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Model of the long-term transfer of radionuclides in forests

Rodolfo Avila, Facilia AB

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Svensk Kärnbränslehantering AB
Swedish Nuclear Fuel
and Waste Management Co
Box 5864
SE-102 40 Stockholm Sweden
Tel 08-459 84 00
+46 8 459 84 00
Fax 08-661 57 19
+46 8 661 57 19

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This report concerns a study which was conducted for SKB. The conclusions and viewpoints presented in the report are those of the author and do not necessarily coincide with those of the client.

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Abstract

This report describes a model of the long-term behaviour in temperate and boreal forests of radionuclides entering the ecosystem with subsurface water. The model can be applied for most radionuclides that are of relevance in safety assessment of repositories for high-level radioactive waste. The model can be used for estimating radionuclide concentrations in soil, trees, understorey plants, mushrooms and forest mammals. The report also includes a discussion on alternatives approaches for dealing with data gaps, in particular for radionuclide transfer factors to plants and animals. A recommended (nominal) value and an interval of variation are provided for each model parameter and a classification of parameters by the degree of confidence in the values is given. Model testing against existing empirical data showing satisfactory results is also presented.

As part of the safety assessment of repositories for final disposal of high-level radioactive waste, it is necessary to carry out simulations of the long-term behaviour of radionuclides in different ecosystems. The forest is one of the ecosystems of interest, being well represented in the candidate sites for location of a repository in Sweden. Forests can play an important role in the spatial and temporal distribution of radionuclides in the environment. This is because roots of forest trees can directly extract radionuclides released to groundwater. Also, the transpiration of forest trees can alter the discharge of radionuclides into watersheds by regulating the run-off. Despite of this, forest ecosystems have not been addressed in previous safety assessments. This can be explained by the fact that a suitable model of the long-term transfer of a wide range of radionuclides in forests has not been readily available.

The objective of this work was to develop a forest model applicable for a wide range of radionuclides of relevance for high level radioactive waste management (Am241, Cl36, Cs135, I129, Ni59, Np237, Pu239, Ra226, Sr90, Tc99, Th232, U238) that can potentially enter the ecosystem with contaminated groundwater.

The model assumes that biomass growth, precipitation and evapotranspiration drive the radionuclide cycling in the system by influencing the uptake of radionuclides by vegetation and their export from the system via runoff. The mathematical model of radionuclide transfer consists of a system of ordinary differential describing the mass balance in different forest compartments, taking into account the fluxes in and out from the compartment and the radionuclides decay. The fluxes between compartments are calculated by multiplying a transfer coefficient (*TC*) by the radionuclide inventory in the compartment. The model assumes that the fluxes of radionuclides are driven by fluxes of water and nutrients and hence the *TCs* are expressed as function of ecological parameters, such as biomass growth, and evapotranspiration. The following radionuclide fluxes are included in the model: flux from soil to tree wood via root uptake, flux from soil to tree leaves via root uptake, flux from soil to understorey (plants and mushrooms) via root (mycelia) uptake, flux from tree leaves to litter by leaves fall, flux from tree wood to litter by wood fall, flux from understorey plants to litter by plant senescence, flux from litter to soil following litter decomposition.

In the report alternative approaches to describe the transfer from soil to plants are presented. In the simpler approach, applicable when soil to plant concentration ratios (*CR*) are available for the radionuclide or its stable analogue, the root uptake rates are calculated by multiplying the concentration in the plant, obtained with the help of the *CR*, by the biomass production. A second approach is based on the assumption that some elements are taken-up passively with the transpiration flux. For them, the total flux from soil to plants can be expressed as a function of the transpiration rate and the radionuclide concentration in the pore water. The *CRs* from soil to understorey plants as well as the *CRs* from soil to tree leaves and tree wood were estimated by performing probabilistic simulations. It was assumed that roots have the same permeability for radionuclides and water.It should be taken into account, that the literature data used in the comparison were obtained at different sites and using different methods. A better agreement could be achieved if the model parameters and the empirical data of *CR*s are obtained for the same site. Hence, it can be concluded that this approach has good perspectives, in particular for radionuclides that are not analogues of plant nutrients, such as the actinides, and also analogues of plant micronutrients, for example Cl36 and Ni59.

