63	1	8	160
Se-79	7	180	600
Sr-90	1	7	69
Zr-93	1	3	3200
Nb-94	3	1100	5200
Mo-93	176	1500	3500
Tc-99	23	120	280
Pd-107	2	800	4400
Ag-108m	8	150	490
Sn-126	1	44	1400
I-129	32	2400	6900
Cs-135	1	3	1000
Cs-137	1	3	5
Sm-151	1	3	8
Ho-166m	2	800	3900
Pb-210	1	3	5
Ra-226	2	430	2600
Ac-227	1	3	7
Th-229	2	1600	8000
Th-230	6	2700	8900
Th-232	6	2700	8900
Pa-231	83	5700	9500
U-233	1	110	2200
U-234	1	110	2200
U-235	1	110	2200
U-236	1	110	2200
U-238	1	110	2200
Np-237	1	150	1800
Pu-238	1	4	100
Pu-239	42	4400	9300
Pu-240	12	3000	8700
Pu-242	67	4900	9400
Am-241	1	6	510
Am-242m	1	3	97
Am-243	1	1000	6900
Cm-244	1	3	6
Cm-245	2	1500	8000
Cm-246	1	940	7000

Table 4-9 Contributions from various exposure pathways for release to running waters (%).

	Consumption of								Inhalation	External exposure
Nuclide	**water**	**fish**	**shell-fish**	**milk**	**meat**	**cereals**	**root crops**	**vege-tables**	**dust**	**ground**
H-3	7	0	0	64	14	10	3	2	0	0
Be-10	10	77	1	0	2	4	4	2	1	0
C-14	0	98	2	0	0	0	0	0	0	0
Cl-36	4	8	2	44	15	20	5	2	0	0
Co-60	9	62	9	3	7	4	3	1	0	0
Ni-59	5	37	2	38	5	5	5	3	0	0
Ni-63	6	45	3	38	4	1	1	1	0	0
Se-79	1	84	1	4	0	7	2	1	0	0
Sr-90	11	49	5	10	4	10	8	3	0	0
Zr-93	16	71	1	0	0	5	3	2	1	0
Nb-94	1	94	0	0	0	0	0	0	0	4
Mo-93	9	7	0	9	5	52	12	6	0	0
Tc-99	16	15	0	1	0	18	14	36	0	0
Pd-107	8	58	12	5	1	6	6	3	0	0
Ag-108m	11	2	46	0	5	18	8	4	0	5
Sn-126	0	95	2	1	0	1	0	0	0	0
I-129	4	30	0	26	21	12	4	3	0	0
Cs-135	0	97	0	1	1	0	0	0	0	0
Cs-137	0	98	0	1	1	0	0	0	0	0
Sm-151	10	21	53	0	9	3	2	1	0	0
Ho-166m	8	16	42	0	6	2	2	1	1	19
Pb-210	7	86	4	1	0	1	1	1	0	0
Ra-226	13	36	20	5	2	13	3	7	0	0
Ac-227	10	34	49	0	0	3	2	1	0	0
Th-229	6	60	16	0	0	6	2	1	9	0
Th-230	6	58	16	0	0	7	1	1	11	0
Th-232	6	58	16	0	0	7	1	1	10	0
Pa-231	21	16	10	0	0	11	6	3	33	0
U-233	32	20	17	3	1	9	9	5	3	0
U-234	32	20	17	3	1	9	9	5	3	0
U-235	32	20	17	3	1	9	9	5	3	0
U-236	32	20	17	3	1	9	9	5	3	0
U-238	32	20	17	3	1	9	9	5	3	0
Np-237	6	75	13	0	0	2	1	1	2	0
Pu-238	24	57	12	0	0	1	1	3	3	0
Pu-239	17	42	9	0	0	1	1	2	28	0
Pu-240	18	45	9	0	0	1	1	2	23	0
Pu-242	17	41	9	0	0	1	1	2	30	0
Am-241	11	31	54	0	0	0	0	1	3	0
Am-242m	11	31	55	0	0	0	0	1	2	0
Am-243	10	28	50	0	0	0	0	1	11	0
Cm-244	11	31	55	0	0	0	0	1	0	0
Cm-245	9	28	48	0	0	0	0	1	13	0
Cm-246	10	28	49	0	0	0	0	1	11	0

4.4 Coast

The most exposed individual in the coast module is exposed through his annual consumption of fish from the brackish water. In addition, he consumes his annual amounts of milk and meat from animals consuming brackish water and water plants during a part of the year. Furthermore, he consumes some algae. No external exposure pathways are considered.

The EDFs at year 10 000 for the coast module, with standard deviations and ranges corresponding to 99 % confidence interval of the calculated distributions, are shown in Table 4-10. Table 4-11 shows the time it takes to reach 50, 90 and 99 % of these EDFs. The relative contributions to the EDF from various exposure pathways are shown in Table 4-12.

The dominant exposure pathway for most nuclides is consumption of fish. For H-3, Cl-36, Mo-93 and I-129 the dominant pathways are consumption of milk and meat due . relatively low transfer factors to fish compared to milk and meat. Nuclides with lower transfer factors to milk and meat, but still above 10^{-3} day/kg, have about equal contributions from consumption of fish as from consumption of milk and meat. This is the case for Sr-90 and Pd-107. Cesium isotopes are however an exception to this due the high transfer of cesium to fish muscle (Rowan & Rasmussen, 1994).

Consumption of algae gives minor contribution to EDF for all radionuclides. This is because it was assumed that the algae were taken from open coast where the concentrations of radionuclides are much lower than in the archipelago. For example, for many radionuclides the bioaccumulation factors for algae are higher or in some cases even substantially higher than the corresponding values for fish.

If algae were taken from the archipelago instead of the open coast Tc-99, Pd-107 and thorium isotopes have about ten times higher EDFs than if algae are taken from the archipelago. Sm-151, Ho-166m and curium isotopes have a threefold increase. However, today normal Swedish diet does not include algae gathered from Sweden.

The variations are in general wider for the coast module than for the lake and running water modules. This is due to higher uncertainties in the determination of volumes for coastal water compartments, in comparison to lakes' volumes, which have distinct limits. There are few exposure pathways in the coast module compared to the lake module.

Table 4-10 EDFs to the most exposed individual (Sv/year) at year 10 000 from continuous releases of 1 Bq/year to a bay of the Baltic Sea. Minimum and maximum values correspond to 99 % confidence interval of calculated distributions.

Nuclide	Mean	Std	Min	Max
H-3	3.2E-18	1.5E-18	9.9E-19	9.0E-18
Be-10	3.9E-16	3.9E-16	3.3E-17	2.1E-15
C-14	1.7E-15	2.4E-16	1.2E-15	2.4E-15
Cl-36	1.6E-16	6.4E-17	6.3E-17	3.9E-16
Co-60	5.1E-16	2.5E-16	1.1E-16	1.3E-15
Ni-59	2.2E-17	9.6E-18	4.7E-18	5.0E-17
Ni-63	5.2E-17	2.2E-17	1.1E-17	1.1E-16
Se-79	1.4E-14	4.3E-15	6.8E-15	2.6E-14
Sr-90	1.3E-15	6.7E-16	3.2E-16	3.6E-15
Zr-93	9.0E-17	5.3E-17	1.6E-17	2.5E-16
Nb-94	2.1E-16	1.7E-16	2.1E-17	8.9E-16
Mo-93	9.2E-17	5.4E-17	1.5E-17	3.1E-16
Tc-99	1.7E-17	1.4E-17	1.4E-18	6.9E-17
Pd-107	1.2E-18	8.4E-19	2.0E-19	4.4E-18
Ag-108m	1.1E-15	5.0E-16	2.9E-16	2.6E-15
Sn-126	9.9E-16	8.3E-16	1.4E-16	4.4E-15
I-129	1.8E-14	8.8E-15	6.0E-15	5.6E-14
Cs-135	7.8E-16	2.4E-16	3.6E-16	1.5E-15
Cs-137	4.8E-15	1.4E-15	2.3E-15	9.2E-15
Sm-151	5.9E-18	4.5E-18	7.3E-19	2.4E-17
Ho-166m	1.3E-16	1.1E-16	1.4E-17	6.0E-16
Pb-210	8.5E-14	2.6E-14	4.0E-14	1.7E-13
Ra-226	1.6E-14	6.4E-15	4.9E-15	3.6E-14
Ac-227	1.8E-13	1.7E-13	1.5E-14	9.1E-13
Th-229	1.3E-14	1.1E-14	8.4E-16	4.9E-14
Th-230	5.4E-15	4.5E-15	3.6E-16	2.1E-14
Th-232	5.9E-15	5.0E-15	3.9E-16	2.3E-14
Pa-231	1.3E-14	1.2E-14	1.1E-15	6.8E-14
U-233	2.6E-15	1.1E-15	7.5E-16	5.9E-15
U-234	2.5E-15	1.1E-15	7.2E-16	5.7E-15
U-235	2.4E-15	1.0E-15	6.9E-16	5.4E-15
U-236	2.4E-15	1.0E-15	6.9E-16	5.4E-15
U-238	2.3E-15	1.0E-15	6.6E-16	5.2E-15
Np-237	2.0E-15	2.0E-15	1.8E-16	1.1E-14
Pu-238	5.1E-15	2.2E-15	1.4E-15	1.1E-14
Pu-239	6.4E-15	2.9E-15	1.5E-15	1.5E-14
Pu-240	6.3E-15	2.9E-15	1.5E-15	1.5E-14
Pu-242	6.1E-15	2.8E-15	1.4E-15	1.4E-14
Am-241	1.6E-14	9.4E-15	2.6E-15	4.6E-14
Am-242m	1.5E-14	8.8E-15	2.5E-15	4.3E-14
Am-243	1.6E-14	9.5E-15	2.6E-15	4.6E-14
Cm-244	4.0E-15	2.7E-15	7.5E-16	1.5E-14
Cm-245	1.6E-14	1.2E-14	2.7E-15	6.5E-14
Cm-246	1.6E-14	1.2E-14	2.7E-15	6.4E-14

Table 4-11 Calculated times (years) to reach 50, 90 and 99 % of the EDFs at 10 000 years for a continuous release to a bay of the Baltic Sea.

Nuclide	50 %	90 %	99 %
H-3	1	1	1
Be-10	1	1	1
C-14	1	1	1
Cl-36	1	1	1
Co-60	1	1	1
Ni-59	1	1	1
Ni-63	1	1	1
Se-79	1	1	1
Sr-90	1	1	1
Zr-93	1	2	2
Nb-94	1	1	1
Mo-93	1	1	1
Tc-99	1	1	1
Pd-107	1	1	1
Ag-108m	1	1	1
Sn-126	1	2	2
I-129	1	1	1
Cs-135	1	1	1
Cs-137	1	1	1
Sm-151	1	2	2
Ho-166m	1	1	1
Pb-210	1	1	1
Ra-226	1	1	1
Ac-227	1	1	1
Th-229	1	3	3
Th-230	1	3	3
Th-232	1	3	3
Pa-231	1	2	2
U-233	1	1	1
U-234	1	1	1
U-235	1	1	1
U-236	1	1	1
U-238	1	1	1
Np-237	1	1	1
Pu-238	1	2	2
Pu-239	1	2	2
Pu-240	1	2	2
Pu-242	1	2	2
Am-241	1	1	1
Am-242m	1	1	1
Am-243	1	1	1
Cm-244	1	2	2
Cm-245	1	3	3
Cm-246	1	3	3

Table 4-12 Contributions from various exposure pathways for release to the coast (%).

	Consumption of			
Nuclide	**Fish**	**Algae**	**Milk**	**Meat**
H-3	1	0	82	17
Be-10	99	0	0	1
C-14	96	0	2	1
Cl-36	1	0	74	25
Co-60	90	0	3	12
Ni-59	63	0	32	4
Ni-63	63	0	32	4
Se-79	99	0	1	0
Sr-90	52	0	32	16
Zr-93	99	1	0	0
Nb-94	99	0	0	0
Mo-93	42	0	38	21
Tc-99	93	3	3	1
Pd-107	54	2	35	10
Ag-108m	98	0	0	1
Sn-126	83	0	12	4
I-129	24	0	43	32
Cs-135	68	0	15	17
Cs-137	68	0	15	17
Sm-151	75	1	1	23
Ho-166m	75	1	0	24
Pb-210	97	0	2	0
Ra-226	84	0	10	5
Ac-227	100	0	0	0
Th-229	97	3	0	0
Th-230	97	3	0	0
Th-232	97	3	0	0
Pa-231	99	0	1	0
U-233	94	0	4	2
U-234	94	0	4	2
U-235	94	0	4	2
U-236	94	0	4	2
U-238	94	0	4	2
Np-237	98	0	0	2
Pu-238	99	1	0	0
Pu-239	99	1	0	0
Pu-240	99	1	0	0
Pu-242	99	1	0	0
Am-241	99	1	0	0
Am-242m	99	1	0	0
Am-243	99	1	0	0
Cm-244	99	1	0	0
Cm-245	99	1	0	0
Cm-246	99	1	0	0

4.5 Agricultural land

The most exposed individual in the agricultural land module takes his vegetative products grown on the top soil compartment. He inadvertentely consumes some soil when consuming vegetables. The animals' foodstuffs are also produced there. He obtains external exposure when staying on the fields and he inhales contaminated soil particles when performing agricultural practices.

The EDFs at year 10 000 with standard deviations and ranges corresponding to 99 % confidence interval of the calculated distributions for the agricultural land module are shown it Table 4-13. Table 4-14 shows the time it takes to reach 50, 90 and 99 % of those EDFs. The relative contribution to total EDF from various exposure pathways is shown in Table 4-15.

Consumption of crops is the dominating exposure pathway for most nuclides except for Zr-93, actinides and for nuclides with high-energy gamma radiation such as Nb-94 and Ho-166m (see Table 2-1). External exposure from ground dominates totally the EDFs for Nb-94 and Ho-166m. The major exposure pathway for actinides is inhalation of resuspended particles from soil and it is also important for Zr-93. This reflects the low bioavailability of these elements and high dose conversion factors for inhalation.

The variation of results are considerably larger from this module in comparison to the others, reflecting the large variations in input data, see further discussion in section 5.4.

Table 4-13 EDFs to the most exposed individual (Sv/year) at year 10 000 from continuous releases of 1 Bq/year to groundwater contaminating agricultural land. Minimum and maximum values correspond to 99 % confidence interval of calculated distributions.

Nuclide	Mean	Std	Min	Max
H-3	1.3E-15	1.8E-15	1.8E-16	1.3E-14
Be-10	1.3E-14	2.1E-14	3.3E-16	1.5E-13
C-14	2.0E-17	3.0E-17	1.5E-18	2.2E-16
Cl-36	4.0E-13	5.6E-13	4.0E-14	4.0E-12
Co-60	5.0E-18	2.5E-17	2.7E-21	2.3E-16
Ni-59	1.1E-14	1.6E-14	3.2E-16	8.8E-14
Ni-63	7.6E-17	2.4E-16	2.8E-19	1.3E-15
Se-79	1.6E-12	2.4E-12	1.4E-13	1.5E-11
Sr-90	2.8E-14	4.8E-14	5.2E-16	3.1E-13
Zr-93	3.0E-15	4.2E-15	7.7E-17	2.6E-14
Nb-94	2.4E-13	2.9E-13	2.6E-14	2.2E-12
Mo-93	8.7E-13	1.1E-12	6.7E-14	8.2E-12
Tc-99	5.9E-14	1.4E-13	1.4E-15	6.5E-13
Pd-107	2.7E-15	3.6E-15	2.3E-16	2.5E-14
Ag-108m	1.8E-14	3.2E-14	3.3E-16	1.9E-13
Sn-126	1.2E-12	1.7E-12	1.0E-13	1.2E-11
I-129	5.0E-11	7.9E-11	2.3E-12	4.9E-10
Cs-135	3.1E-13	5.2E-13	5.4E-15	3.5E-12
Cs-137	1.7E-16	5.0E-16	6.5E-19	3.8E-15
Sm-151	7.0E-19	1.3E-18	9.7E-21	9.8E-18
Ho-166m	3.4E-14	5.0E-14	2.1E-15	3.5E-13
Pb-210	3.4E-15	1.1E-14	9.6E-18	7.7E-14
Ra-226	7.2E-12	1.1E-11	4.2E-13	7.4E-11
Ac-227	7.4E-16	1.9E-15	1.1E-17	1.2E-14
Th-229	4.6E-12	1.1E-11	6.4E-14	7.9E-11
Th-230	3.0E-12	7.5E-12	4.2E-14	5.2E-11
Th-232	3.4E-12	8.5E-12	4.7E-14	5.9E-11
Pa-231	6.1E-12	1.2E-11	9.8E-14	6.9E-11
U-233	3.7E-13	5.4E-13	2.2E-14	3.9E-12
U-234	3.6E-13	5.3E-13	2.1E-14	3.8E-12
U-235	3.4E-13	5.0E-13	2.0E-14	3.6E-12
U-236	3.4E-13	4.9E-13	2.0E-14	3.6E-12
U-238	3.1E-13	4.5E-13	1.9E-14	3.3E-12
Np-237	2.1E-12	3.3E-12	7.8E-14	2.2E-11
Pu-238	3.6E-16	1.0E-15	5.8E-18	6.0E-15
Pu-239	1.2E-12	2.8E-12	2.6E-14	1.8E-11
Pu-240	7.7E-13	1.9E-12	1.7E-14	1.2E-11
Pu-242	1.3E-12	3.2E-12	2.9E-14	2.0E-11
Am-241	6.9E-14	1.3E-13	1.2E-15	1.1E-12
Am-242m	6.6E-15	1.3E-14	1.1E-16	1.1E-13
Am-243	2.6E-12	4.0E-12	7.3E-14	2.8E-11
Cm-244	4.1E-18	8.1E-18	4.9E-20	4.8E-17
Cm-245	1.0E-12	1.7E-12	1.7E-14	1.0E-11
Cm-246	7.1E-13	1.2E-12	1.2E-14	7.4E-12

Table 4-14 Calculated times (years) to reach 50, 90 and 99 % of the EDFs at 10 000 years for continuous release to groundwater contaminating agricultural land.