For some radionuclides, the approach based on transpiration fluxes gave an underestimation (Cesium and Radium) or overestimation (Iodine and Technetium) of the *CRs*. The overestimation could be partly explained by the fact that the permeability coefficients were set equal to 1, when they are most likely lower than one. However, in some cases the differences might abide more fundamental reasons, for instance that the implicit assumption of linear proportionality of the radionuclide uptake rates to the transpiration rate and the radionuclide concentration in the soil solution might not hold. For these radionuclides, and in particular for analogues of plant macronutrients, an alternative approach was implemented based on the assumption that their uptake by plants is modulated by the plant uptake of the nutrient. This means that the radionuclide and its corresponding analogue nutrient are taken up by plants in an identical manner via the same carrier molecules. Assuming that only ions in the soil solution near the roots, where the radionuclide concentrations are much lower than analogue concentrations, are available for transition into the roots, the transition of radionuclides from soil to plants can be represented as an independent Poisson process. In this case, the uptake rate of the radionuclide will be proportional to the uptake rate of the analogue nutrient and the concentration of the radionuclide in the soil solution near the roots and inversely proportional to the analogue concentration in the soil solution near the roots.

Transfer factors to forest wild animals are lacking for many of the relevant radionuclides. Hence, an alternative approach was introduced which uses an allometric equation relating the radionuclide concentration in the animal diet to the radionuclide concentration in the animal body. The allometric equation was derived by incorporating allometric relationships for the dry matter intake by the animal; the animal lifetime and the radionuclide biological half-time into a kinetic model of the radionuclides turnover in animals. In order to test the model, predictions of the transfer factor (*TF*) from soil to herbivores (expressed in Bq/kg fresh weight per Bq/kg dry weight) were compared with empirical values found in the literature. For Caesium and Strontium the predicted *TFs* were within the range of empirical observations. The model predictions were slightly higher for Radium and Uranium and slightly lower for Thorium. However, it should be noted that the intervals given for these three elements are based on few empirical data.

Sammanfattning

Denna rapport beskriver en modell av långsiktig transport av radionuklider i tempererade/boreala skogsekosystem. Syftet med detta arbete är att utveckla en skogsmodell som går att applicera på en mängd olika radionuklider som kan vara av intresse för hantering av högaktivt avfall (Am-241, Cl-36, Cs-135, I-129, Ni-59, Np-237, Pu-239, Ra-226, Sr-90, Tc-99, Th-232, U-238), och med en potential att kontaminera grundvattnet. Modellen kan, förutom att beräkna nuklidtransport, användas för att uppskatta koncentrationen av radionuklider i jord, växter, svampar och däggdjur. I rapporten diskuteras också alternativa sätt att hantera brister i data, speciellt när det gäller omvandlingsfaktorer till växter och djur. Dessutom rekommenderas värden och variationsintervaller för varje parameter i modellen. Parametrarna klassificeras sedan med hjälp av deras konfidensintervall, och modellen testas mot platsspecifika data.

Som en del av säkerhetsanalysen för slutförvaring av högaktivt avfall är det nödvändigt att göra långtidssimuleringar för att kunna beräkna radionuklidtransporten i olika ekosystem. Skogen är ett ekosystem som är av stort intresse eftersom det är välrepresenterat i båda svenska kandidatområdena. Skogen har också en betydande roll för utbredningen av radionuklider i både tid och rum, eftersom trädrötter kan ta upp radionuklider direkt från grundvattnet. Avdunstningen från träden kan dessutom styra utsläpp av radionuklider som sker genom avrinningen. Trots detta har skogsekosystemen fått liten uppmärksamhet i tidigare säkerhetsanalyser. Detta kan dock förklaras av att endast få modeller för långtidssimuleringar av radionuklidtransporter i skog tidigare varit tillgängligt.

Modellen förutsätter att produktion av biomassa, nederbörd och avdunstning driver radionuklidtransporten i ekosystemen, där nukliderna tas upp av vegetationen via vatten och lämnar systemet via avrinning. Den matematiska modellen för transport av nuklider beskriver massbalansen i skogsekosystemets olika delar, vilket inkluderar fluktuationer mellan delar samt ackumulation av radionuklider i vissa delar av systemet. Flödet inom ekosystemet har räknats fram genom att multiplicera transportkoefficienten (TC) med mängden radionuklider i varje del. Modellen förutsätter att flödet av radionuklider drivs med hjälp av flödet av vatten och näring, vilket gör att TC kan uttryckas som en funktion av ekologiska parametrar, som till exempel transport från jord till blad i trädskit och fältskit via trädrötter, eller omvänt, transport från löv och andra växtdelar till jord och nedbrytning av förna.

I rapporten presenteras alternativa sätt att beskriva transporten av radionuklider från jord till växter. Ett relativt enkelt tillvägagångssätt, som kan tillämpas när jord-plant koncentrationsfaktorer för radionuklider eller deras analoger (CR) finns att tillgå, är att multiplicera biomassproduktionen med den beräknade koncentrationen i växten från CR. Ett annat tillvägagångssätt är baserat på antagandet att olika element tas upp passivt i växten med hjälp av avdunstningen. För dessa element kan det totala flödet från jord till växt uttryckas som en funktion avdunstningen och nuklidkoncentrationen som finns i jorden. CR från jord till fältskiktet liksom från jord till träd (löv och trä) kan uppskattas med hjälp av sannolikhetssimuleringar, där rötterna förväntades ha samma permeabilitet för radionuklider som för vatten. För simuleringarna användes litteraturdata från olika platser framtagna med olika metoder, vilket medför viss osäkerhet. En större samstämmighet uppnås om man istället använder parameterdata och empiriska data för CR från samma plats. Modeller har dock goda förutsättningar att vidareutvecklas, speciellt för nuklider som inte är analoga till växtnäring, som aktinider och för de som är analoga till näringsspårämnen exempelvis Cl-36 och Ni-59.

Transport av radionuklider beräknat med hjälp av avdunstningen underskattade CR för vissa nuklider (cesium och radium), medan andra överskattades (jod och teknetium). Överskattningen kan delvis förklaras genom att permeabilitetskoefficienten antogs vara lika med 1, fastän den antagligen är lägre. Det kan också bero på att upptaget av radionuklider inte är linjärt proportionellt mot avdunstningen eller att uppskattningen av radionuklidkoncentrationen i jorden är osäker. För dessa nuklider, användes att alternativt sätt att beräkna upptag som bygger på antagandet att nukliduppptaget är anpassat till näringsupptaget i växten. Detta betyder att nukliden och det analoga växtnäringsämnet tas upp i växten på samma sätt via samma transportmolekyler. Upptaget av radionuklider kommer därför att vara proportionellt mot upptaget av det analoga växtnäringsämnet och koncentrationen av nuklider i jorden nära rötterna, medan det kommer att vara omvänt proportionellt mot koncentrationen av växtnäringsämnet i jorden nära rötterna.

Det finns fortfarande brister i kunskapen om överföring av radionuklider till skogslevande däggdjur. För att ändå kunna beräkna detta, användes ett alternativt sätt där koncentrationen av radionuklider i djurets diet relaterades till djurets storlek. Den allometriska relationen mellan mängden foder (torrvikt) upptaget av djuret, djurets livslängd och radionuklidens biologiska halveringstid analyserades i en kinetisk modell av nukliders omsättning i djur. För att testa modellen, överföring av radionuklider från jord till gräsätande djur (uttryckt som Bq/kg friskvikt per Bq/kg torrvikt), jämfördes platsspecifika data med litteraturdata. För cesium och strontium överensstämde överföringsfaktorn i litteraturen med observerade data. Modellens prediktioner för radium och uran visade däremot något för höga värden, medan torium uppvisade för låga värden. Det bör dock nämnas att det endast fanns få observerade data för dessa nuklider.

Contents

1 Introduction

As part of the safety assessment of the proposed repository for final disposal of high-level radioactive waste, it is necessary to carry out simulations of the long-term behaviour of radionuclides in different ecosystems. The forest is one of the ecosystems of interest, being well represented in the candidate sites for location of a repository in Sweden. Moreover, forests can play an important role in the spatial and temporal distribution of radionuclides in the environment. This is because roots of forest trees can directly extract radionuclides released to groundwater. Also, forests can alter the discharge of radionuclides into watersheds via transpiration fluxes. Despite this, forest ecosystems have not been addressed in previous safety assessments. This can be explained by the fact that a suitable model of the long-term transfer of radionuclides in forests has not been available. Consequently, a project was started with the aim of developing a forest model that could be used for relevant radionuclides in scenarios of belowground contamination.

In this report, a first version of a forest model is presented, which is based on models and data found in the literature. The main emphasis in this version of the model has been to describe the transfer to plants and animals from a single soil layer receiving a radionuclide input. Hence, this version can be directly used for forests with shallow groundwater tables, but it could be easily adapted for situations with deep groundwater tables, for example by introducing several soil layers.

In Chapter 2 an overview of models found in the literature is presented. The proposed model is described in Chapter 3, including the conceptual assumptions, the mathematical equations and tables with ecological and radionuclide-dependent parameters required by the model. The estimation of parameter values is discussed in detail, including an evaluation of the degree of confidence in the values provided. In Chapter 4 examples of model simulations are presented, including comparisons of model predictions against empirical data. Chapter 5 contains a discussion of the approaches adopted for modelling different transfer processes, as well as on processes not included in the model. In particular, alternative approaches for modelling the uptake of radionuclides by plants are discussed in detail.