Nuclide	50%	90%	99%
H-3	5	12	22
Be-10	6100	9200	10000
C-14	66	180	350
Cl-36	51	140	280
Co-60	22	43	69
Ni-59	5300	8800	10000
Ni-63	320	700	1100
Se-79	470	1300	2300
Sr-90	63	140	240
Zr-93	6100	9200	10000
Nb-94	5100	8600	9900
Mo-93	2200	5000	7600
Tc-99	130	360	700
Pd-107	4100	7600	9400
Ag-108m	330	800	1200
Sn-126	3400	7200	9500
I-129	5000	8700	9900
Cs-135	6100	9200	10000
Cs-137	120	230	380
Sm-151	310	700	1100
Ho-166m	2700	5900	8900
Pb-210	77	160	260
Ra-226	2100	4800	7700
Ac-227	86	180	280
Th-229	6200	9300	10000
Th-230	6800	9500	10000
Th-232	7000	9500	10000
Pa-231	6700	9400	10000
U-233	2900	6100	8300
U-234	2900	6100	8300
U-235	2900	6100	8300
U-236	2900	6100	8300
U-238	2900	6100	8300
Np-237	2900	6100	8300
Pu-238	320	700	1100
Pu-239	6700	9400	10000
Pu-240	6127	9300	10000
Pu-242	6900	9500	10000
Am-241	1200	2700	4500
Am-242m	500	1100	1700
Am-243	5600	9100	10000
Cm-244	72	150	240
Cm-245	6300	9300	10000
Cm-246	5800	9100	10000

Table 4-15 Contributions from various exposure pathways for release to agricultural land (%).

	Consumption of						Inhalation	External exposure
Nuclide	**milk**	**meat**	**cereals**	**root crops**	**vegetables**	**soil**	**dust**	**ground**
H-3	37	8	42	8	5	0	0	0
Be-10	0	2	38	32	18	1	9	0
C-14	57	30	0	0	0	3	10	0
Cl-36	34	11	43	9	3	0	0	0
Co-60	3	4	63	5	1	0	0	24
Ni-59	40	5	22	25	8	0	0	0
Ni-63	41	5	23	26	5	0	0	0
Se-79	17	4	60	12	6	0	0	0
Sr-90	8	4	39	12	37	0	0	0
Zr-93	0	1	44	9	3	4	40	0
Nb-94	0	0	3	1	0	0	0	96
Mo-93	5	3	71	15	6	0	0	0
Tc-99	0	0	8	1	91	0	0	0
Pd-107	7	2	35	40	16	0	0	0
Ag-108m	0	3	59	20	6	0	0	11
Sn-126	7	3	48	23	19	0	0	0
I-129	31	23	30	10	5	0	0	0
Cs-135	26	30	19	16	9	0	0	0
Cs-137	26	29	20	16	5	0	0	3
Sm-151	1	13	2	1	24	1	57	1
Ho-166m	0	0	0	0	2	0	5	92
Pb-210	1	0	50	43	4	1	1	0
Ra-226	2	1	64	2	30	0	1	0
Ac-227	0	0	1	4	11	0	83	0
Th-229	0	0	45	0	0	0	54	0
Th-230	0	0	47	0	0	0	52	0
Th-232	0	0	48	0	0	0	51	0
Pa-231	0	0	21	3	1	0	74	0
U-233	1	0	7	18	5	1	67	0
U-234	1	0	7	18	5	1	68	0
U-235	1	0	7	18	6	1	64	3
U-236	1	0	7	18	6	1	67	0
U-238	1	0	7	19	6	1	65	0
Np-237	0	0	5	5	8	0	81	0
Pu-238	0	0	0	0	0	0	99	0
Pu-239	0	0	0	0	0	0	99	0
Pu-240	0	0	0	0	0	0	99	0
Pu-242	0	0	0	0	0	0	99	0
Am-241	0	0	0	0	0	0	99	0
Am-242m	0	0	0	0	0	0	99	0
Am-243	0	0	0	0	0	0	99	0
Cm-244	0	0	0	0	0	0	99	0
Cm-245	0	0	0	0	1	0	98	0
Cm-246	0	0	0	0	1	0	98	0

4.6 Peat bog

The most exposed individual in the peat bog module uses the peat bog for production – directly on the peat bog or after it has been drained - of his annual demand of vegetative products such as cereals and root-crops. The vegetables grown on this soil are also assumed to be unproperly washed leading to inadvertent consumption of soil together with the vegetables. The milk and meat he annually consumes comes from cattle fed on hay and cereals grown on the peat bog. In addition, it is conservatively assumed that a part of the peat is used for burning, thus giving rise to exposure by inhalation of particles from the gases. He also inhales some soil particles when performing agricultural practices.

The EDFs at year 10 000 with standard deviation and ranges corresponding to 99 % confidence interval of the calculated distributions are shown for the peat module in Table 4-16. Table 4-17 shows the time it takes to reach 50, 90 and 99 % of those EDFs. The percentage contribution from various exposure pathways to the total EDF is shown in Table 4-18.

Several terrestrial exposure pathways are considered in the peat bog module in similarity to the agricultural land module. In addition inhalation of gases from combustion of peat is included. This pathway, however, gives insignificant contributions to any exposure. Inhalation of dust consisting of resuspended particles is on the other hand a major exposure pathway for Zr-93, Sm-151, Ho-166m and actinides, because of high accumulation, low bioavailability and high values for dose conversion factors for inhalation, see Table 2-1.

The consumption of cereals and root crops dominate the EDFs for nuclides with high root uptake factors e.g. Cl-36, Se-79, and Mo-93, I-129, Cs-isotopes, Pb-210 and Ra-226. Milk and meat give a major contribution to the total EDF for nuclides with high transfer factors to milk and meat, see e.g. C-14 and I-129.

External exposure gives important contributions to the total EDFs for Nb-94 and Ho-166m due to their relatively high γ-energies. The relative contribution is however lower than in the results for the agricultural land module due to various volumes of the peat and top soil compartments.

The variation in results is large in similarity to the results for the agricultural land module. This reflects the lack of knowledge of all these elements' behaviour in peat.

Table 4-16 EDFs to the most exposed individual (Sv/year) at year 10 000 from continuous releases of 1 Bq/year to a peat bog. Minimum and maximum values correspond to 99 % confidence interval of calculated distributions.

Nuclide	Mean	Std	Min	Max
H-3	3.6E-16	4.2E-16	6.1E-17	2.7E-15
Be-10	8.6E-13	1.7E-12	7.7E-14	9.8E-12
C-14	6.5E-15	1.3E-14	1.7E-16	9.8E-14
Cl-36	2.2E-11	3.0E-11	1.7E-12	2.0E-10
Co-60	2.8E-13	4.5E-13	1.0E-14	2.7E-12
Ni-59	2.7E-13	3.9E-13	4.1E-14	2.5E-12
Ni-63	1.6E-13	2.4E-13	8.0E-15	1.6E-12
Se-79	1.7E-09	2.4E-09	1.7E-10	1.7E-08
Sr-90	1.8E-11	2.6E-11	1.3E-12	1.8E-10
Zr-93	4.4E-13	5.6E-13	3.8E-14	3.5E-12
Nb-94	2.0E-12	3.3E-12	1.4E-13	2.4E-11
Mo-93	2.5E-12	3.6E-12	2.6E-13	2.6E-11
Tc-99	4.2E-13	1.5E-12	9.4E-16	1.1E-11
Pd-107	6.4E-14	9.0E-14	7.6E-15	7.1E-13
Ag-108m	1.9E-11	2.9E-11	8.8E-13	2.3E-10
Sn-126	8.6E-11	1.3E-10	8.7E-12	9.1E-10
I-129	3.0E-11	4.1E-11	3.3E-12	2.8E-10
Cs-135	2.7E-12	3.3E-12	3.7E-13	2.2E-11
Cs-137	3.5E-12	6.4E-12	1.9E-13	2.9E-11
Sm-151	6.0E-15	1.0E-14	4.4E-16	8.1E-14
Ho-166m	1.9E-12	2.8E-12	1.2E-13	2.0E-11
Pb-210	1.6E-11	3.0E-11	5.7E-13	2.3E-10
Ra-226	1.2E-09	2.1E-09	1.1E-10	1.6E-08
Ac-227	6.2E-11	9.2E-11	3.5E-12	6.5E-10
Th-229	7.0E-09	1.1E-08	5.1E-10	8.6E-08
Th-230	4.0E-09	6.4E-09	3.0E-10	5.0E-08
Th-232	4.4E-09	7.1E-09	3.4E-10	5.5E-08
Pa-231	3.5E-09	5.6E-09	2.0E-10	4.3E-08
U-233	6.1E-12	1.3E-11	5.7E-14	8.7E-11
U-234	5.9E-12	1.2E-11	5.5E-14	8.4E-11
U-235	5.4E-12	1.1E-11	5.1E-14	7.6E-11
U-236	5.5E-12	1.1E-11	5.2E-14	7.8E-11
U-238	5.1E-12	1.0E-11	4.8E-14	7.2E-11
Np-237	1.1E-10	1.5E-10	1.2E-11	1.0E-09
Pu-238	3.7E-11	5.4E-11	1.8E-12	3.6E-10
Pu-239	4.1E-10	8.1E-10	8.4E-12	6.4E-09
Pu-240	3.6E-10	6.7E-10	8.3E-12	4.9E-09
Pu-242	4.1E-10	8.3E-10	8.1E-12	6.8E-09
Am-241	2.0E-10	3.2E-10	1.3E-11	2.6E-09
Am-242m	6.3E-11	1.0E-10	4.1E-12	8.1E-10
Am-243	1.8E-09	2.8E-09	1.1E-10	2.5E-08
Cm-244	5.5E-12	8.3E-12	3.6E-13	6.1E-11
Cm-245	9.8E-10	1.7E-09	4.4E-11	1.2E-08
Cm-246	8.1E-10	1.4E-09	4.0E-11	1.0E-08

Table 4-17 Calculated times (years) to reach 50, 90 and 99 % the EDFs at 10 000 years for a continuous release to a peat bog.

Nuclide	50%	90%	99%
H-3	3	7	14
Be-10	1300	3600	5900
C-14	39	130	270
Cl-36	8	26	53
Co-60	6	18	35
Ni-59	500	1600	3100
Ni-63	72	240	500
Se-79	900	2800	4800
Sr-90	18	60	120
Zr-93	2200	5700	8100
Nb-94	900	2700	4800
Mo-93	18	60	120
Tc-99	5	14	28
Pd-107	370	1200	2300
Ag-108m	130	420	900
Sn-126	900	2800	4700
I-129	18	61	120
Cs-135	200	700	1400
Cs-137	24	80	170
Sm-151	79	270	600
Ho-166m	600	1800	3600
Pb-210	22	74	150
Ra-226	500	1700	3400
Ac-227	22	72	150
Th-229	3500	8100	9900
Th-230	4400	8800	9900
Th-232	4500	8800	9900
Pa-231	2100	5500	8000
U-233	140	490	1000
U-234	140	490	1000
U-235	140	490	1000
U-236	140	490	1000
U-238	140	490	1000
Np-237	430	1400	2800
Pu-238	74	250	500
Pu-239	900	2700	4700
Pu-240	800	2500	4500
Pu-242	900	2800	4800
Am-241	430	1500	2900
Am-242m	150	500	1100
Am-243	3500	8200	9900
Cm-244	18	60	130
Cm-245	2100	5800	8400
Cm-246	1800	5300	8100

Table 4-18 Contributions from various exposure pathways for release to peat (%).

	Consumption of					Inhalation		External exposure
Nuclide	**milk**	**meat**	**cereals**	**root crops**	**vege-tables**	**soil**	**dust**	**ground**
H-3	37	8	43	8	5	0	0	0
Be-10	0	2	28	24	30	2	14	0
C-14	57	31	0	0	0	3	10	0
Cl-36	34	11	43	9	3	0	0	0
Co-60	3	8	72	6	4	0	0	6
Ni-59	40	5	20	23	12	0	0	0
Ni-63	41	5	21	24	9	0	0	0
Se-79	18	4	60	13	6	0	0	0
Sr-90	7	4	32	10	47	0	0	0
Zr-93	0	1	33	7	3	5	51	0
Nb-94	0	0	18	4	2	1	1	75
Mo-93	5	3	70	15	8	0	0	0
Tc-99	0	0	4	0	95	0	0	0
Pd-107	6	2	32	38	22	0	0	0
Ag-108m	0	3	64	23	9	0	0	1
Sn-126	7	2	47	21	23	0	0	0
I-129	32	23	29	7	9	0	0	0
Cs-135	24	28	18	16	14	0	0	0
Cs-137	25	29	19	17	10	0	0	0
Sm-151	1	10	1	1	26	1	60	0
Ho-166m	0	2	0	0	10	1	35	52
Pb-210	2	0	46	43	6	1	2	0
Ra-226	2	1	56	1	37	0	1	0
Ac-227	0	0	1	1	13	0	84	0
Th-229	0	0	31	0	0	0	68	0
Th-230	0	0	32	0	0	0	67	0
Th-232	0	0	33	0	0	0	66	0
Pa-231	0	0	9	2	1	0	88	0
U-233	1	0	5	13	6	1	74	0
U-234	1	0	5	13	6	1	74	0
U-235	1	0	5	14	6	1	72	0
U-236	1	0	5	13	6	1	73	0
U-238	1	0	5	14	6	1	72	0
Np-237	0	0	7	6	9	0	78	0
Pu-238	0	0	0	0	0	0	99	0
Pu-239	0	0	0	0	0	0	99	0
Pu-240	0	0	0	0	0	0	99	0
Pu-242	0	0	0	0	0	0	99	0
Am-241	0	0	0	0	0	0	99	0
Am-242m	0	0	0	0	0	0	99	0
Am-243	0	0	0	0	0	0	99	0
Cm-244	0	0	0	0	0	0	98	0
Cm-245	0	0	0	0	1	0	98	0
Cm-246	0	0	0	0	1	0	98	0

5 Discussion

All models are simplifications of our perception of reality, and results from them therefore have an inherent uncertainty. Factors affecting the reliability can, according to IAEA (1989), be grouped into five classes.

1. Specification of the problem (scenario)
2. Formulation of the conceptual model
3. Formulation of the computational model
4. Estimation of parameter values
5. Calculation and documentation of results

The best way of studying model confidence is to compare model results with independent data sets. This is, however, not possible for studies of this type. It is possible to evaluate parts of biosphere processes during limited time periods. Knowledge about long-term processes may be improved by studies of natural analogues (Miller et al, 1994 and Bergström & Landström, 1993). One process that has been thoroughly investigated is the uptake of Cs-137 in fresh-water fish. Observations from the Chernobyl fallout have been used in several studies testing the compartment modelling for simulating the behaviour of Cs-137 in lake ecosystems (BIOMOVS, 1990a, IAEA draft, 1998). However, these tests were performed for a pulse release in contrast to the continuous releases relevant for safety analyses of high-level waste. The main conclusions were that compartment models are suitable to describe the turnover of Cs-137 in aquatic ecosystems and that increased model complexity did not increase the precision in model results (IAEA draft, 1998).

5.1 Uncertainty due to scenario description

Uncertainties due to specification of the scenario involve the whole method applied for the assessments. One simplification is that the modules describe constant conditions over time, i.e. no change of the biosphere is considered. There are high uncertainties coupled to any prognoses of future states of the biosphere and also of future behaviour of man. The use of constant conditions in the basic scenarios is, however, justified by the authorities (SSI, 1998), stating that the safety analysis always shall include a case based on current biosphere conditions. In SR 97 the future ecosystems affected by landrise are considered. One possible scenario is drying up of lakes. According to model calculations on drying up of a specific lake (Sundblad et al, 1988), EDFs for certain radionuclides increase. This was specially pronounced for Pu-239. The dose increase was hundredfold when sediments were transformed to soil.

Processes causing a possible retention of nuclides between point of release and primary recipients are not considered in this study. Such processes may cause a delay before nuclides reach biosphere, and therefore result in an overestimate of EDFs. On the other hand, if e.g. nuclides that have accumulated in a zone become available for exposure of

man an underestimation of EDFs can be expected. Such a case is formation of soil from former sediments due to land-rise.

Another simplification is the assumption that radionuclides released to ecosystem surface water (lake, running water, coast) first enter a compartment consisting of water within which they are immediately homogeneously mixed. It is possible that radionuclides enter through sediments instead, where they may be exposed to drastic changes in redox conditions, bacterial activity, temperature etc. This could cause an accumulation of elements in the sediments resulting in an inhomogeneous distribution and delayed entrance into the water mass. On the other hand, studies on groundwater seepage to lakes, show that there are distinct areas, mostly in shallow parts of lakes into which outflow of groundwater occurs (Sundblad et al, 1991a). The sedimentation rates are low in such areas leading to a low transfer of elements from the groundwater to the sediments, probably due to the high flow rates of the incoming groundwater. Vanek (1985) points out in a literature review that the lake sediments are of minor importance compared to the geological and hydrogeological conditions for the seepage of groundwater to lakes bottoms. More thorough-going studies of lakes in potential siting areas of a repository should therefore show whether this simplification by direct entrance of radionuclides to lake water is justified or not.

If the radionuclides pass through sediments, bottom-living organisms could be higher exposed by accumulated radionuclides compared to when nuclides are mixed in water before transfer to the sediments. This can lead to higher concentrations of nuclides for fish species feeding on benthic species. If an annual catch of fish is considered, higher values in some fish species may be compensated by lower in others. This gives an average concentration for the total catch from a specific lake.

5.2 Uncertainties in the conceptual model

5.2.1 General

A basic concept in all the models set up for the modules so far is that the ecosystems are divided into compartments between which there is an annual transfer of water and thereby material. Each compartment is assumed to comprise a physical area with the same properties regarding e.g. chemistry. Radionuclides are also assumed to be homogeneously distributed within the compartments. This is never the case in reality.