2 Published models of radionuclide transfer in forests

The earliest mathematical models of radionuclide transfer in forest ecosystems were developed from data obtained in experiments with caesium inoculation /Olson 1965/ and from measurements of quantities and fluxes of stable strontium and manganese in a tropical rain forest /Jordan et al. 1973/. Other early models were based on knowledge obtained from studies in forests contaminated by fallout from weapon tests /Croom and Ragsdale 1980/, by releases during the Kyshtym accident /Prohorov and Ginzburg 1973, Alexakhin et al. 1976/ and by releases during the Manhattan project /Garten et al. 1978, Van Voris et al. 1990/. The Chernobyl accident lead to high radioactive contamination of large forested areas where substantial amount of empirical data were collected and subsequently incorporated into several models of radiocaesium transfer in forests /Bergman et al. 1993, Schell et al. 1996, Avila 1998, Mamikhin and Klyashtorin 1999, Belli 2000/. In addition, a few models have been developed to address forest contamination from disposed radioactive waste /Garten 1987, 1999/ and more recently to describe the transfer in forests of some radionuclides present in spent nuclear fuel, Tc99, I129 and Cl36 /Bostock 2004/. Comparative analyses of models published before 1999 can be found in several published reviews /Avila 1998, Riesen et al. 1998, Shaw et al. 2003/.

Table 2-1 provides information on the radionuclides, forest types, compartments and contamination sources considered in published models. All these models belong to the type commonly known as compartment models that describe the transfer of radionuclides in the system with ordinary differential equations (ODE). Nine out of the 15 published models deal with Cs137 and have been developed to address scenarios of atmospheric contamination. Some of the models, /Avila et al. 2001a/, FORESTPATH, FORESTLAND and RIFE, have been compared with each other and with empirical data in the frame of an EC project /Moberg et al. 1999/ and the IAEA Programme BIOMASS /IAEA 2001/. The comparisons have shown good agreement between models, as well as between the models and the data /Goor and Avila 2003/. There exist three models for Sr90, two of which deal with atmospheric deposition /Jordan et al. 1973, Prohorov and Ginzburg 1973, Alexakhin et al. 1976, 1994/ and one with inputs via contaminated groundwater /Garten 1999/. These models cover a wide range of forest conditions, temperate coniferous and deciduous forests, as well as tropical rain forests. Other radionuclides for which models have been proposed are Pu239 /Garten et al. 1978/, Tc99 /Garten 1987, Bostock 2004/, I129 and Cl36 /Bostock 2004/. These models have been parameterised for specific sites.

Table 2-1. Published models of radionuclide transfer in forest ecosystems.

Radio-nuclide	Forest type	Compartments	Source	Reference model name
Cs137	Liriodendron trees	Leaves, bark, roots, undercover, littermate, soil	Inoculated caesium	/Olson 1965/
Cs137	Deciduous	Tree, litter, lower soil, upper soil available, upper soil unavailable	Inoculated caesium	/Croom and Ragsdale 1980/
Cs137	Deciduous	Soil, roots, bole, branches, leaves, litter, understorey	Atmosphere	/Van Voris et al. 1990/ RADFORET
Cs137	Coniferous Deciduous	Tree, understorey, labile upper soil, fixed upper soil, deep soil	Atmosphere	/Schell et al. 1996/ FORESTPATH
Cs137	Coniferous	Bark, branches, needle 1, needle 2, understorey, fresh litter, litter, soil	Atmosphere	Described in /Shaw et al. 2003/ FORESTLIFE
Cs137	Coniferous Deciduous	Leaves (or needles of different age), wood, available xylem, unavailable xylem, litter, available organic soil, unavailable organic soil, fixed organic soil, available mineral soil, unavailable mineral soil, fixed mineral soil, understorey, mushrooms, game animals	Atmosphere	/Avila et al. 1999b/ FORESTLAND
Cs137	Coniferous Deciduous	Tree external, tree internal, litter, soil organic, soil mineral, understorey, mushrooms	Atmosphere	/Belli 2000/ RIFE
Cs137	Deciduous	Leaves, branches, bark, wood, distributive pool, large roots, small roots, soil	Atmosphere	/Mamikhin and Klyashtorin 1999/ ECORAD
Sr90 Mn54	Tropical rain forest	Canopy, litter, soil and wood	Atmosphere	/Jordan et al. 1973/
Sr90	Coniferous Deciduous	Upper litter, lower litter, soil, branches, wood, bark, leaves and herbs	Atmosphere	/Prohorov and Ginzburg 1973, Alexakhin et al. 1976, 1994/
Sr90	Deciduous	Soil, litter layer, tree wood, tree leaves, herbs	Groundwater	/Garten 1999/
Pu239	Deciduous	Soil, litter, soil fauna, roots, wood, leaves, ground vegetation, consumers	Soil	/Garten et al. 1978/
Tc99	Deciduous	Soil available pool, soil unavailable pool, roots, litter, wood, leaves	Groundwater	/Garten 1987/
Tc99 I129 Cl36	Coniferous	Organic soil (soil solution, fast sorption, slow sorption), mineral soil (soil solution, fast sorption, slow sorption), trunk wood, needle	Atmosphere Groundwater	/Bostock 2004/