All processes in the typical ecosystems were modelled by rate constants. This may be a crude simplification concerning areas with gradients such as undisturbed soils and sediments. The steady-state values of the concentrations of radionuclides in soils are on the other hand not much affected by this simplification when ploughed soils and long periods of time are considered. Two different model approaches were compared in an earlier study: compartment concept and a model based on advection-dispersion equations with consideration of diffusion (Argärde et al, 1988). Concentrations of I-129 and Np-237 in a root zone after migration of radionuclides entering the soil from below were calculated with the two methods. The comparison showed that there was good agreement for the concentrations of the radionuclides in soil while the dynamics varied between them. The results from the more physically based model showed shorter times

for the build-up of radionuclides in soil, usually with a difference within a factor of two. The simplification with linear rate constants will consquently affect the times to reach stady-state conditions more than the obtained concentrations as those are so dependent on the value of the distribution coefficients.

Simplifications and assumptions in process descriptions relaated to the components in the modules are discussed in the sections below. It was, however, out of scope of this study to perform analyses of different conceptual models due to the large number of radionuclides and modules. Such a study ought to be performed separately for nuclides and modules that are crucial for exposure to humans.

5.2.2 Water components

Water turnover is a main process for the initial mixing of radionuclides released to water and also for their transfer further on in the ecosystem. It is the only process considered for the turnover of radionuclides in two of the modules, well and running waters. If irrigation is considered in these two modules this will also lead to a recuction, though minor, of the radionuclide content in water.

Well
EDFs obtained for the well module are inversely proportional to the well capacities. The annual release of radionuclides to the biosphere was assumed to be homogeneously mixed in a volume corresponding to the annual well capacity. Decreasing water capacities may therefore lead to increases of EDFs. This annual volume for mixing was obtained by assuming that reported capacities per hour were valid during a year. This may be questionable. Another assumption is that no processes are considered within the aquifer itself, such as adsorption of elements to solid material. Furthermore, a reduction of radionuclide concentrations due to filtration before water use is not considered. If this is taken into account lower EDFs may be expected, especially for elements that are strongly attached to particulate matter.

Running waters
No transfer of radionuclides from water to sediments is considered in the running waters module, as mentioned above. Homogenous dispersion is assumed over the cross section of running waters, and the concentrations of radionuclides in the water therefore only depend on the flow. This is of course a simplification since running waters may be effective in transporting suspended matters (Hynes, 1970). In many running waters of meandering character there are zones where material is deposited and others from which erosion occurs (Hynes, 1970). As no depletion of radionuclides due to transfer to the sediments was assumed, the levels of radionuclides in running water may be conservatively biased.

Lake and coast
For the other surface water modules, lake and coast, an average turnover rate of water was used for the whole water mass. In reality surface water consists of ”water parcels” with varying retention times, especially in lakes and bays. Water in sheltered areas and bays has longer retention time than water in open areas. Differences in temperature and salinity reduce the mixing between upper and deeper water layers. The simplification for

lake water, assuming homogeneous mixing, is probably of minor importance for small dimictic lakes as the time scales considered in the calculations cover thousands of years.

The coast module consists of compartments representing archipelago and the open coast. The open coast compartment represents a small part of the open sea in association with the bay. The boundaries of the open sea compartment is arbitrarily selected. However, one criterion was that it should be large enough to contain all fish for the group's consumption. On the other hand, EDFs for the open sea should be much lower than the coastal area's EDFs, which are the lowest among the modules.

Turnover rates of fresh water are quite easily obtained from regional information on runoff or site specific information on water retention times or flows. Such information is, however, lacking for coastal areas. In Engqvist (1997), water level fluctuations were considered, as the main driving force and this was the approach used to obtain data for these calculations. Engqvist (1997) has studied the water turnover rates in bays surrounding Aberg, so site specific values are available for that site. It is important to have good knowledge about water retention times.

5.2.3 Sediment processes

Transfer from water to sediments
In addition to water turnover, transfer of radionuclides and other elements to sediments is an important process for depletion of radionuclide content in water (Meili et al, 1989, Meili, 1994 and Andersson et al, 1990). The transfer may occur via gravitational settling of particles and diffusion. The main process for transfer of immobile elements from the water column to sediment seems to be settling of suspended particles **(**Bergström & Nordlinder, 1991a and Broberg, 1994). These particles may be inorganic as well as organic. Shaw et al (1994) has shown that the transport of uranium to sediments is due to transport of detritus. Such transfer is not taken into account. The transfer of all radionuclides to sediments is described by a generic expression based on K_d-values, mass sedimentation rates and concentrations of suspended matter, see section 3.4, while diffusion is neglected. Neglecting diffusion in the modules may lead to an over-estimation of the content in water of mobile radionuclides e.g. Cl-36 and I-129.

The expression adopted for modelling the transfer of radionuclides from water to sediments has been used since the 1980-ies (Hill & Lawson, 1980, Bergström, 1983, Bergström & Nordlinder, 1991b and BIOMOVS II, 1996b). In studies of Cs-137 fall-out it was recognised that this expression does not model the fast transport of Cs-137 to sediments satisfactorily (BIOMOVS, 1990a). Nordlinder et al (1997) therefore used an approach with two compartments, one for the dissolved fraction and another one for the particulate fraction. From the particle fraction the transfer to sediments was modelled. This approach gave a better agreement for the levels of Cs-137 in water, but not for the levels in sediments. When using such an approach in the lake module we obtained a much lower transfer of radionuclides to the sediments. One explanation to this could be that the conditions after the Chernobyl fall-out differed from the ones in this study. In the former, Cs-137 entered the water as a pulse release in particulate form, while the scenarios in this study were based on a continuous leakage of dissolved radionuclides. In a Canadian study (Davis et al, 1993) element specific net rates for transfer to the sediments were used. In other words they considered resuspension implicitly. The

values were based on varying sources and showed high variabilities even for the same element. Many earlier studies have also pointed out the difficulties with modelling the transfer of radionuclides to sediments.

Transfer from sediments to water
It is a well-known fact that earlier deposited elements are refound in the water mass. Resuspension has been shown to be an important process for Cs-137 transfer back to water in a shallow lake (Sundblad et al, 1991) and resuspended particles are a major source for sedimentation in brackish water (Blomqvist, 1992). There are several processes included in the transfer of elements from sediments to water as for example resuspension of particles to the water column due to waves and currents. Diffusion of radionuclides from sediment to the dissolved fraction in water is another process. Bioturbation, respiration and decomposition of organic material within the sediment also contribute to a transport back to the water column (Petr, 1977). The transfer of radionuclides back to water from the sediment is simplified with a generic expression, which is solely based on observed values for Cs-137. The problem of resuspension needs further consideration.

Turnover within sediments
The behaviour of radionuclides in sediments depends on chemical, biological and physical conditions in the sediments. According to Andersson et al (1992) the most important chemical factors for the fate of metal ions in sediments are pH, Eh, ionic strength and the amount of complex formers. Only physical processes are, however, considered in these calculations, assuming a high retention of all radionuclides in the sediments. In a field study, spiked sediments cores were studied during three years (Andersson et al, 1992). The study confirmed a high sorption of 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. Andersson et al (1992) verified that the diffusion of Tc-99 is low in sediments due to reducing conditions.

Addition of new top-sediments results in a transfer of the lowest layers of the top sediments to deeper sediments in the compartment models. The thickness of the top sediment is thus assumed constant. Studies of plutonium in sediments have verfified that a pulse release is refound as a peak in the sediments (Östlund & Hallberg, 1991). The organic fraction is, in the calculations, assumed to decompose completely, and does therefore not contribute to the growth of the sediment. This probably causes an overestimate of the concentrations of the nuclides in the sediments. However, as the EDFs do not include any exposure pathways related to the sediments, the turnover of radionuclides in the sediments is of minor importance for the results. The opposite would be valid if evolution causes former sediments to soil or a benthic food-chain is included.

5.2.4 Irrigation

Garden plots as well as agricultural areas are assumed to be irrigated with 150 mm water per year. This value is judged to be reasonable as a long term average from information on Swedish practices (Linner, SLU personal communication). The quantities vary from

one year to another, of course, due to variations in precipitation and temperature. The irrigation can also be seen as a result of flooding, causing transfer of radionuclide from water to soils but not to a high degree directly to vegetation. An increased irrigation would result in proportionally increased EDFs from exposure pathways related to irrigation such as consumption of cereals, root crops and vegetables. The EDFs could also increase with up to about 20 % if the irrigation occasions are doubled in the well module. Greenhouses, which obtain all water demands from irrigation (about 700 mm/year), are not considered for two reasons. First, it is unlikely that the major fraction of the annual consumption of crops is produced in greenhouses. Second, data for uptake in vegetation are not based on conditions applicable for greenhouses. If considering a greenhouse scenario higher exposure from consumption of vegetables could be expected if comparing the amounts of radionuclides transferred to soil by irrigation. This is most notable for Tc-99, see Table 4-3. For the other radionuclides the increase in EDF should be minor if assuming similar up-take as for an irrigated garden plot.

5.2.5 Turnover in soil and peat

Advection, bioturbation and erosion are the three main processes considered for turnover of radionuclides in soils in the modules. Capillary rise and diffusion are aggregated into one factor for the agricultural land module. This is of course a simplification of the complex interactions of chemical, biological and physical processes in soil. Microbial activities causing e.g. mobilisation of organic substances in soil (Wood, 1987), is not considered explicitly. The chemical speciation of an element and its chemical properties as well as the bioavailability are of fundamental importance for its future fate. In addition, environmental conditions such as climate, grain sizes, pH-values, Eh-potential and organic content in soils play an important role **(**Bunzel et al, 1986). Underestimation of the levels obtained for soils gives underestimates of the contribution to the values of EDFs for consumption of grops and vice versa.

Advection
The advective vertical transport is simulated with K_d-values and physical properties such as porosity and density in combination with the turnover of water (see section 3.2.2). The turnover of water in soil is based on the water balance assuming that the fraction of water transferred through a soil volume was the same as the annual runoff. This was considered to be a plausible assumption as most runoff consists of outflow of ground water due to low surface runoff (Grip & Rodhe, 1985). The hydraulic conductivity, expressed in metres per year, is not used since it is only valid at saturated conditions. A simplification is that the total pore volume is assumed to be water filled and available for exchange. This is not the case as there is hygroscopically bound water in soils, which does not participate in percolation. The hygroscopically bound water constitute 5 – 11 % of the volume depending on soil properties (Van Genuchten, 1980). The simplified approach would lead to an underestimate of the water turnover rate, and thus to higher EDFs.

Flow of water and supply of oxygen are major forces for migration of elements transferred to upper soil as well as out of the soil. This has also been verified from the control programmes of the restoration of the former uranium mine at Ranstad. That study showed a rapid decrease of heavy-metal concentrations in leakage water when the

tailings were covered with layers preventing infiltration and supply of oxygen (Aquilonius et al, 1999).

So far, no model has considered root-uptake as a pathway for transfer of radionuclides between deeper and upper soil. It was shown in a lysimeter experiment that roots transfer Cs-137 from deeper to upper soil (BIOMOVS II, 1996c). Field studies of naturally occurring Ra-226 has also shown that there is an accumulation of Ra-226 in the upper soil-horizons probably due to a transfer via root systems (Landström & Sundblad, 1986). This means a faster transport to upper soils from below than was used in the calculations. The consequence of this is that the time to obtain maximum EDFs in compartments with soil and peat may be shorter than is shown in Table 4-14. Improved knowledge about such processes for radionuclide transport is, hence, identified as an area for further studies.

Distribution factors, K_d
In dose assessments of this type it is generally assumed that elements entering the biosphere are dissolved, but that they immediately react with organic or inorganic matter (Davies et al, 1993 and BIOMOVS II, 1996). A distribution factor describes the steady-state concentration of the element on solids relative to dissolved. The distribution factor is usually element specific and dependent on soil properties or other factors. The interaction between the dissolved fraction of an element and its surrounding media result in retardation relative to the transport of water in porous media (soil or peat).

For heavy metals it has been shown that there are significant correlations between sorption on soil and organic soil content, pH, cation exchange capacity (CEC) and the content of iron (Lundberg et al, 1994). The mass of the nuclides is low compared to the naturally occurring mass of the same elements in the soil. Thus it is unlikely that radionuclides will interact physically, or chemically with the soil, while on the other hand the chemical conditions may influence the behaviour of the radionuclides. The values of the distribution factor influence the resulting concentration of the radio-nuclides in soils and thereby the EDFs. It is a major source to variations in results especially for the peat-module, see section 5.4.

Most elements are assumed to be strongly retarded in peat, as peat accumulates metals effectively (Owen & Otton, 1995). There is also a pronounced accumulation of naturally occurring elements like uranium and thorium in peat implying an active interaction between groundwater and the biosphere (Hedvall, 1997). It has been shown that gradients of uranium concentrations occur within peat bogs (Statens energiverk, 1985). Data for modelling a realistic treatment of the variuos elements, handled in this study were, however, not found. The EDFs from the peat-bog module are therefore affected by high uncertainties.

Several studies have tried to assess distribution factors for elements in soil. In a Canadian study, e.g. values for the same element varied within several orders of magnitude, up to a maximum of 5 orders for the same type of soil (Sheppard, 1990). Unfortunately it is difficult to get an explanation, since there usually is a lack of information on environmental conditions. Microbial activity may be as important as fixation into minerals for the variation (Wood, 1987). Moreover, naturally occurring elements may be located in mineral lattices leading to an overestimate of the available

solid form of the elements. Puigdomènech & Bergström (1995) performed a theoretical study of distribution factors for some elements typical of high-level waste by applying surface complexiation models. The results were in general lower than those found in field studies, implying that the incorporation in organic matter is an important process. One such example is iodine for which a low value is found for the distribution factor in geospheric systems, also often used in the biosphere, while field studies around a reprocessing plant showed that the half-depths varied from 4 to 10 cm after about 20 years implying a higher sorption capacity (Kantelo et al, 1981) than generic K_d-values show. (The half-depth is the depth required for the concentration to decrease to one half of its surface value.)

Biological processes
Bioturbation is considered as soil transport by earthworms. This has shown to be an important process for the redistribution of immobile radionuclides in soils (Müller-Lemans & van Dorp, 1996). Bioturbation is of special interest for radionuclides with high K_d-values implying a strong accumulation in soils. The rate of soil transported by earthworms used in the calculations is taken from Müller-Lemans & van Dorp (1996) and will of course vary due to soil properties and the number of earthworms. This variation does, however, not appear in the uncertainty analyses (see section 5.4), as it is overwhelmed by the large variation in K_d-values. If neglecting bioturbation the EDFs for actinides should increase, due to the high contribution to EDF from inhalation of contaminated soil particles.

Root uptake of radionuclides is not considered as a transport process in the modules. Results from a screening calculation and from Coughtrey et al (1983) indicate that root uptake factors must be higher than 10 to cause a significant reduction of the radionuclide content in soil. There are some radionuclides with such high root uptake factors for which this process should be relevant to consider. Those are Cl-36 and Tc-99, see Tables A-1 – A-4. For Tc-99, laboratory studies have shown root uptake factors up to 400, however declining with time (Echevarria et al 1995). Such high values are not used mainly due to the long time periods considered in the studies. Probably the EDFs for Cl-93 and Tc-99 are conservatively biased as no reduction to uptake in biota is considered.

Erosion
There is a loss of elements due to erosion or removal of soil during agricultural practices. Geographic conditions, soil/sediment types and meteorological parameters such as wind and rain influence the magnitude of erosion. The size and density of the eroded particles are also of significance. Vegetation covered land or sediments are less exposed to erosion since the roots bind the material and the plants may cover parts of the surfaces. It was, however, not possible to find any pertinent literature data on erosion. A recent literature survey showed ranges of several orders of magnitude for erosion of agricultural fields in southern Sweden (Lustig, 1996). If the contaminated soil in the modules had high erosion rates, lower EDFs could be expected.

5.2.6 Biological uptake

Uptake of radionuclides in vegetation plays an important role when terrestrial pathways dominate the exposure, especially for radionuclides with high bioavailability such as

Cl-36, Se-79, Tc-99, I-129, and Cs-137 (see Chapter 4). The concept of a constant relation between concentration in soil and vegetation during a continuous addition of radionuclides can be expected to be a conservative assumption. After the Chernobyl accident, the uptake of Cs-137 in agricultural crops has shown a high decrease with time due to fixation of Cs-137 within minerals (IAEA, 1996).

Translocation
Crops were also considered to be contaminated by initial retention of radionuclides from irrigation. A fraction of the radionuclides from irrigation water was then assumed to be transferred at each irrigation occasion to edible parts of cereals and root crops by translocation, see Appendix B. This process was intensely studied for Cs-137 and Sr-90 after the Chernobyl fall-out. The concentrations in cereals of Cs-137 and Sr-90 due to translocation are strongly related to the time between contamination and harvest (Aarkrog, 1994). A short time between contamination and harvest results in higher concentrations of radionuclides. The simplified approach used in these calculations did not consider this aspect, probably resulting in overestimated values on contamination due to translocation. Furthermore, it was difficult to find data for most nuclides on translocation. Data were therefore used with large ranges of variations.

Aquatic biota
The uptake of radionuclides in aquatic vegetation in the surface water ecosystems was not considered earlier. This pathway contributes much to the value of EDFs for radionuclides with efficient transfer to milk and meat due to cattle's consumption of aquatic plants during part of the year. The whole amount of radionuclides in water passing through the vegetation is conservatively assumed to be taken up and thereby no element specific behaviour is considered, see Appendix B. This simplification causes an overestimate of radionuclide accumulation in aquatic plants. It is, however, not possible to be more precise due to lack of data for uptake of elements in such plants. This approach was therefore chosen in order not to underestimate the transfer to aquatic plants.