2.1 Approaches to describe transfer

As mentioned above, most of the published models, with the exception of FORESTLIFE, are of the type known as compartment models. In these models the fluxes between compartments, corresponding to different forest components, are represented as the product of the radionuclide inventory in the compartment and the transfer rate coefficient. The models differ in the selection of compartments (see Table 2-1), as well as in the way the transfer rate coefficients, corresponding to different transfer processes, are formulated. Below the approaches used in the models to represent transfer processes are discussed.

Vertical redistribution of radionuclides in soil

Most of the models, except FORESTPATH, ECORAD and /Bostock 2004/, include a compartment corresponding to litter and use a rate constant as transfer rate coefficient from litter to soil. In FORESTPATH and /Bostock 2004/ the ground is divided into two soil layers and litter is included in the upper soil layer. Other models that divide the soil into an upper and a lower layer are: /Croom and Ragsdale 1980/, FORESTLAND and RIFE. All models that include more than one soil layer describe the transfer from the upper to the lower layer using a rate constant, obtained by model calibration using site-specific data. None of the models describe the upward transfer of radionuclides in soil. In the models with only one soil layer (see Table 2-1), the vertical transfer is not at all addressed, except for FORESTLIFE, which uses an advection dispersion equation to describe the downward movement of radionuclides in soil.

Leaching of radionuclides from the soil is either not at all considered, or a rate constant is used as transfer rate coefficient from soil out from the system. The only exception are the models of /Garten et al. 1978, Garten 1999, Bostock 2004/, which describe leaching from soil, using the approach described in /Baes and Sharp 1983/, as a function of precipitation, evapotranspiration and sorption in soil.

Soil-to-plant transfer

The majority of the models describe the transfer from soil to plant by multiplying the total radionuclide inventory in soil, or in a soil layer, by a transfer rate coefficient. Some of the models (/Croom and Ragsdale 1980/, FORESTPATH, FORESTLAND, /Garten 1987/ and /Bostock 2004/) further divide the soil into available (labile fraction or soil solution) and unavailable (fixed, absorbed) fractions and assume that the uptake rate by plants is proportional to the radionuclide inventory in the available fraction. The soil to plant transfer rate coefficients are commonly rate constants obtained by model calibration. Some of the models, RADFORET, FORESTLIFE, FORESTLAND, /Garten 1999/ and /Garten et al. 1978/ assume that the radionuclide transfer rate from soil to plant is proportional to biomass production.

Translocation in plants

Only the Pu239 model of /Garten et al. 1978/ and the Tc99 model of /Garten 1987/ include a compartment for roots. These models describe the translocation from roots to other plant parts with simple functions of biomass production. In other models translocation from roots is implicitly considered in the soil to plant transfer rate coefficient. The translocation between aboveground plant parts is commonly described with rate constants. FORESTLAND provides a more detailed description of translocation of Cs137 in trees using two compartments for the available and unavailable fractions in the xylem. This model also describes translocation between needles of different age and back from leaves (needles) to the xylem before leaves (needles) fall.

Transfer from plants to the soil-litter layer

All models describe the transfer from plants to the soil-litter layer in roughly the same way, using rate constants as transfer rate coefficients corresponding to weathering and leaf fall processes.

Transfer to fauna

In FORESTLAND the transfer of Cs137 to roe deer and moose is modelled with a simple kinetic model, assuming that the radionuclide assimilated in the gut is released from the organism at a constant rate, with a biological half-life of around 15 days for roe deer and 30 days for moose. In the Pu239 model of /Garten et al. 1978/ it is assumed that small mammals consume 33% of their biomass each day and that most of the Pu is present in the gut with a turnover time of 1 day. Other models do not include fauna.

3 Description of the model

3.1 Conceptual model

Figure 3-1 shows a schematic representation of the chosen conceptual model of radionuclide long-term transfer in a forest ecosystem. Biomass growth, precipitation and evapotranspiration drive the radionuclide cycling in the system by influencing the uptake of radionuclides by vegetation and their export from the system via runoff.