The uptake and accumulation of radionuclides in fish, crustacean, invertebrates and algae are described by element-specific bioaccumulation factors. These factors express the ratio between the concentration of the element of interest within the organism and the concentration in water. This is a simplification of the assimilation of radionuclides for some aquatic species, especially for organisms absorbing radionuclides both directly from water and from food in the gastrointestinal tract. The use of bioaccumulation factors is, however, practical and common in radioecological models (Davis et al, 1993). A disadvantage is the wide ranges of values found for the element. However, there are several reasons to the wide ranges for bioaccumulation factors. One source of variability is the interpretation of measured radioactivity levels in fish and water. Levels in fish are results of integration over long time periods while measurements of radionuclide concentrations in water give momentary values. Other reasons for wide ranges may be that:

- Reported values can be either from measurements of levels in whole fish or in fish muscle
- Water samples may be filtered or not before measurements
- The environmental conditions were not specified.

It is shown from results for cesium that environmental conditions may influence the uptake in aquatic biota. It is well known since the 1960's that the uptake of cesium is negatively correlated to the potassium concentration in water (Kohlemainen & Miettinen, 1968). Cesium also accumulates along the food web, ending up with the highest bioaccumulation factors for predatory fish (Rowan & Rasmussen, 1994). The corresponding information is usually lacking for other radionuclides. Poston & Klopfer (1986) report a dependency between uptake and trophic level of the organism for some elements, such as americium. This is considered in the selection of data, see Tables A-12 and A-13 where high values are selected because predatory fish dominate human's fish diet.

Bioaccumulation factors for brackish water are scarce. Most literature deals with fresh or marine waters. Values for fresh-water conditions are used when specific data for brackish water are missing. This is done because bioaccumulation factors for fresh water usually are higher than those for marine waters.

Transfer to milk and meat
Contaminated vegetation, water and soil are used by cattle and thus radionuclides are transferred to milk and meat. Cow's inhalation is not considered as a pathway for intake of radionuclides. The experiences from the Chernobyl fallout showed that although Cs-137 was transferred by air, the main pathways for cows intake of Cs-137 were via water and food, while inhalation gave a minor contribution (BIOMOVS, 1990b). This could be expected to be true for the other elements as well, and inhalation is therefore excluded as a pathway for cows in this assessment.

The excretion of radionuclides by cattle is not considered, because the irrigation of pasture fields is always assumed to take place from the same source. The quantities of water consumed by cattle are small compared to the irrigation amounts.

5.2.7 Exposure situations

The selection of exposure pathways was based on earlier experience of dominant exposure pathways for the radionuclides of concern. This leads to a variety of internal exposure pathways while the only external exposure pathway considered is exposure from contaminated ground.

Crops are classified into the three main groups, vegetables, cereals and root crops. Consumption of fruits and berries is not considered, mostly due to lack of data for all elements and it is assumed to have small contribution to EDFs.

The exposure pathways considered are in good agreement with current Swedish consumption patterns, with the exception of algae, which in the coast module is assumed to be 2 kg annually. This is a conservative assumption. The reported values for uptake of elements in algae is in general much higher than those for fish, see Tables A-13 – A-14. This leads to higher exposures from consumption of algae compared to exposures from fish from the same water. Consumption of algae is not used in dose assessments for normal operation of Swedish nuclear power plants (Bergström & Nordlinder, 1992a). However, those assessments considered practices of today, while

this report considers future populations, which may need algae as a source of protein. Moreover, the high dilution in the coast module will in any case give low EDFs.

The calculated EDFs are proportional to the consumption, except for when inhalation or external irradiation dominates the exposure. Consumption is therefore an important feature for definition of the most exposed individual. Average consumption values may be obtained from statistics but are they representative when the most exposed individual should be considered? These individuals could be physically active and therefore have a large total consumption. It is furthermore difficult to judge the actual amounts of foodstuff consumed by man from sales statistics (Becker et al, 1985) due to losses during preparation etc. As the uptake of elements in foodchains varies, conservative exposures due to high consumption of one foodstuff may lead to non-conservative values for another. Increasing the quantities of all types of foodstuff may on the other hand lead to gross overestimates of the total consumption.

Consumption of fish is probably the most sensitive factor for the EDF values in aquatic ecosystems. A three times higher consumption of fish would lead to the corresponding increases of EDFs for nuclides with high uptake in fish e g Cs-isotopes. Most EDFs for a coastal area should increase since consumption of fish is a major exposure pathway in the coastal ecosystem. On the other hand, the EDFs for a coastal area are the lowest due to high dispersion volumes.

One possibility in future model assessments is to use the intake of essential substances of food like proteins, carbohydrates etc. This should be a better way for avoiding unrealistic compositions and variability in results due to consumption values. Another possibility in future studies is to see how the EDFs are influenced by one-sided consumption habits (e.g. vegetarian food, or very large shares of meat or fish) although with normal total food quantities consumed.

Meat is in the calculations assumed to consist of beef. According to a Finnish investigation of actual foodstuff consumption, the consumption of pork was about 20 % higher than that of beef (IAEA, 1996). The levels of Cs-137 due to the Chernobyl fallout showed on the other hand about five times higher levels in beef than in pork (IAEA, 1996). This is because cows by their grazing integrate nuclides over much larger areas than pigs do. They also consume more water and locally produced food. Thus considering beef as the only type of meat does not underestimate the exposure.

Inhalation of eroded soil particles is only considered while inhalation of radionuclides from sea spray is neglected. Data for an open sea showed that exposure via sea spray give a minor contribution to the total EDF for the actinides and none for other nuclides (Bergström & Nordlinder, 1991c). Data for dust content in air, from agricultural practices, are taken from Haak (1983). The data have high uncertainties especially concerning exposure times, which were conservatively selected.

5.3 Uncertainties in the computational model

Verification and validation are of major importance concerning the confidence in model results. Verification applies to the accuracy in the numerical methods used for the

programs. The codes need to be verified before entering the step of validation when the model results are compared and evaluated against independent data. Verification is a necessity to avoid programming mistakes.

The solution methods applied in BIOPATH have been subject to several verifications. Already in the end of the 70's comparisons were done with the Lindif-method (Forssén, 1977) against an alternative solution method, Kutta-Merson and an analytical solution (Persson & Nilsson 1978). The results showed, for a theoretical case, that discrepancies of about 2 % could arise if the ranges of rate constants for a compartment were about ten orders of magnitude. When comparisons were made with a specific case, 17 compartments, only considerable deviation was identified for the compartments where the ranges of in- and output were the highest. So was, however, not the case in the calculations of EDFs. Probably minor time steps used in Lindif should have reduced this discrepancy. It is recognised the Lindif method can be sensitive to the lengths of time steps if there are strong gradients in the source terms or the equations are very stiff.

Recently, both BIOPATH and PRISM were parts of one total model intercomparison where all models used the same expressions and parameter values for all rate constants and for calculating the EDFs to man from a variety of exposure pathways (Klos et al, 1993). The test included one case with a single nuclide and another case with a three-nuclide chain. Deterministic as well as probabilistic answers were requested. Three of the seven models used the same solutions methods as those in BIOPATH. For the deterministic calculations in this test results were applied for the two-solution methods, see above. When comparing these results with those not using BIOPATH (one analytical), the results showed good agreement. Minor deviations from the other results could occur, initially or at the end, for decay products, both of which the concentrations of the radionuclides were of no importance from the EDF point of view. This was either due to the treatment of source terms, which we, in contrast to the other participants, handled as one compartment or to the initial time steps chosen.

Concerning the probabilistic calculations, four models used strict Monte Carlo sampling (MC) and the rest used Latin Hypercube Sampling (LHS). The results were in good agreement and ours were in total agreement to four of the other. Results from two of the codes showed anomalies of about 10 % compared to the others. However, it was not possible to explain this within the scope of the study. This study was very valuable, especially for the identification of coding errors in new developed models.

These verification tests show that uncertainties due to computational errors are of much lower importance than uncertainties due to conceptual modelling and parameter values.

5.4 Results from uncertainty analyses

In contrast to uncertainties in results due to conceptual modelling, the variations due to selection of parameter are easier to quantify by use of error propagation methods. Such an approach is used in the calculations, i.e. all values of parameters, except for the dose conversion factors, are randomly generated from predescribed distributions.

The radionuclides C-14, Cl-36, Ni-59, Mo-93, I-129, Cs-135 and Np-237 were selected for an identification of parameters contributing most to the uncertainties to the calculated EDFs. These nuclides are released to the biosphere in relatively high quantities, and are also transferred from the repository to the biosphere quite rapidly.

The parameters are classified into three main categories: biological, chemical-physical and human related parameters. The first category involves all parameters attributed to biological processes and parameters such as root-uptake (BFX), uptake in fish (BFF), crustacean and aquatic plants, translocation within vegetation, cattles' consumption, metabolism of an element in cattle, yields and water retention on surfaces of vegetation. The second category, chemical-physical parameters describes the physical properties of the area, such as volume, water flow, porosity, size and K_d-factors. The third category includes those parameters related to human behaviour, such as quantity of irrigation water, frequency of irrigation, keeping cattle outdoors and consumption.

The results from the regression analysis show that the correlation coefficients are high enough ($R^2 > 0.8$) to conclude that model parameters are adequate to explain the variations in EDFs (Gardner et al, 1983). The relative contributions to the uncertainties in EDFs for the three categories are shown in charts below for the six modules (Figures 5-1, 5-2, 5-5, 5-6, 5-9, 5-10 and 5-13).

In general, the category of biological processes dominates the uncertainties followed by the category related to physical-chemical properties. However, for the agricultural land and peat ecosystems the physical chemical parameters dominate the variation. The relative ranges for parameters common for the lake and agricultural land modules are also shown in Figures 5-3, 5-4, 5-7, 5-8, 5-11 and 5-12 below for Cl-36, Mo-93 and Cs-135, respectively. The figures also show relative ranges for the total EDF for those nuclides. Below the abscissa of each of these figures, the relative contribution (%) to the uncertainty in the results from each parameter is shown within brackets.

Usually, in studies of this type the main contribution to the overall uncertainty is the wide range in generic data on dispersion (BIOMOVS II, 1996b). In this study site-specific data are used for surface water, which reduce the variation. For agricultural land and peat bog modules, however, the areas and volumes are more difficult to delimit which contributed considerably to the variations, see Figures 5-3, 5-7 and 5-11. The human factors have narrow ranges, due to the selection of living habits for the most exposed individual, and thus give the smallest contributions to the variations.

Carbon-14
For C-14, see Figure 5-1, the main uptake in vegetation is through assimilation and therefore no root uptake was considered. A specific activity model would have been better to use for C-14. Such model is, however, not included in this study due to the high number of various elements to handle. Furthermore, C-14 does not give a large contribution to the potential exposure from a deep rock repository. This is shown in results from previous safety assessments for high-level waste (Bergström, 1983).

The uptake of carbon in fish dominates the calculated uncertainties in the fresh water modules. This uptake is described by a bioaccumulation factor for fresh water having a mean value of 50 000, ranging from 1 000 to 60 000. In an earlier study a value of 5 000

was used (Bergström & Puigdomènech, 1987). A model test performed within the BIOMOVS II study (BIOMOVS II, 1996d), showed that there were large uncertainties in the results for C-14 turnover in a Canadian shield lake. The use of a wide range for the bioaccumulation factor in this study reflects the present difficulties in modelling the C-14 turnover in lakes.

For brackish water, a narrow range is used for the bioaccumulation factor, in contrast to fresh water, which is questionable. The values are based on data from a literature review on values for bioaccumulation factors. This was done for the modelling of uptake in biota from water-borne releases of radionuclides from normal operation of Swedish nuclear plants (Bergström & Nordlinder, 1992b). The bay is also well described regarding volume and water turnover. Depending on the narrow ranges for the parameters above, the consumption rate of fish contributes most to the uncertainty in the results in this case.

The chemical-physical category of parameters shows a large contribution to the variation in the results for the agricultural land and peat bog modules, see Figure 5-1. This can be explained by the difficulties to determine compartment volumes for these modules, and in describing the accumulation of C-14. Wider ranges are therefore used for compartment volumes for the agricultural land and peat bog modules than for the surface water modules.

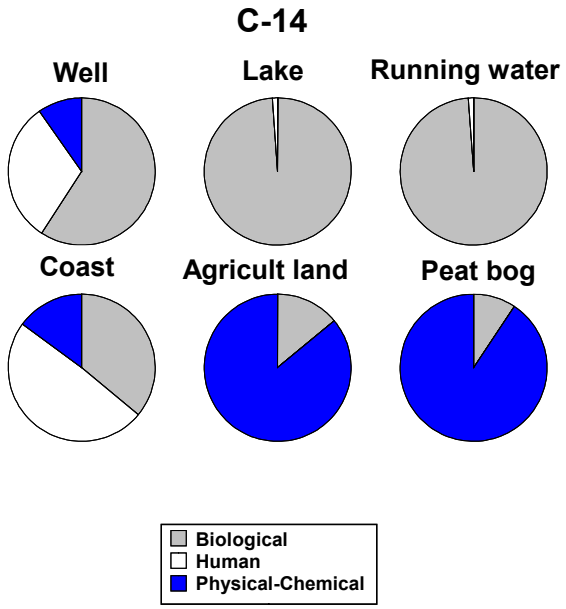

Figure 5-1 Relative contribution from the three categories of parameters to total uncertainty in EDFs for C-14 for the different types of ecosystems.

Chlorine-36

Chlorine is a highly bioavailable and mobile element in the biosphere (Coughthrey et al, 1983). Relatively low K_d-values are therefore used to describe the retardation of the element in soil. However, accumulation in soil contributes to the uncertainty in the results for all modules with a soil compartment as the migration in soil is a common feature when calculating the uptake in vegetation, see Figure 5-2. The ranges used for

uptake in biota and transfer to milk and meat are narrow for chlorine, in contrast to many other elements, because chlorine is efficiently taken up in vegetation. Coughtrey et al (1983) recommend values of 50 for root-uptake in soils with low or neutral pH-values. However, only few studies have been done on Cl-36 specifically, most studies on chlorine are dealing with salt stress to plants.

The main exposure pathways for Cl-36 in the coast module are consumption of milk and meat. Data on uptake in water plant, which are consumed by cattle, is therefore a main contributor to the variation for that case.

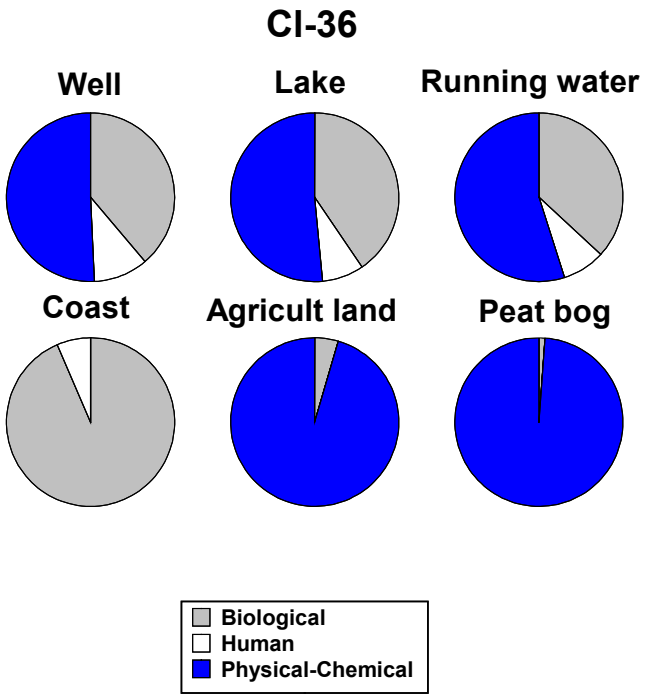

Figure 5-2 Relative contribution from the three categories of parameters to total uncertainty in EDFs for Cl-36 for the different types of ecosystems.

When comparing ranges of EDF for agricultural land and lake modules, respectively, see Figures 5-3 and 5-4, the ranges are the largest for the agricultural land module, larger than any of the parameters contributing to the variation, see Figure 5-3. So is not the case for the lake module, see Figure 5-4.

In similarity to C-14 the chemical-physical category of parameters shows a large contribution to variation in the results for the agricultural land and peat bog modules, see Figures 5-2 and 5-3.

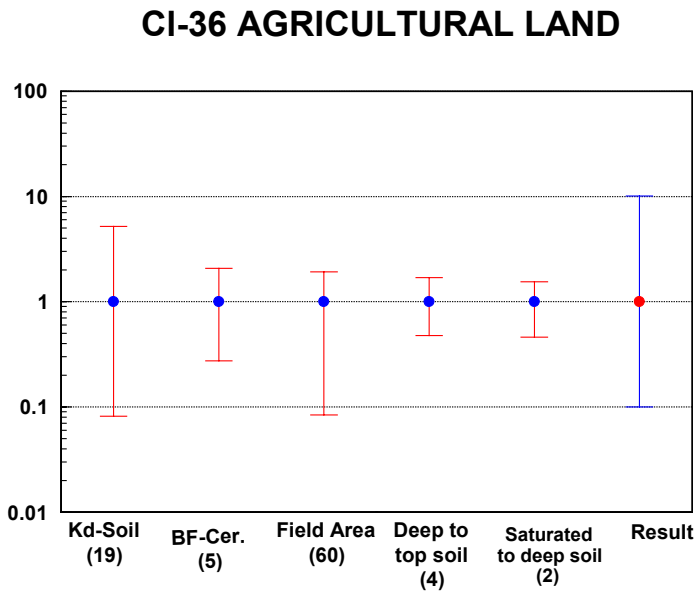

Figure 5-3 *Relative ranges for input parameters and for resulting EDF for Cl-36 in the agricultural land module. Within brackets the percentage contribution of each parameter to total uncertainty is shown. The parameters "deep to top soil" and "saturated to deep soil" represent the water flows. BF-Cer means root-uptake factor for cereals.*

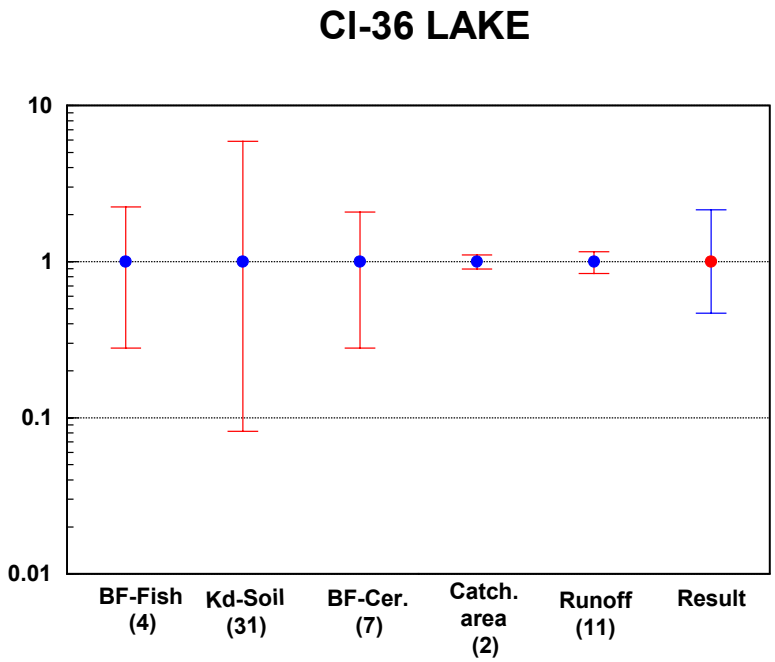

Figure 5-4 *Relative ranges for input parameters and for resulting EDF for Cl-36 in the lake module. Within brackets the percentage contribution of each parameter to total uncertainty is shown. BF-Cer means root-uptake factor for cereals and BF-Fish bioaccumulation factor for fish.*

Nickel-59
The main sources to uncertainties in results for Ni-59 are the parameters describing the uptake in organisms, i.e. biological ones, see Figure 5-5. Root-uptake to root crops dominates the uncertainty in the results for the well module. This is due to higher root-uptake (see Tables A-3. A-4) and higher consumption rates of root crops than of vegetables (see Table B-2), which are the only irrigated crops in the well module. In the lake and running water modules, bioaccumulation factors for fish contribute the most to the uncertainty, followed by the distribution factor for milk. Nickel is efficiently transferred to milk, see Table A-10 in Appendix A.