The boxes in Figure 3-1 correspond to different forest components (compartments) and the arrows to net radionuclide fluxes between compartments, including:

1. Flux from soil to tree wood via root uptake, F_{SToW}.
2. Flux from soil to tree leaves via root uptake, F_{SToL}.
3. Flux from soil to understorey (plants and mushrooms) via root (mycelia) uptake, F_{SToU}.
4. Flux from tree leaves to litter by leaf fall, F_{LToLi}.
5. Flux from tree wood to litter by wood fall, F_{WToLi}.
6. Flux from understorey plants to litter by plants senescence, F_{UToLi}.
7. Flux from litter to soil following litter decomposition, F_{LiToS}.

The dashed arrows correspond to inputs to the system (sources) and outputs (runoff) and the dashed lines indicate transfer processes (transfer from vegetation to fauna) that are not included in the mass balance of the system.

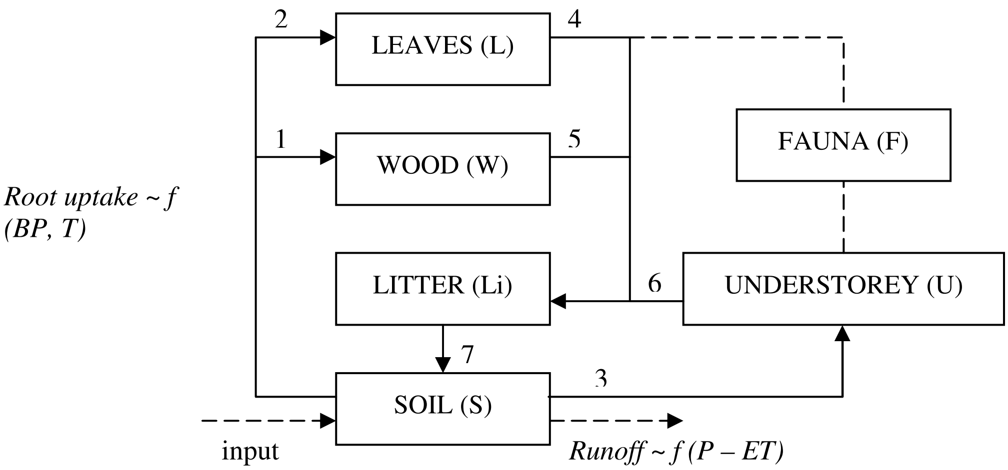

Figure 3-1. *Schematic representation of the adopted conceptual model of radionuclide transfer in a forest ecosystem (BP – biomass production, T – transpiration, P- precipitation, ET – evapotranspiration). The numbers correspond to fluxes between compartments (see text below).*

3.1.1 Assumptions and simplifications

The following are implicit assumptions and simplifications in the proposed conceptual model:

1. The radionuclides are homogeneously distributed in one soil layer, where all active roots are located. The model assumes that all radionuclides, nutrients and water are taken up by plants from this layer. This is a reasonably assumption in situations when most of the subsurface groundwater flow in the unsaturated soil zone occurs at shallow depths, or when a forest grows on a previously submerged land, for example on a former sediment of a lake. In situations with a deep groundwater table it is important to describe the vertical redistribution of radionuclides in soil. For this, it would be necessary to introduce several soil layers and consider the root distribution between layers.
2. Roots are not described explicitly, but implicitly as part of the soil compartment. Hence, roots are assumed to be in equilibrium with the soil. The radionuclides that are taken up by roots are readily transferred to the aboveground parts of the plant. These assumptions also imply that a constant radionuclide concentration is quickly achieved in the xylem sap.
3. The process of resuspension of radionuclides from soil is not described. It is assumed that the fluxes by soil resuspension have low significance in forest where a litter cover is always present.
4. Volatilisation of radionuclides is not considered in this version of the model, although it is recognised that this could be an important process for some elements, for example selenium or iodine.
5. The transfer from vegetation to litter by leaching processes is not included. It was considered that for long-term assessments when the contamination source is below ground, the fluxes via this transfer process are of less importance, comparing with fluxes by leaves fall and plant senescence.
6. The model assumes a constant intake rate of radionuclides by animals during their lifetime and a linear relationship between the concentration of radionuclides in the animal diet and the animal body.

Other more specific assumptions are given below in the description of the mathematical model.

3.2 Mathematical model

3.2.1 Production of vegetation biomass

The model uses average values of vegetation aboveground biomass and production that are characteristic of the Forsmark area (see Table 3-1). Hence, the forest growth is not modelled, but rather an averaged 50 years old tree is considered at any moment of time with a lifetime of 100 years, varying between 70 and 200 years.

3.2.2 Model of radionuclide transfer

The mathematical model of radionuclide transfer consists of a system of linear ordinary differential equations of the form:

$$\frac{dA_k^j}{dt} = F_{Out\,to\,k}^j - F_{k\,to\,Out}^j + \sum_i F_{i\,to\,k}^j - \sum_i F_{k\,to\,i}^j - \lambda^j * A_k^j$$

$$F_{i\,to\,k}^j = TC_{i\,to\,k}^j * A_i^j$$

$$F_{k\,to\,i}^j = TC_{k\,to\,i}^j * A_k^j \qquad (1)$$

$$F_{k\,to\,Out}^j = TC_{k\,to\,Out}^j * A_k^j$$

where,

A_k^j is the inventory of the ***j-th*** radionuclide in compartment ***k*** [Bq/m^2],

A_i^j is the inventory of the ***j-th*** radionuclide in compartment ***i*** [Bq/m^2],

$F_{Out\,to\,k}^j$ is the flux of the ***j-th*** radionuclide from outside the system (source) into compartment ***k*** [Bq/m^2/y],

$F_{k\,to\,Out}^j$ is the flux of the ***j-th*** radionuclide from compartment ***k*** out from the system [Bq/m^2/y],

$F_{i\,to\,k}^j$ is the flux of the ***j-th*** radionuclide from compartment ***i*** to compartment ***k*** [Bq/m^2/y],

$F_{k\,to\,i}^j$ is the flux of the ***j-th*** radionuclide from compartment ***k*** to compartment ***i*** [Bq/m^2/y],

$TC_{i\,to\,k}^j$ is the transfer rate coefficient of the ***j-th*** radionuclide from compartment ***i*** to compartment ***k*** [1/y],

$TC_{k\,to\,i}^j$ is the transfer rate coefficient of the ***j-th*** radionuclide from compartment ***k*** to compartment ***i*** [1/y],

$TC_{k\,to\,Out}^j$ is the transfer rate coefficient of the ***j-th*** radionuclide from compartment ***k*** out from the system [1/y],

λ^j is the decay rate coefficient of the ***j-th*** radionuclide [1/y].

Hence, the mathematical model of radionuclide transfer consists of a simple mass balance equation for each compartment, which accounts for fluxes into and out from the compartment and disintegration of the radionuclides. The fluxes between compartments are calculated by multiplying a transfer rate coefficient (*TC*) by the radionuclide inventory in the compartment. The model includes eight such *TC*s, which are calculated with the equations given below. Table 3-1 provides an overview of the compartments (state variables), fluxes and transfer rate coefficients in the model. The compartment's mushrooms and fauna are not included in the mass balance. The concentrations of radionuclides in these compartments are calculated from integrated values in other compartments as shown in epigraph 3.2.3.

Table 3-1. State variables (compartments), fluxes and transfer rate coefficients in the radionuclide transfer model.

Symbol	Description	Unit
State Variables		
A_S	Radionuclide inventory in the upper soil layer of depth h, assumed to be the rooting zone of the soil	Bq/m^2
A_{Li}	Radionuclide inventory in the litter layer above the soil	Bq/m^2
A_W	Radionuclide inventory in tree wood including living, dead wood and bark	Bq/m^2
A_L	Total radionuclide inventory in leaves including yearly and older leaves	Bq/m^2
A_U	Radionuclide inventory in above ground part of understorey plants	Bq/m^2
Fluxes		
F_{OutToS}	Radionuclide sub-surface flux into the soil	$Bq/m^2/y$
F_{SToW}	Radionuclide flux from soil to tree wood	$Bq/m^2/y$
F_{SToL}	Radionuclide flux from soil to tree leaves	$Bq/m^2/y$
F_{SToU}	Radionuclide flux from soil to understorey plants	$Bq/m^2/y$
F_{WToLi}	Radionuclide flux from tree wood to litter	$Bq/m^2/y$
F_{LToLi}	Radionuclide flux from tree leaves to litter	$Bq/m^2/y$
F_{UToLi}	Radionuclide flux from understorey plants to litter	$Bq/m^2/y$
F_{LiToS}	Radionuclide flux from litter to soil	$Bq/m^2/y$
F_{SToOut}	Radionuclide sub-surface flux from soil out of the system	$Bq/m^2/y$
Transfer Rate Coefficients		
TC_{SToW}	Transfer rate coefficient from soil to tree wood	1/y
TC_{SToL}	Transfer rate coefficient from soil to tree leaves	1/y
TC_{SToU}	Transfer rate coefficient from soil to understorey plants	1/y
TC_{WToLi}	Transfer rate coefficient from tree wood to litter	1/y
TC_{LToLi}	Transfer rate coefficient from tree leaves to litter	1/y
TC_{UToLi}	Transfer rate coefficient from understorey plants to litter	1/y
TC_{LiToS}	Transfer rate coefficient from litter to soil	1/y
TC_{SToOut}	Transfer rate coefficient from soil out from the system	1/y