The physical-chemical category gives the highest contribution to uncertainties in the agricultural land and peat bog modules in similarity to the results for other radionuclides. The uncertainty in EDF for a peat bog is dominated by the variation in size of the contaminated area as this volume is directly inversally proportional to the EDFs. The area size is also the second largest source to uncertainty in the agricultural land module, for which the K_d-value is the most important, see Table A-6 and Figure 5-5.

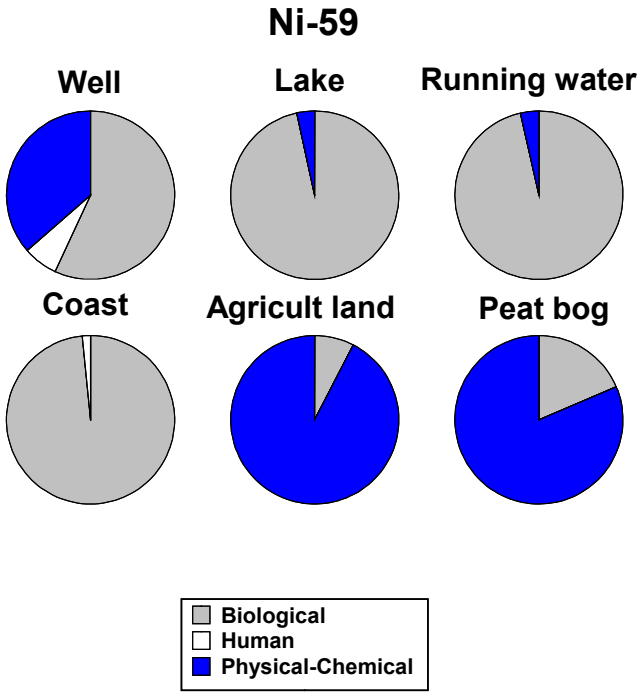

Figure 5-5 Relative contribution from the three categories of parameters to total uncertainty in EDFs for Ni-59 for the different types of ecosystems.

Molybdenum-93
The K_d-factor is the parameter dominating the uncertainty in EDF for Mo-93 in the results for the well module (see Figure 5-6). This is explained by the high contribution to exposure from consumption of vegetables as all root-uptake factors are negatively correlated to the K_d-values.

The transfer of Mo-93 to biota dominates totally the variation in the coast module. The bioaccumulation factor is the most important one, followed by the transfer factors for milk and meat. This is due to the exposure from consumption of milk and meat, contaminated by cows grazing water plants. This causes a main contribution to the total EDF for Mo-93 from these pathways for the coast module, in similarity to other bioavailable radionuclides, see Table 4-12.

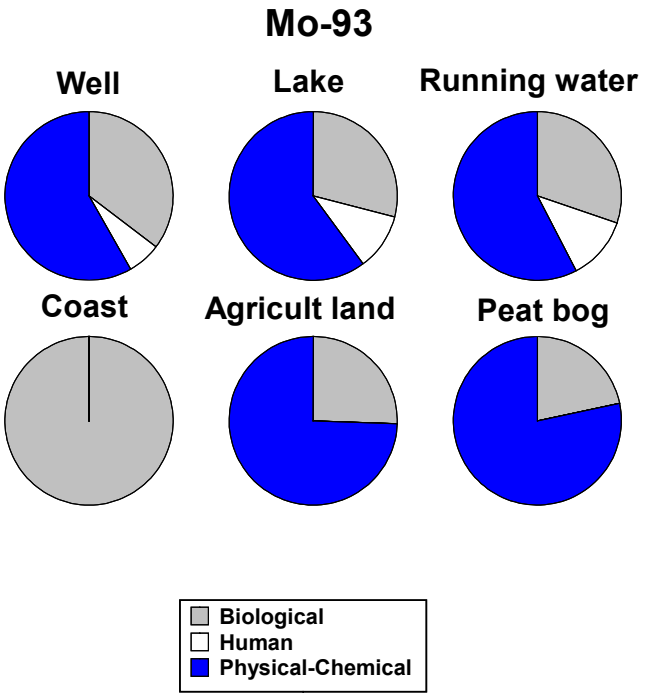

Figure 5-6 Relative contribution from the three categories of parameters to total uncertainty in EDFs for Mo-93 for the different types of ecosystems.

The size of the areas gives the major contribution to uncertainties in the agricultural land and peat bog modules, see Figures 5-6 and 5-7. The area for agricultural land is set to vary within a factor of ten, while the K_d-values ranged over two orders of magnitudes. The retention in soil is, however, not linearly proportional to the K_d-value explaining its minor contribution to the uncertainty. The same conditions are also valid for the peat bog module.

In similarity to other radionuclides, ranges of variation are larger for the agricultural land module than for the lake moduel, see Figures 5-7 and 5-8.

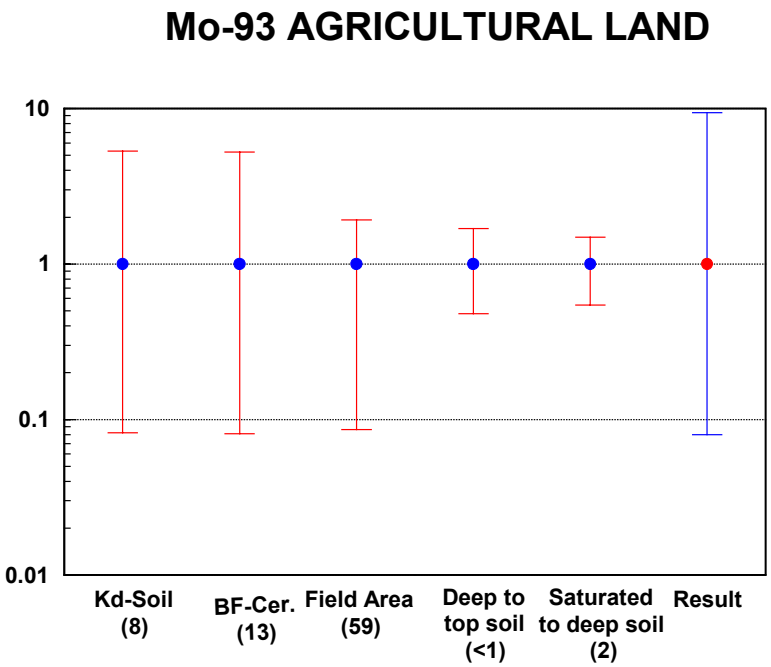

Figure 5-7 *Relative ranges for input parameters and for resulting EDF for Mo-93 in the agricultural land module. Within brackets the percentage contribution of each parameter to total uncertainty is shown. The parameters "deep to top soil" and "saturated to deep soil" represent the water flows. BF-Cer means root-uptake factor for cereals.*

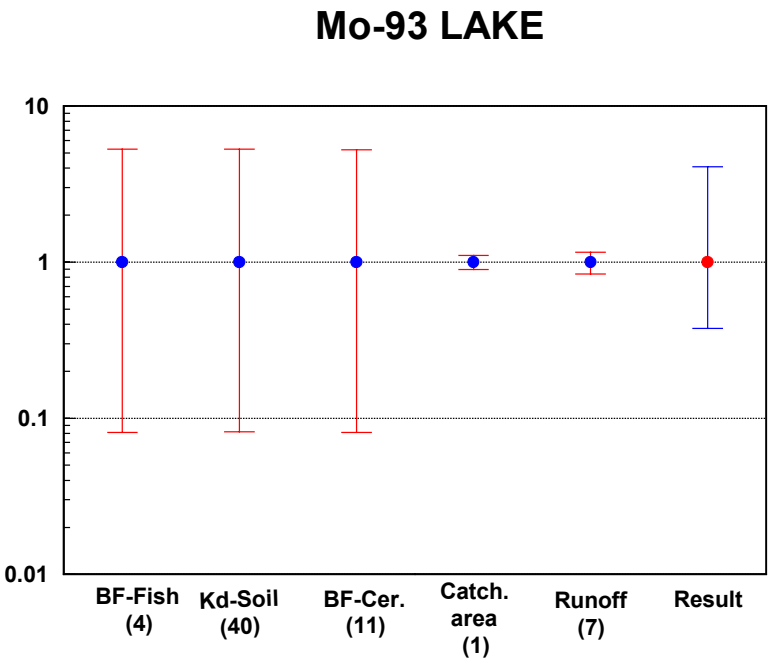

Figure 5-8 *Relative ranges for input parameters and for resulting EDF for Mo-93 in the lake module. Within brackets the percentage contribution of each parameter to total uncertainty is shown. BF-Cer means root-uptake factor for cereals and BF-Fish bioaccumulation factor for fish.*

Iodine-129
The root uptake factor for root crops dominates the uncertainty in the results for the well module completely for I-129, see Figure 5-9. Iodine has a high transfer rate to milk, but the pathway for cows' intake of radionuclides in the well module is only due to consumption of water, which result in a minor contribution from the milk exposure pathway, see Tab1e 4-3.

The contributions to variation are quite similar for the running waters and lake modules, and are mostly due to bioaccumulation factor for fish, root uptake factor for cereals and the transfer to milk.

For the coast module the uptake of I-129 to aquatic plants and further transfer to milk and meat dominates the uncertainties. The exposure and thereby the contribution to the uncertainty from consumption of fish is lower than for the milk and meat exposure pathways, see Table 4-12.

Also for I-129, the size of the peat bog is a major contributor to the uncertainty in the peat bog module. Iodine is quite mobile, in similarity to the other radionuclides discussed above. For the agricultural land module, however, the root uptake factor to cereals dominate the uncertainty over the size of the area. Consumption of cereals was also one of the dominating exposure pathways for the EDF.

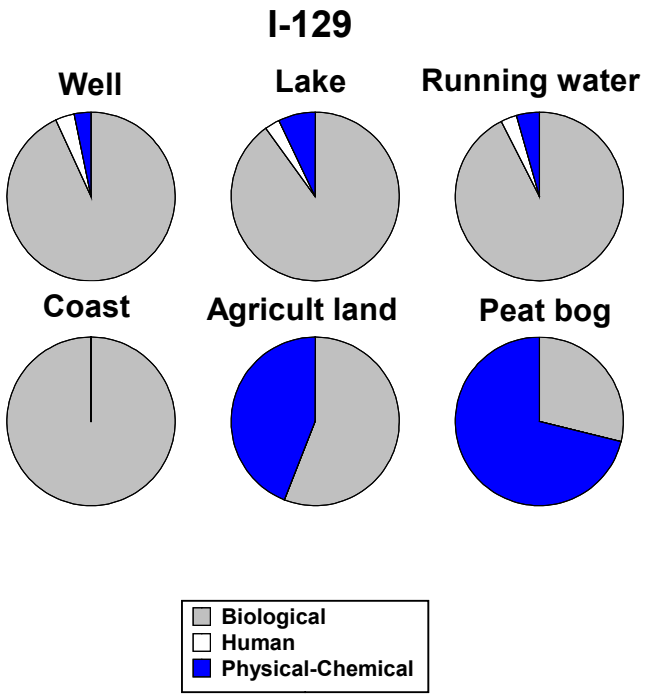

Figure 5-9 Relative contribution from the three categories of parameters to total uncertainty in EDFs for I-129 for the different types of ecosystems.

Cesium-135
The root uptake to root-crops is the dominating contributor to uncertainty in the results for the well module for Cs-135, see Figure 5-10.

Consumption of fish is the main exposure pathway for running water as well as for the lake module. It is, hence, the bioaccumulation factor which dominates the uncertainty, although it has narrower ranges than many other parameters, see Figure 5-12. Cs-137 is one of the most studied radionuclides and special attention has been paid to its uptake in fresh water fish. The results is that values for the bioaccumulation factors may be determined with higher accuracy than for other elements

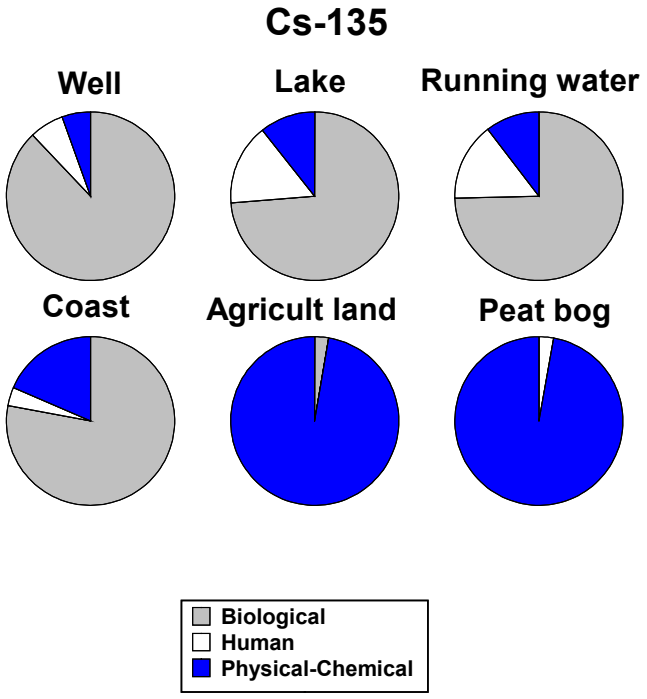

Figure 5-10 Relative contribution from the three categories of parameters to total uncertainty in EDFs for Cs-135 for the different types of ecosystems.

Also for Cs-135, K_ds and the sizes of areas are important for the uncertainty in the results from the agricultural land and peat bog modules. K_d-values contribute to the uncertainty with about 50 %, see Figure 5-11.

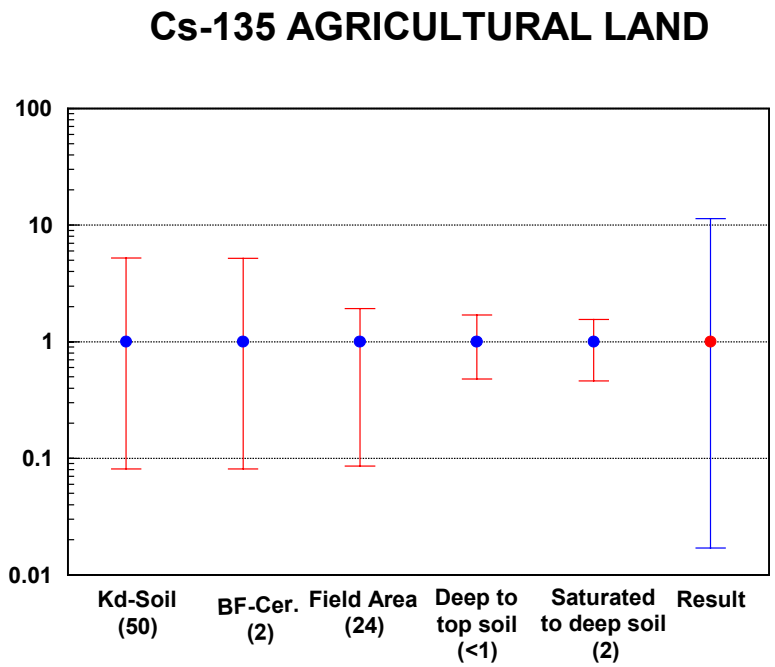

Figure 5-11 Relative ranges for input parameters and for resulting EDF for Cs-135 in the agricultural land module. Within brackets the percentage contribution of each parameter to total uncertainty is shown. The parameters "deep to top soil" and "saturated to deep soil" represent the water flows. BF-Cer means root-uptake factor for cereals.

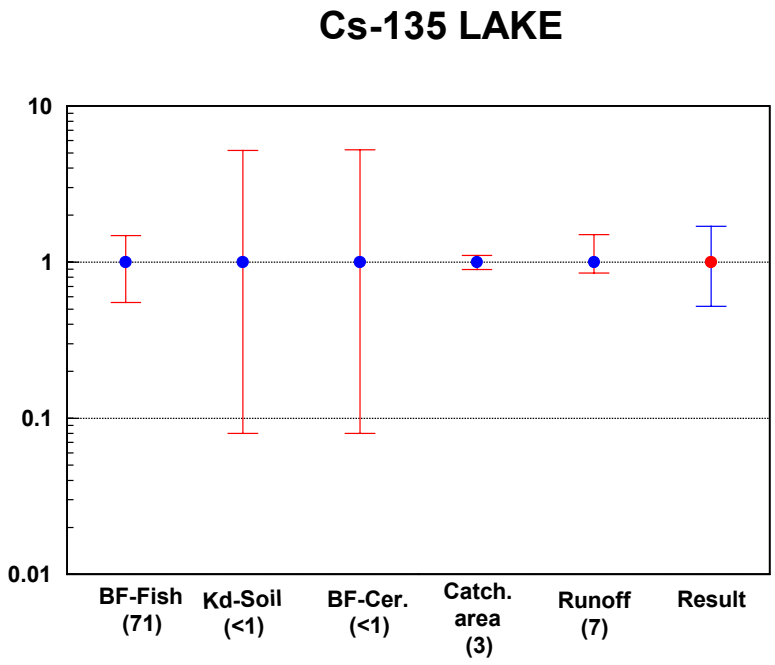

Figure 5-12 Relative ranges for input parameters and for resulting EDF for Cs-135 in the lake module. Within brackets the percentage contribution of each parameter to total uncertainty is shown. BF-Cer means root-uptake factor for cereals and BF-Fish bioaccumulation factor for fish.

Neptunium-237
In contrast to the other radionuclides, the physical-chemical parameters dominate the uncertainty in the results for Np-237 for the well module, see Figure 5-13. The major source to the uncertainty is the K_d-values for soil, followed by the dust concentration in air. Inhalation of resuspended particles from soil was the second dominant exposure pathway after consumption of water, see Table 4-3. The latter is attributed to small ranges of uncertainties due to narrow ranges of mixing volumes and consumption values. Exposure from inhalation is on the other hand attributed to large uncertainties due to calculation of concentration of Np-237 in soil and levels of suspended matter in air. Another consequence of the dominance of these pathways to the total EDF is that human factors such as irrigation frequency and exposure times give a significant contribution to the uncertainty.

Consumption of fish is the dominant exposure pathway for the surface water ecosystems. This is because the skewed distribution used for the bioaccumulation factor, see Table A-12, leads to some high values. The skewed distribution reflects the high variation coupled to this value. This consequently overwhelms any contribution to the variation from the transfer of Np-237 from water to sediments, as the bioaccumulation to fish dominates totally.

Inhalation of dust is the dominant exposure pathways for the agricultural land and peat bog modules. Sizes of the areas are therefore important according to the uncertainty analysis, especially for the peat bog module but also for agricultural land. For the latter, however, the K_d-values are also of importance.

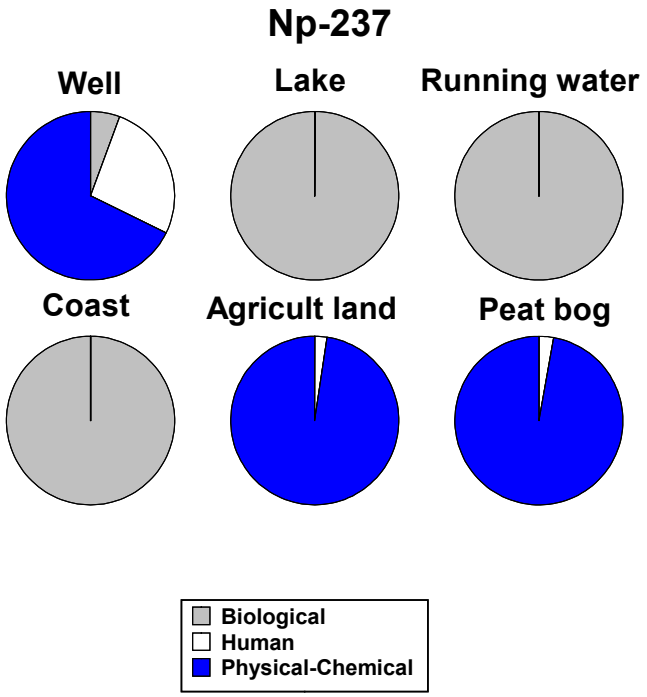

Figure 5-13 Relative contribution from the three categories of parameters to total uncertainty in EDFs for Np-237 for the different types of ecosystems.

6 Conclusions

The realisation of a deep repository for nuclear waste is a step by step process where information is successively built-up and detailed for decisions that will carry the process forward. Safety analyses constitute thereby important parts of the information on which the decisions are based. The need for a detailed basis for decisions as well as the possibilities to perform detailed safety analyses depend on the stage of work at hand. In an early stage when for example the principles for the function of the repository are decided, the safety analyses are generic and simplified (KBS-3, 1983 and SKB-91, 1992). The turnover of radionuclides in the biosphere and the EDFs to man may then be calculated by the use of standard or reference biospheres. When the design and localisation of the repository are finalized, the basis for the studies will be more detailed and the safety analyses may be more site specific. The use of modules may be seen as a first step towards applying site specific data to biosphere model calculations. They may also be used as tools for a sensitivity analysis to determine the need of precision in measured data at a site of potential interest.

Modules may be one useful tool to evaluate discharge areas and for screening between potential sites for location of a reppository. The preliminary calculations show for example that it is not necessarily a well that gives the largest contribution to the total EDF. It may instead be areas with high potential accumulation of radionuclides as peat bog areas and cultivated land areas. In this study the calculations show that the highest EDFs occur in the peat module followed by the well and agricultural land modules. The lowest EDFs are found in the coast module whereas the factors are somewhat higher for the lake and running water modules. Therefor with the same leakage rate of radionuclides the highest dose could be expected if the nuclides reach the biosphere within a peat bog area then if the groundwater discharge is to a coast area.

The variation analyses performed showed that biological parameters contribute most to the variation in results for the well, lake, coast and running waters modules. Physical-chemical parameters dominated, on the other hand, the variation in results for the agricultural land and peat bog modules. The EDFS for these latter ecosystems show also the largest uncertainty, because of incomplete knowledge about the behaviour of radionuclides in such ecosystems.They have not been considered in earlier safety assessments. Emphasis should be put on a deper understanding of relevant processes for the transport and accumulation in soils and peat bogs.

There is a need for a further development of the module approach to biosphere safety assessments. The present models do not consider an evolution with time within each module and they are not linked between one another. A module for forest ecosystems, which is a common Swedish ecosystem is required. Interactive studies of specific sites for model calculations can give input on how a real biosphere may be divided into the modules specified modules and on how site specific data may be obtained and applied.

The calculations are based on a continuous release of 1 Bq/year during 10 000 years for each radionuclide considered. The results may therefore be used to present EDFs to man at the time of 10 000 years from a release of radionuclides from a repository. If the results are used to calculated EDFs at earlier times e g 1 000 years it may therefore lead to an overestimate of the EDFs for most nuclides. This is due to a build-up of the concentrations of some nuclides in the biosphere during long times and that steady-state conditions are not prevailing during the times mentioned above for those nuclides. For time-varying releases the use of EDFs must be performed with cautions for the same reason as above.

7 Acknowledgement

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Parameter Values

A.1 Introduction

Element specific data used in the modules are presented in this Appendix. A literature review was performed concerning values of transfer factors to vegetation.

Other element specific data are listed with references without any further comments. Those are distribution coefficients for elements between solid and dissolved fractions in various types of soil, distribution coefficients for elements between solution and suspended particles in lakes and the Baltic Sea and transfer factors to milk and meat, respectively.

All data are given as mean values. Maximum and minimum of ranges and type of distribution used in the calculations are given in each table.

A.2 Transfer factors to vegetation

A.2.1 Literature review

The uptake of radionuclides from soil through the roots of vegetation play an important role for the radionuclide content in the plant. This process is described by a root uptake factor (transfer factor, TF) which shows the relation between radionuclide concentration in crop and the soil in which those are grown. The factors are expressed as Bq/kg dry or wet weight crop per Bq/kg dry weight soil. A literature review was performed for updating and reviewing the values of radionuclide transfer factors used in this study. The review focused on the root uptake to pasturage, vegetables, root crops and cereals, four main groups of crop. The studied elements were strontium, technetium, and cesium. These were the only radionuclides for which new and relevant information was found.

The International Union of Radioecologist (IUR) have continuously published recommended soil-to-plant transfer factors since 1982 when the working group started. A great deal of studies performed lately concerning root uptake and transfer factors have been done in Asia (i.e. Japan and China). The studies are mainly laboratory studied or pot studies but in some cases field studies were performed.

The radionuclides that reach the root systems can either have been deposited directly on the ground or been supplied through contaminated irrigation water. In both cases, the plants can also be contaminated directly through deposition on the plant itself. Radionuclides migrating from deeper soil layers can also reach the root zone.

The values of the TFs are dependent on the vegetation type but a more important factor seems to be the soil type. Soils differ in pH as well as the content of organic matter, parameters which have an impact on the migration and availability of radionuclides in soil. Furthermore, the time that has elapsed since the contamination occurred also affects the root uptake rate, because the bioavailability of some radionuclides decreases with time.

Soil properties directly influencing the soil exchange capacity and exchangeable cations was found to influence the values of TFs (Roca & Vallejo, 1995, Mascanzoni, 1988, Skarlou et al, 1994).

The ”treatment” of the soil (ploughing, irrigation, fertilisation) also strongly affects the transfer factor (IAEA, 1994). According to Lönsjö et al (1994) a reduction of the TFs of Cs by an order of 10 or more was shown after ploughing grasslands in Sweden and Russia contaminated by the Chernobyl fallout. Irrigation may enhance the uptake of radionuclides and is therefore a parameter that has to be taken into account when studying the root-uptake. Potassium fertilisation of contaminated grassland in Sweden and Russia (Chernobyl fallout areas) has shown to reduce the transfer factor by a factor of 20-50 % compared to unfertilised plots (Lönsjö et al, 1994).

Strontium

Chemically, the behaviour of strontium is similar to that of Ca (Roca & Vallejo, 1995). Strontium taken up by plant roots will be translocated to the leaves where it accumulates. Therefore, strontium concentration in the leaves increase with age (Ban-Nai et al, 1995).

Yasuda & Uchida (1994) performed a statistical study of TF values of Sr and Cs for the leafy plants cabbage, spinach and grass. The values showed log-normal distributions and the geometrical means was significantly different between the different plants.

Technetium

Low pH values in soils gives higher technetium sorption and hence higher soil content. Acid soils therefore shows relatively higher technetium content than other soils types.

In a study by Masson et al (1989), the value of transfer factor in radishes was reached after only two days and then remained constant during the lifetime of the plant. The presence of other radionuclides (uranium, plutonium and americium) enhanced the technetium uptake. The root uptake decreased with a factor 2-4 after 2 years compared with 2 months after contamination (Masson et al, 1989). That the uptake decreases with time is confirmed by i.e. IAEA (1994). The availability of technetium in soils are reduced with time due to microbial immobilisation (Echevarria et al, 1995). The study also showed that the availability is high immediately after contamination but decreases very quickly (within a few weeks). The uptake showed an exponential decrease which makes it possible to predict the accumulation of technetium in the plants.

As for strontium, the maximum accumulation of technetium in plants are generally found in the leaves (Echevarria et al, 1995, Yanagisawa, 1994). The transfer factors for leaf vegetables were considerably higher than those for non-leaf vegetables (Yanagisawa, 1994).

A stimulation in growth of rye grass at low (2.5 kBq/ kg) technetium concentrations has been showed according to (Echevarria et al, 1995). The study also showed that a decrease in uptake and accumulation of technetium was observed with increasing phosphate fertilisation, which is in accordance with other results concerning that fertilisation decreases the root uptake.

Cesium

Chemically, the behaviour of cesium is similar to that of potassium but a study by Roca & Vallejo (1995) concluded that the plant selectivity for potassium always showed to be higher than that for cesium-134. The content of clay and organic matter in the soil are important parameters affecting the TF values of cesium. Organic rich soils show a high and constant root uptake (Eriksson & Rosén, 1989). Furthermore, according to Eriksson & Rosén (1989) the transfer by root uptake to grain crops was lower than to hay crops on ploughed land (in a Chernobyl fallout region). Potassium fertilisers efficiently reduces the transfer of Cs from soil to grass and barley crops (Rosén, 1989).

According to (Ban-Nai et al, 1995) the transfer factors of Cs for different organs of leaf vegetables shows to be rather homogenous which is roughly in agreement with Skarlou et al (1994).

Am

According to Li-Yunlong (1994) the transfer factors of Am-241 from soil to plant in 12 species of vegetables and corn by root uptake in the area around China Institute of Atomic Energy (the average concentration of Am-241 in this area is 43+/- 14 mBq/ kg dry weight soil due to the world-wide fallout) ranges between 6.2E-04 and 5.1E-03.

A.2.2 Data used for this study

In the tables below the values used in this study are given.

Table A-1 Element specific root uptake factors (transfer 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**	**Low**	**High**	**Reference**
H	5E+1	LT	2E+1	8E+1	*
Be	1E-2	LT	1E-3	1E-1	Davis et al, 1993
C	-	-	-	-	**
Cl	3E+1	T	1E+1	1E+2	Coughtrey et al, 1985
Co	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder 1991b
Ni	2E-1	LT	2E-2	2E+0	IAEA, 1994
Se	2E+1	LT	1E+0	3E+1	Coughtrey et al, 1985
Sr	1E+0	LT	4E-1	3E+0	IAEA, 1994
Zr	1E-3	LT	1E-4	1E-2	IAEA, 1994
Nb	5E-3	LT	5E-4	5E-2	Davis et al, 1993
Mo	8E-1	LT	8E-2	8E+0	IAEA, 1994
Tc	8E+0	LT	8E-1	8E+1	IAEA, 1994
Pd	2E-1	LT	2E-2	2E+0	***
Ag	5E-1	LT	5E-2	4E+0	Bergström & Nordlinder 1991b
Sn	1E-1	LT	1E-2	2E+0	Bergström & Nordlinder 1991b
I	6E-1	LT	6E-2	6E+0	Bergström & Nordlinder 1991b
Cs	2E-1	LT	2E-2	2E+0	IAEA, 1994
Sm	1E-2	LT	1E-3	1E-1	Aggeryd & Bergström, 1990
Ho	1E-3	LT	1E-4	1E-2	Aggeryd & Bergström, 1990
Pb	1E-2	LT	1E-3	1E-1	IAEA, 1994
Ra	8E-2	LT	8E-3	8E-1	IAEA, 1994
Ac	5E-4	LT	3E-5	7E-3	Bergström & Nordlinder, 1990a
Th	1E-2	LT	1E-3	1E-1	IAEA, 1994
Pa	3E-3	LT	3E-4	3E-2	Bergström & Nordlinder, 1990a
U	2E-2	LT	2E-3	2E-1	IAEA, 1994
Np	7E-2	LT	7E-3	7E-1	IAEA, 1994
Pu	4E-4	LT	5E-5	7E-1	IAEA, 1994
Am	1E-3	LT	5E-4	2E-1	IAEA, 1994
Cm	1E-3	LT	1E-4	4E-3	IAEA, 1994

LT = Logtriangular distribution.
T = Triangular distribution.

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

** No root uptake considered.

*** Due to lack of data the same values as for nickel were used, as recommended in Bergström et al (1986).

Table A-2 Element specific root uptake factors (transfer factors) for cereals ([Bq/kg w.w. cereals]/[Bq/kg d.w. soil]).

Cereals (grain/dry soil)					
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	5E+1	LT	2E+1	8E+1	*
Be	3E-3	LT	3E-4	3E-2	Davis et al, 1993
C	-	-	-	-	**
Cl	3E+1	T	9E+0	9E+1	Coughtrey et al, 1985
Co	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder 1991b
Ni	3E-2	LT	3E-3	3E-1	IAEA, 1994
Se	2E+1	LT	9E-1	3E+1	Coughtrey et al, 1985
Sr	2E-1	LT	2E-2	1E+0	IAEA, 1994
Zr	9E-4	LT	9E-5	9E-3	IAEA, 1994
Nb	4E-3	LT	4E-4	4E-2	Davis et al, 1993
Mo	7E-1	LT	7E-2	7E+0	IAEA, 1994
Tc	6E-1	LT	6E-2	3E+0	IAEA, 1994
Pd	3E-2	LT	3E-3	3E-1	***
Ag	4E-1	LT	4E-2	3E+0	Bergström & Nordlinder, 1991b
Sn	4E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1991b
I	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1991b
Cs	2E-2	LT	2E-3	2E-1	IAEA, 1994
Sm	1E-4	LT	1E-5	1E-3	Aggeryd & Bergström, 1990
Ho	1E-4	LT	1E-5	1E-3	Aggeryd & Bergström, 1990
Pb	4E-3	LT	4E-4	4E-2	IAEA, 1994
Ra	7E-2	LT	7E-3	7E-1	IAEA, 1994
Ac	4E-4	LT	1E-5	1E-3	Bergström & Nordlinder, 1990a
Th	1E-2	LT	1E-3	1E-1	IAEA, 1994
Pa	3E-3	LT	3E-4	3E-2	Bergström & Nordlinder, 1990a
U	1E-3	LT	1E-4	1E-2	IAEA, 1994
Np	2E-3	LT	2E-4	2E-2	IAEA, 1994
Pu	7E-6	LT	7E-7	7E-5	IAEA, 1994
Am	2E-5	LT	2E-6	2E-4	IAEA, 1994
Cm	2E-5	LT	1E-6	3E-4	IAEA, 1994

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

** No root uptake considered.

*** Due to lack of data the same values as for nickel were used, as recommended in Bergström et al (1986).

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

	Root-crops (fresh veg/dry soil)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	1E+1	T	5E+0	2E+1	*
Be	3E-3	LT	3E-4	3E-2	Davis et al, 1993
C	-	C	-	-	**
Cl	6E+0	T	2E+0	2E+1	Coughtrey et al, 1985
Co	1E-2	LT	1E-3	1E-1	Bergström & Nordlinder 1991b
Ni	4E-2	LT	4E-3	4E-1	IAEA, 1994
Se	4E+0	LT	2E-1	6E+0	Coughtrey et al, 1985
Sr	6E-2	LT	1E-2	3E-1	IAEA, 1994
Zr	2E-4	LT	2E-5	2E-3	IAEA, 1994
Nb	1E-3	LT	1E-4	1E-2	Davis et al, 1993
Mo	2E-1	LT	2E-2	2E+0	IAEA, 1994
Tc	5E-2	LT	5E-3	5E-1	IAEA, 1994
Pd	4E-2	LT	4E-3	4E-1	***
Ag	2E-1	LT	2E-2	1E+0	Bergström & Nordlinder, 1991b
Sn	5E-2	LT	1E-2	1E+0	Bergström & Nordlinder, 1991b
I	1E-2	LT	1E-3	1E+0	Bergström & Nordlinder, 1991b
Cs	2E-2	LT	2E-3	2E-1	IAEA, 1994
Sm	4E-5	LT	4E-6	4E-4	Aggeryd & Bergström, 1990
Ho	9E-5	LT	9E-6	9E-4	Aggeryd & Bergström, 1990
Pb	4E-3	LT	4E-4	4E-2	IAEA, 1994
Ra	2E-3	LT	2E-4	2E-2	IAEA, 1994
Ac	5E-5	LT	2E-5	1E-2	Bergström & Nordlinder, 1990a
Th	1E-5	LT	1E-6	1E-4	IAEA, 1994
Pa	6E-4	LT	6E-5	6E-3	Bergström & Nordlinder, 1990a
U	3E-3	LT	3E-4	3E-2	IAEA, 1994
Np	2E-3	LT	2E-4	2E-2	IAEA, 1994
Pu	3E-5	LT	3E-6	3E-4	IAEA, 1994
Am	4E-5	LT	4E-6	4E-4	IAEA, 1994
Cm	3E-5	LT	2E-6	5E-4	IAEA, 1994

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

** No root uptake considered.

*** Due to lack of data the same values as for nickel were used, as recommended in Bergström et al (1986).

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

	Vegetables (fresh veg/dry soil)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	1E+1	T	5E+0	2E+1	*
Be	3E-3	LT	3E-4	3E-2	Davis et al, 1993
C	-	C	-	-	**
Cl	3E+0	T	1E+0	1E+1	Coughtrey et al, 1985
Co	1E-2	LT	1E-3	1E-1	Bergström & Nordlinder, 1991b
Ni	2E-2	LT	2E-3	2E-1	IAEA, 1994
Se	2E+0	LT	1E-1	3E+0	Coughtrey et al, 1985
Sr	3E-1	LT	3E-2	3E+0	IAEA, 1994
Zr	1E-4	LT	1E-5	1E-3	IAEA, 1994
Nb	5E-4	LT	5E-5	5E-3	Davis et al, 1993
Mo	8E-2	LT	8E-3	8E-1	IAEA, 1994
Tc	2E+1	LT	1E-1	8E+1	IAEA, 1994
Pd	2E-2	LT	2E-3	2E-1	***
Ag	1E-1	LT	1E-2	8E-1	Bergström & Nordlinder, 1991b
Sn	6E-2	LT	1E-2	1E+0	Bergström & Nordlinder, 1991b
I	3E-2	LT	3E-3	3E-1	Bergström & Nordlinder, 1991b
Cs	2E-2	LT	2E-3	2E-1	IAEA, 1994
Sm	3E-3	LT	3E-4	3E-2	Aggeryd & Bergström, 1990
Ho	3E-3	LT	3E-4	3E-2	Aggeryd & Bergström, 1990
Pb	1E-3	LT	1E-4	1E-2	IAEA, 1994
Ra	5E-2	LT	5E-3	5E-1	IAEA, 1994
Ac	4E-3	LT	2E-4	8E-2	Bergström & Nordlinder, 1990b
Th	2E-4	LT	2E-5	2E-3	IAEA, 1994
Pa	3E-4	LT	3E-5	3E-3	Bergström & Nordlinder, 1990b
U	1E-3	LT	1E-4	1E-2	IAEA, 1994
Np	4E-3	LT	4E-4	4E-2	IAEA, 1994
Pu	2E-5	LT	2E-6	2E-4	IAEA, 1994
Am	7E-5	LT	7E-6	7E-4	IAEA, 1994
Cm	2E-4	LT	2E-5	2E-3	IAEA, 1994

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

** No root uptake considered.

*** Due to lack of data the same values as for nickel were used, as recommended in Bergström et al (1986).

Table A-5 Element specific translocation factors from surface to edible part of cereals and root crops ([Bq/kg w.w.]/[Bq/m^2]).

	Translocation factors (m^2/kg)			
Element	**Mean**	**Min**	**Max**	**Reference**
H	1E-01	1E-02	3E-01	Estimated
Be	1E-01	2E-02	3E-01	Estimated
C	1E-01	1E-02	3E-01	Estimated
Cl	1E-01	1E-02	3E-01	Estimated
Co	2E-01	1E-01	3E-01	Bergström & Nordlinder, 1991b
Ni	1E-02	5E-03	4E-02	Estimated
Se	1E-01	1E-02	3E-01	Estimated
Sr	4E-01	1E-01	7E-01	IAEA, 1994
Zr	1E-01	5E-02	2E-01	Bergström & Nordlinder, 1991b
Nb	2E-01	1E-01	3E-01	Bergström & Nordlinder, 1991b
Mo	1E-01	1E-02	3E-01	Estimated
Tc	5E-01	4E-01	6E-01	Bergström & Nordlinder, 1991b
Pd	1E-01	1E-02	3E-01	Estimated
Ag	1E-01	1E-02	3E-01	Bergström & Nordlinder, 1991b
Sn	1E-01	1E-02	3E-01	Bergström & Nordlinder, 1991b
I	1E-01	5E-02	2E-01	Bergström & Nordlinder, 1991b
Cs	2E-01	1E-01	3E-01	IAEA, 1994
Sm	1E-01	1E-02	3E-01	Estimated
Ho	1E-01	1E-02	3E-01	Estimated
Pb	3E-02	1E-02	1E-01	IAEA, 1994
Ra	1E-01	1E-02	3E-01	Estimated
Ac	1E-01	1E-02	3E-01	Estimated
Th	1E-01	1E-02	3E-01	Estimated
Pa	1E-01	1E-02	3E-01	Estimated
U	1E-01	1E-02	3E-01	Estimated
Np	1E-01	5E-02	2E-01	Estimated
Pu	2E-02	1E-02	3E-02	Bergström & Nordlinder, 1991b
Am	1E-02	5E-03	2E-02	Estimated
Cm	2E-02	1E-02	3E-02	Estimated

A.3 Distribution factors

Table A-6 Element specific distribution coefficients (Kd) for soil, concentration in solid matter/concentration in solution ([Bq/kg d.w.]/[Bq/m^3]).

	Kd-Soil (m^3/kg)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	-	-	-	-	*
Be	1E+0	LT	1E-1	1E+1	IAEA, 1994
C	1E-3	LT	4E-4	1E-2	Bergström & Nordlinder, 1990b
Cl	1E-3	LT	1E-4	1E-2	McKinley & Scholtis, 1992
Co	1E+0	LT	1E-2	2E+1	IAEA, 1994
Ni	5E-1	LT	5E-2	5E+0	IAEA, 1994
Se	1E-2	LT	1E-3	1E-1	Bergström & Nordlinder, 1990b
Sr	1E-2	LT	1E-3	1E-1	IAEA, 1994
Zr	1E+0	LT	1E-1	1E+1	IAEA, 1994
Nb	5E-1	LT	5E-2	5E+0	IAEA, 1994
Mo	1E-1	LT	1E-2	1E+0	IAEA, 1994
Tc	5E-3	LT	1E-3	1E-2	Bergström & Nordlinder, 1990b
Pd	2E-1	LT	2E-2	2E+0	IAEA, 1994
Ag	1E-1	LT	1E-2	1E+0	IAEA, 1994
Sn	1E-1	LT	5E-2	5E-1	Bergström & Nordlinder, 1990b
I	3E-1	LT	1E-1	1E+0	Bergström & Nordlinder, 1990b
Cs	1E+0	LT	1E-1	1E+1	Bergström & Nordlinder, 1990b
Sm	1E+0	LT	1E-1	1E+1	IAEA, 1994
Ho	1E+0	LT	1E-1	1E+1	IAEA, 1994
Pb	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1990b
Ra	5E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1990b
Ac	1E+0	LT	1E-1	1E+1	Bergström & Nordlinder, 1990b
Th	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Pa	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
U	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1990b
Np	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1990b
Pu	5E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Am	2E+0	LT	2E-1	2E+1	IAEA, 1994
Cm	1E+1	LT	1E+0	1E+2	IAEA, 1994

* Not considered.

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

	Kd-Peat (m^3/kg)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	-	-	-	-	*
Be	3E+0	LT	3E-1	3E+1	IAEA, 1994
C	7E-2	LT	7E-3	7E-1	Davis et al, 1993
Cl	1E-2	LT	1E-3	1E-1	Estimated
Co	1E+0	LT	5E-2	2E+1	IAEA, 1994
Ni	1E+0	LT	2E-1	7E+0	IAEA, 1994
Se	2E+0	LT	2E-1	2E+1	IAEA, 1994
Sr	2E-1	LT	4E-3	6E+0	IAEA, 1994
Zr	7E+0	LT	7E-1	7E+1	IAEA, 1994
Nb	2E+0	LT	2E-1	2E+1	IAEA, 1994
Mo	3E-2	LT	3E-3	3E-1	IAEA, 1994
Tc	2E-3	LT	4E-5	6E-2	IAEA, 1994
Pd	7E-1	LT	7E-2	7E+0	IAEA, 1994
Ag	2E+1	LT	2E+0	9E+1	IAEA, 1994
Sn	2E+0	LT	2E-1	2E+1	IAEA, 1994
I	3E-2	LT	3E-3	3E-1	IAEA, 1994
Cs	3E-1	LT	1E-1	3E+0	IAEA, 1994**
Sm	3E+0	LT	3E-1	3E+1	IAEA, 1994
Ho	3E+0	LT	3E-1	3E+1	IAEA, 1994
Pb	2E+1	LT	8E+0	6E+1	IAEA, 1994
Ra	2E+0	LT	2E-1	2E+1	IAEA, 1994
Ac	5E+0	LT	5E-1	5E+1	IAEA, 1994
Th	9E+1	LT	9E+0	9E+2	IAEA, 1994
Pa	7E+0	LT	7E-1	7E+1	IAEA, 1994
U	4E-1	LT	3E-3	4E+0	IAEA, 1994
Np	1E+0	LT	5E-1	3E+0	IAEA, 1994
Pu	2E+0	LT	2E-1	2E+1	IAEA, 1994
Am	1E+2	LT	1E+1	1E+3	IAEA, 1994
Cm	1E+1	LT	1E+0	1E+2	IAEA, 1994

* Not considered.

** The range given in the reference has been decreased.

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

	Kd-Lake sediment (m^3/kg)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	-	-	-	-	*
Be	1E+0	LT	1E-1	1E+1	Estimated
C	1E-3	LT	1E-4	1E-2	Bergström & Nordlinder, 1990b
Cl	1E+0	LT	1E-1	1E+1	McKinley & Scholtis, 1992
Co	5E+0	LT	1E+0	7E+1	IAEA, 1994
Ni	1E+1	LT	1E+0	1E+2	McKinley & Scholtis, 1992
Se	5E+0	LT	1E+0	1E+1	Bergström & Nordlinder, 1990b
Sr	1E+0	LT	1E-1	1E+1	IAEA, 1994
Zr	1E+0	LT	1E-1	1E+1	IAEA, 1994
Nb	1E+1	LT	1E+0	1E+2	McKinley & Scholtis, 1992
Mo	1E-3	LT	1E-4	1E-2	Aggeryd & Bergström, 1990
Tc	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1990b
Pd	2E+0	LT	2E-1	2E+1	McKinley & Scholtis, 1992
Ag	2E+0	LT	2E-1	2E+1	McKinley & Scholtis, 1992
Sn	5E+1	LT	1E+1	1E+2	Bergström & Nordlinder, 1990b
I	3E-1	LT	1E-1	1E+0	Bergström & Nordlinder, 1990b
Cs	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Sm	5E+0	LT	5E-1	5E+1	McKinley & Scholtis, 1992
Ho	3E-1	LT	3E-2	3E+0	McKinley & Scholtis, 1992
Pb	5E-2	LT	1E-2	1E-1	Bergström & Nordlinder, 1990b
Ra	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Ac	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Th	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1990b
Pa	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1990b
U	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Np	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
Pu	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1990b
Am	5E+0	LT	5E-1	5E+1	IAEA, 1994
Cm	5E+0	LT	1E-1	7E+1	IAEA, 1994

* Not considered.

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

	Kd-Baltic Sea sediment (m^3/kg)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	1E-3	LT	5E-5	1E-2	IAEA, 1985
Be	1E+0	LT	1E-1	1E+1	Estimated
C	1E-3	LT	1E-4	1E-2	McKinley & Scholtis, 1992
Cl	1E-3	LT	1E-4	1E-2	IAEA, 1985
Co	1E+2	LT	1E+0	2E+2	Bergström & Nordlinder, 1993
Ni	1E+1	LT	1E+0	1E+2	McKinley & Scholtis, 1992
Se	5E+0	LT	1E+0	1E+1	Bergström & Nordlinder, 1991c
Sr	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1991c
Zr	5E+1	LT	5E+0	5E+2	Bergström & Nordlinder, 1991c
Nb	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Mo	1E-3	LT	1E-4	1E-2	Aggeryd & Bergström, 1990
Tc	1E-1	LT	1E-2	1E+0	Bergström & Nordlinder, 1991c
Pd	1E+1	LT	1E+0	1E+2	McKinley & Scholtis, 1992
Ag	1E+0	LT	1E-1	1E+1	Bergström & Nordlinder, 1993
Sn	5E+1	LT	1E+1	1E+2	Bergström & Nordlinder, 1991c
I	3E-1	LT	1E-1	1E+0	Bergström & Nordlinder, 1991c
Cs	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Sm	1E+2	LT	1E+1	1E+3	McKinley & Scholtis, 1992
Ho	1E-1	LT	1E-2	1E+0	McKinley & Scholtis, 1992
Pb	5E-2	LT	1E-2	1E-1	Bergström & Nordlinder, 1991c
Ra	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Ac	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Th	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1991c
Pa	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1991c
U	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Np	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Pu	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1991c
Am	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Cm	1E+3	LT	1E+1	2E+3	Bergström & Nordlinder, 1993

A.4 Transfer coefficients for milk and meat

Table A-10 Element specific transfer coefficients to cow milk from daily intake ([Bq/l]/[Bq intake/day]).

		F-milk (day/l)			
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	2.0E-02	LT	1.0E-2	3.0E-2	Bergström & Nordlinder, 1993
Be	9.0E-07	LT	9.0E-8	9.0E-6	Estimated
C	1.0E-02	LT	5.0E-3	2.0E-2	Bergström & Nordlinder, 1990b
Cl	1.7E-02	T	1.5E-2	2.0E-2	IAEA, 1994
Co	3.0E-04	LT	6.0E-5	1.0E-2	IAEA, 1994
Ni	2.0E-02	LT	2.0E-3	5.0E-2	IAEA, 1994
Se	4.0E-03	LT	4.0E-4	4.0E-2	Davis et al, 1993
Sr	2.8E-03	LT	1.0E-3	3.0E-3	IAEA, 1994
Zr	6.0E-07	LT	6.0E-8	6.0E-6	IAEA, 1994
Nb	4.0E-07	LT	1.0E-7	4.0E-6	IAEA, 1994
Mo	1.0E-03	LT	1.0E-4	1.0E-2	IAEA, 1994
Tc	2.0E-05	LT	1.0E-5	1.0E-3	IAEA, 1994
Pd	1.0E-03	LT	1.0E-4	1.0E-2	Bergström et al, 1986
Ag	5.0E-05	LT	5.0E-6	5.0E-4	IAEA, 1994
Sn	3.0E-03	LT	1.0E-3	1.0E-2	Bergström & Nordlinder, 1990b
I	1.0E-02	LT	1.0E-3	4.0E-2	IAEA, 1994
Cs	8.0E-03	LT	1.0E-3	3.0E-2	IAEA, 1994
Sm	2.0E-05	LT	2.0E-6	2.0E-3	Aggeryd & Bergström, 1990
Ho	2.5E-06	LT	3.0E-7	3.0E-5	Aggeryd & Bergström, 1990
Pb	3.0E-04	LT	2.0E-5	2.0E-3	Bergström & Nordlinder, 1990b
Ra	1.3E-03	LT	1.0E-4	2.0E-3	IAEA, 1994
Ac	3.0E-07	LT	3.0E-8	3.0E-6	Bergström & Nordlinder, 1990b
Th	5.0E-06	LT	1.0E-7	1.0E-4	Bergström & Nordlinder, 1990b
Pa	5.0E-05	LT	1.0E-6	1.0E-4	Bergström & Nordlinder, 1990b
U	4.0E-04	LT	7.0E-5	6.0E-4	IAEA, 1994
Np	5.0E-06	LT	5.0E-7	5.0E-5	IAEA, 1994
Pu	1.0E-06	LT	3.0E-9	3.0E-6	IAEA, 1994
Am	2.0E-06	LT	4.0E-7	2.0E-5	IAEA, 1994
Cm	2.0E-05	LT	2.0E-6	2.0E-4	Bergström & Nordlinder, 1993

Table A-11 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**	**Low**	**High**	**Reference**
H	1.0E-2	LT	1E-3	1E-1	Bergström & Nordlinder, 1993
Be	1.0E-3	LT	1E-4	1E-2	Estimated
C	3.0E-2	LT	1E-3	1E-1	Bergström & Nordlinder, 1990b
Cl	2.0E-2	LT	1E-2	4E-2	IAEA, 1994
Co	1.0E-2	LT	4E-5	7E-2	IAEA, 1994
Ni	5.0E-3	LT	5E-4	5E-2	IAEA, 1994
Se	1.5E-2	LT	1E-4	2E-2	Davis et al, 1993
Sr	8.0E-3	LT	3E-4	1E-2	IAEA, 1994
Zr	1.0E-6	LT	1E-7	1E-2	IAEA, 1994
Nb	3.0E-7	LT	3E-8	1E-2	IAEA, 1994
Mo	2.0E-3	LT	2E-4	2E-2	IAEA, 1994
Tc	1.0E-4	LT	1E-5	1E-3	IAEA, 1994
Pd	1.0E-3	LT	1E-4	1E-2	Bergström et al, 1986
Ag	3.0E-3	LT	2E-3	6E-3	IAEA, 1994
Sn	3.0E-3	LT	3E-4	3E-2	Bergström & Nordlinder, 1990b
I	4.0E-2	LT	7E-3	5E-2	IAEA, 1994
Cs	5.0E-2	LT	1E-2	6E-2	IAEA, 1994
Sm	5.0E-3	LT	5E-4	5E-2	Aggeryd & Bergström, 1990
Ho	5.0E-3	LT	5E-4	5E-2	Aggeryd & Bergström, 1990
Pb	4.0E-4	LT	1E-4	7E-4	IAEA, 1994
Ra	9.0E-4	LT	5E-4	5E-3	IAEA, 1994
Ac	1.0E-5	LT	1E-6	1E-4	Bergström & Nordlinder, 1990b
Th	6.0E-6	LT	6E-7	6E-5	Davis et al, 1993
Pa	1.0E-5	LT	1E-6	1E-4	Davis et al, 1993
U	3.0E-4	LT	3E-5	3E-3	IAEA, 1994
Np	1.0E-4	LT	1E-5	1E-3	Coughtrey et al, 1984
Pu	1.0E-5	LT	2E-7	2E-4	IAEA, 1994
Am	4.0E-5	LT	4E-6	1E-4	IAEA, 1994
Cm	2.0E-5	LT	2E-6	2E-4	Bergström& Nordlinder, 1993

A.5 Bioaccumulation factors to aquatic organisms

Table A-12 Element specific bioaccumulation factors to fish in fresh water ([Bq/kg w.w.]/[Bq/l]).

	Cf-lake fish (l/kg w.w.)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	1E+0	LT	6E-1	1E+0	IAEA, 1994
Be	1E+2	LT	1E+1	1E+3	IAEA, 1994
C	5E+4	LT	1E+3	6E+4	Bergström & Nordlinder, 1990b
Cl	5E+1	LT	1E+1	1E+2	Coughtrey et al, 1985
Co	3E+2	LT	1E+1	4E+2	IAEA, 1994
Ni	1E+2	LT	1E+1	1E+3	IAEA, 1994
Se	2E+3	LT	5E+2	5E+3	Bergström & Nordlinder, 1990b
Sr	6E+1	LT	1E+0	1E+3	IAEA, 1994
Zr	2E+2	LT	3E+0	3E+2	IAEA, 1994
Nb	3E+2	LT	1E+2	3E+4	IAEA, 1994
Mo	1E+1	LT	1E+0	1E+2	IAEA, 1994
Tc	2E+1	LT	2E+0	8E+1	IAEA, 1994
Pd	1E+2	LT	1E+1	1E+3	Bergström et al, 1986
Ag	5E+0	LT	2E-1	1E+1	IAEA, 1994
Sn	3E+3	LT	3E+2	3E+4	IAEA, 1994
I	2E+2	LT	1E+1	6E+2	Bergström & Nordlinder, 1990b
Cs	1E+4*	LT	5E+3*	2E+4*	Bergström & Nordlinder, 1990b
Sm	3E+1	LT	2E+0	3E+2	Aggeryd & Bergström, 1990
Ho	3E+1	LT	3E+0	3E+2	Aggeryd & Bergström, 1990
Pb	3E+2	LT	1E+2	4E+2	IAEA, 1994
Ra	5E+1	LT	1E+1	2E+2	IAEA, 1994
Ac	1E+2	LT	1E+1	2E+2	Bergström & Nordlinder, 1990b
Th	1E+2	LT	3E+1	1E+3	Poston, 1986 + IAEA, 1994
Pa	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1990b
U	1E+1	LT	2E+0	5E+1	IAEA, 1994
Np	5E+1	LT	1E+1	3E+3	Poston, 1986 + IAEA, 1994
Pu	3E+1	LT	4E+0	3E+2	IAEA, 1994
Am	3E+1	LT	1E+1	3E+2	IAEA, 1994
Cm	3E+1	LT	1E+1	3E+2	IAEA, 1994

* Oligotrophic fresh water.

Table A-13 Element specific bioaccumulation factors for fish to be used in the coast modules ([Bq/kg w.w.]/[Bq/l]).

	Cf-Baltic fish (l/kg w.w.)				
Element	**B.E**	**Distr**	**Low**	**High**	**Reference**
H	1E+0	T	8E-1	1E+0	Bergström & Nordlinder, 1992b
Be	2E+2	LT	2E+1	2E+3	Estimated
C	2E+3	T	2E+3	3E+3	Bergström & Nordlinder, 1992b
Cl	1E+0	LT	1E-1	1E+1	Coughtrey et al, 1985
Co	3E+2	LT	3E+1	5E+2	Bergström & Nordlinder, 1992b
Ni	3E+2	LT	3E+1	5E+2	Bergström & Nordlinder, 1992b
Se	4E+3	LT	2E+3	8E+3	Bergström & Nordlinder, 1991c
Sr	3E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1992b
Zr	1E+2	LT	1E+1	2E+2	Bergström & Nordlinder, 1992b
Nb	1E+2	LT	1E+1	5E+2	Bergström & Nordlinder, 1992b
Mo	1E+1	LT	1E+0	5E+1	Bergström & Nordlinder, 1992b
Tc	3E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1992b
Pd	1E+1	LT	1E+0	1E+2	Bergström et al, 1986
Ag	5E+2	LT	1E+2	1E+3	Bergström & Nordlinder, 1992b
Sn	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1992b
I	3E+1	LT	1E+1	1E+2	Bergström & Nordlinder, 1992b
Cs	2E+2	LT	1E+2	5E+2	Bergström & Nordlinder, 1992b
Sm	3E+1	LT	3E+0	3E+2	Aggeryd & Bergström, 1990
Ho	3E+1	LT	3E+0	3E+2	Aggeryd & Bergström, 1990
Pb	1E+2	LT	5E+1	2E+2	Bergström & Nordlinder, 1991c
Ra	5E+1	LT	1E+1	1E+2	Bergström & Nordlinder, 1991c
Ac	1E+2	LT	1E+1	1E+3	Bergström & Nordlinder, 1991c
Th	3E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
Pa	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1991c
U	5E+1	LT	1E+1	1E+2	Bergström & Nordlinder, 1991c
Np	1E+1	LT	1E+0	1E+2	Bergström & Nordlinder, 1992b
Pu	3E+1	LT	5E+0	5E+1	Bergström & Nordlinder, 1992b
Am	1E+2	LT	1E+1	2E+2	Bergström & Nordlinder, 1992b
Cm	5E+1	LT	1E+1	3E+2	Bergström & Nordlinder, 1992b

Table A-14 Element specific bioaccumulation factors for fresh water invertebrates and for marine waters*, water to marine plants ([Bq/kg w.w.]/[Bq/l]) (Thompson et al, 1972).

	Invertebrates (l/kg -w.w.)				Marine plants (l/kg w.w.)			
Element	**B.E**	**Distr**	**Low**	**High**	**B.E**	**Distr**	**Low**	**High**
H	1E+0	LT	5E-1	2E+0	1E+0	LT	5E-1	2E+0
Be	1E+1	LT	1E+0	1E+2	1E+3	LT	1E+2	1E+4
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	T	1E+0	1E+2	1E+1	0	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
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
Ho	1E+3	LT	1E+2	1E+4	5E+3	LT	1E+2	1E+4
Pb	1E+2	LT	1E+1	1E+3	1E+3	LT	1E+2	1E+4
Ra	3E+2	LT	3E+1	3E+3	1E+2	LT	1E+1	1E+3
Ac	1E+3	LT	1E+2	1E+4	5E+3	LT	1E+2	1E+4
Th	5E+2	LT	5E+1	5E+3	3E+3	LT	3E+2	1E+4
Pa	1E+2	LT	1E+1	1E+3	6E+0	LT	6E-1	6E+1
U	1E+2	LT	1E+1	1E+3	7E+1	LT	7E+0	7E+2
Np	4E+2	LT	4E+1	4E+3	6E+0	LT	6E-1	6E+1
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

* Values used for algae in the Baltic Sea.

Expressions for exposure pathways

B.1 Introduction

Doses from exposure pathways are calculated based upon concentration of each radionuclide in relevant compartments.

The method for dose calculations follows the principles recommended by IAEA for radiological impact assessment (IAEA 1994).

The following expression is used for the dose via ingestion

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

and via inhalation

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

and external doses

$$D_{ext} = C_s \cdot \rho \cdot H_i \cdot DC_{ext}$$

where

DC_{ing} = Dose coefficients for ingestion [Sv/Bq] according to ICRP, see Table 2-1
HC_i = Consumption rate for pathway i [kg or litre per year], see Table B-2
U_i = Concentration in foodstuff i [Bq per kg or litre] (expression given below)
DC_{inh} = Dose coefficients for inhalation [Sv/Bq] according to ICRP, see Table 2-1
C_a = Concentration of radionuclides in air [Bq/m^3]
IH = Inhalation rate [m^3/h], see Table B-2
H_i = Exposure time [h/year], see Table B-2
DC_{ext} = External dose coefficients [(Sv/h)/(Bq/m^3)], see Table 2-1
C_s = Concentration of radionuclides in soil [Bq/kg]
ρ = Soil density [kg/m^3], see Table 3-1

B.2 Concentration in food-stuff

Consumption of agricultural products is represented by four groups of food, i.e. milk, meat, vegetables and cereals. Three groups, fish, crustacea and algae represent food from aquatic systems. For each of these groups the concentration in the food is calculated as follows.

B.2.1 Milk and meat

Transfer of radionuclides to milk U_m and meat U_f originates in these modules from cattle's intake of fodder, soil and water. The concentration in milk is obtained from

$$U_m = In \cdot F_m$$

and in meat

$$U_f = In \cdot F_f$$

where

In = Daily intake of nuclides [Bq/day]

F = Element specific distribution coefficient for milk and meat, respectively [day/litre, day/kg], see Tables A-10 and A-11.

Fodder is contaminated through root uptake and surface retention. Fodder consists of pasturage and cereals. Cereals represent concentrated food. Additionally, some inadvertent consumption of soil when grazing is assumed. The cow's daily intake of radionuclides (In) is calculated from

$$In = MC_w \cdot UC_w + MC_p \cdot UC_p + MC_c \cdot UC_c + MC_s \cdot UC_s$$

where

MC_i = The daily consumption [kg or litre per day], see Table B-1

UC_i = The concentration of radionuclides in foodstuff, water and soil, respectively [Bq per kg or litre] (expression given below)

i = w for water
i = p for pasturage
i = c for concentrated food (cereals)
i = s for soil

The concentrations of radionuclides in water and soil are obtained from the dispersion model. The concentration in cereals that are used as concentrate to cattle is assumed to have the same concentration as in cereals for human consumption, which is given below. The resulting intake of radionuclides due to consumption of pasturage and water plants is expressed by

$$UC_p = (365\text{-}ND)/365 \quad UC_{pg} + ND/365 \cdot UC_{pw}$$

obtained from consumption of pasturage

$$UC_{pg} = C_s \cdot B_p + \frac{C_w}{Y_p} \cdot \frac{I}{t_{tot}} \cdot \int_0^{t_n} e^{-\tau \cdot t_n} dt$$

and consumption of water plants

$$UC_{pw} = \frac{C_w \cdot TR \cdot GD \cdot 10^{-3} \cdot 24}{Y_{wp}}$$

where

C_s = Concentration of radionuclides in soil [Bq/kg]

B_p = Soil to pasturage transfer factors [(Bq/kg)/(Bq/kg)], see Table A-1

C_w = Concentration of radionuclides in irrigation water [Bq/l]

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

t_n = Time between irrigation occasion and end of irrigation period [days]

t_{tot} = Irrigation period, see Table 3-1

τ = $\ln 2/T_{½W}$ where $T_{½W}$ = weathering constant [day^{-1}], see Table 3-1

n = Number of irrigation occasions, see Table 3-1

Y_p = Yield of pasturage [kg/m^2], see Table B-3

TR = Transpiration [g/(m^2,h)], see Table 3-6

ND = Number of days of the year cattle grazing on shores, see Table 3-6

GD = Mean average time for plant transpiration before animal´s consumption (days), see Table B-3

Y_{wp} = Yield of water plants [kg/m^2], see Table 3-6

10^{-3} = l/g

24 = h/day

B.2.2 Crops

Crops are represented by cereals, root crops and green vegetables. Grain and root crops are contaminated from root uptake, retention on surfaces and translocation. The expression for resulting concentration in cereals and root-crop products is

$$U_i = C_s \cdot B_i + \sum_{o}^{n} I_n \cdot TL \cdot C_w$$

where

C_s = Concentration of radionuclides in soil [Bq/kg]

B_i = Soil to plant transfer factors [(Bq/kg)/Bq/kg)], see Table A-2, A-3
i = c for cereals
i = r for root-crops

n = Number of irrigation occasions, see Table 3-1

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

TL = Translocation from surface to edible parts of plant, [(Bq/kg)/(Bq/m^2)], see Table A-18

C_w = Concentration of radionuclides in irrigation water [Bq/m^3]

Vegetables are 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. Therefore the mean concentration of surface contamination is calculated. The expression for the resulting content of radionuclides in vegetables is

$$U_v = C_s \cdot B_v + \frac{C_w}{Y_w} \cdot \frac{I}{t_{tot}} \cdot \int_{n\ 0}^{t_n} e^{-\lambda t_n} dt$$

where

C_s = Concentration of radionuclides in soil [Bq/kg]
B_v = Soil to plant transfer factor [(Bq/kg)/Bq/kg)], see Table A-4
C_w = Concentration of radionuclides in irrigation water [Bq/l]
Y_w = Yield of vegetables [kg/m^2]

For other abbreviations, see B.1.1 above.

B.2.3 Food from aquatic systems

Food from aquatic systems is represented by fish, crustacea and algae. The concentrations of radionuclides in those groups of species are obtained by use of bioaccumulation factors for edible parts of the species relative to total concentration of respective nuclide in water and suspended matter. The bioaccumulation factors are valid for steady-state conditions and implicitly consider all paths from the surrounding environment. The expression for resulting content of radionuclides in fish, crustacea and algae is

$$U_i = B_i \cdot C_w$$

where

B_i = Concentration factors water to edible part of the species [(Bq/kg)/(Bq/l)], see Tables A-15 – A-17

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

C_w = Concentration of soluble and suspended matter in surrounding water [Bq/l]

B.3 Inhalation

Radionuclides in air emanate from two sources; dust in air from soil and exhaust gases from heating plants fuelled with peat. Dust is assumed to have the same concentration of radioactivity as soil. The concentration of radionuclides in air (C_a) is expressed by

$$C_a = C_s \cdot S$$

and

$$C_a = C_p \cdot FC \cdot RD \cdot FE$$

where

C_s = Concentration of radionuclides in soil or peat [Bq/kg]

S = Dust content in air [kg/m^3]. see Table 3-17

C_p = Concentration of radionuclides in peat [Bq/kg]

FC = Fuel consumption [kg/s], see Table 3-17

RD = Relative dispersion as an annual mean value [s/m^3], see Table 3-17

FE = Fraction of nuclides which leaves the plant via exhaust gases (efficiency of the filter system), see Table 3-17

B.4 Data

The consumption rates used for cattle and man are shown in tables below. I addition, inhalation rates and exposure times are shown in Table B-2. Other generic data for the dose calculations are summarised in Table B-3.

Table B-1 Data to calculate cattle's intake of radionuclides, triangularly distributed.

Parameter	Unit	Mean	Min/Std	Max	Reference
Water	l/day	70	60	80	Bertilsson (personal comm)
Pasturage/water plants	kg dw/day	5	4	6	Morén (personal comm)
Cereals	kg/day	12	10	15	Morén (personal comm)
Soil	kg/day	0.1	0.05	0.15	Assumed values
Average time plant transpiration	day	100	80	120	Assumed values
Shore grazing period	days/year	90	75	100	Assumed values

Table B-2 Human's consumption rates used in the dose calculations.

Parameter	Distr**	Unit	Mean	Std	Max	Reference
Water	N	l/year	600	10 %		Bergström & Nordlinder, 1990b
Milk	N	l/year	200	10 %		Bergström & Nordlinder, 1990b
Meat	N	kg/year	55	10 %		Bergström & Nordlinder, 1990b
Vegetables	N	kg/year	40	10 %		Bergström & Nordlinder, 1990b
Root-crops/Potatoes	N	kg/year	70	10 %		Bergström & Nordlinder, 1990b
Cereals	N	kg/year	80	10 %		Bergström & Nordlinder, 1990b
Soil	N	kg/year	0.01	10 %		Assumed values
Fish	N	kg/year	30	10 %		Bergström & Nordlinder, 1990b
Crustacea	N	kg/year	2	10 %		Assumed values
Algae	N	kg/year	2	10 %		Assumed values
Exposure time*	T	h/year	100	50	150	Assumed values
Inhalation rate	T	m^3/h	1	0.8	1.2	ICRP 23

* Time of external exposure from soil, and for inhalation doses' duration in contaminated air.

** Type of distribution: N = normal distribution, T = triangular distribution.

Table B-3 Data of common parameters in the modules used in the dose calculations, triangularly distributed.

Parameter	Unit	Mean	Min/Std	Max	Reference
Yield of pasturage	kg dw/m^2	0.5	0.4	0.6	Haak, 1983
Yield of cereals	kg dw/m^2	0.5	0.4	0.6	Assumed values
Yield of root crops	kg dw/m^2	3	2	4	Assumed values
Yield of vegetables	kg dw/m^2	3	2	4	Assumed values
Dust concentration in air	kg/m^3	1E-4	3E-5	3E-4	Haak, 1983