Transfer Rate Coefficient from soil to tree wood, tree leaves and understorey plants (TC_{SToW}, TC_{SToL} and TC_{SToU})

The transfer rate coefficient of radionuclides from soil to wood, leaves and understorey plants is calculated with the following equations:

$$TC^{j}_{S\,To\,W}(t) = WP * \frac{CR^{j}_{W}}{\rho * h}$$
$$TC^{j}_{S\,To\,L}(t) = LP * \frac{CR^{j}_{L}}{\rho * h} \qquad (2)$$
$$TC^{j}_{S\,To\,U}(t) = UP * \frac{CR^{j}_{U}}{\rho * h}$$

where,

$TC^{j}_{S\,To\,W}(t)$ is the transfer rate coefficient of the ***j-th*** radionuclide from ***soil*** to ***tree wood*** [y-1]

$TC^j_{S\,To\,L}$ is the transfer rate coefficient of the ***j-th*** radionuclide from ***soil*** to ***tree leaves*** [y-1],

$TC^j_{S\,To\,U}$ is the transfer rate coefficient of the ***j-th*** radionuclide from ***soil*** to ***understorey plants*** [y-1],

WP is the yearly production of tree wood [kg/m^2/y dry weight],

LP is the yearly production of tree leaves [kg/m^2/y dry weight],

UP is the yearly production of understorey plants [kg/m^2/y dry weight],

CR^j_w is the concentration ratio of the ***j-th*** radionuclide from soil to tree wood [–],

CR^j_l is the concentration ratio of the ***j-th*** radionuclide from soil to tree leaves [–],

CR^j_U is the concentration ratio of the ***j-th*** radionuclide from soil to understorey plants [–],

ρ is the soil bulk density [kg dw/m^3],

h is the thickness of the soil rooting layer [m].

Apart from the assumptions outlined in epigraph 3.1.1 the above equations imply that there is a linear relationship with zero intercept between the total radionuclide concentration in soil and plants, which is not always the case /Sheppard and Evenden 1988/. Moreover, the radionuclide uptake rates are often proportional to the available fraction, rather than to the total radionuclide concentration in the soil. As a result, a large variation in the concentration factors is commonly observed. The above shortcomings can be partly overcome by using probability distributions, instead of single values, for the concentration ratios (CR). It has been also proposed to assume a negative correlation between the concentration ratios and the radionuclide distribution coefficients in soil /Sheppard and Evenden 1988, Sheppard and Sheppard 1989/, although the correlation coefficients show substantial variability /Alexakhin and Krouglov 2001/. Another problem with the use of *CR*s is that for many radionuclides of interest values are lacking. Some of the data gaps could be filled by performing site-specific measurements of the radionuclide concentrations, or the corresponding stable analogues, in soil and vegetation. However, it should be taken into account that *CR*s obtained for the stable analogue and the radionuclide may differ, for example if their availability for uptake by plants is different.

Transfer Rate Coefficient from vegetation to litter (TC_{WToLi}, TC_{LToLi} and TC_{UToLi})

The transfer rate coefficients from above-ground vegetation to litter are simple rate constants (in units of 1/y) equal to the yearly fractional loss of biomass from these compartments. Losses due to weathering are not considered. For simulation times greater than the lifetime of a tree, the level in wood and the flux from wood to litter would be overestimated if the limited lifetime of a tree is not taken into account. For this reason, the wood inventory is corrected as follows:

$$A^j_{W,corr}(t) = A^j_W(t) - A^j_W(t - T_{life}) * \exp\left[-\left(TC_{WToLi} + \lambda^j\right) * T_{life}\right] \qquad (3)$$

where,

$A^j_{W,corr}(t)$ is the corrected inventory of the ***j-th*** radionuclide in wood at time ***t*** [Bq/m^2],

$A^j_W(t)$ is the non-corrected inventory of the ***j-th*** radionuclide in wood at time ***t*** [Bq/m^2],

$A^j_W(t - T_{life})$ is the non-corrected inventory of the ***j-th*** radionuclide in wood at time $t\text{-}T_{life}$ [Bq/m^2],

TC_{WToLi} is the transfer rate coefficient from wood to litter [1/y],

λ^j is the decay rate coefficient of the ***j-th*** radionuclide [1/y].

The above correction is equivalent to assuming that the whole radionuclide inventory in wood is transferred to the soil-litter layer when the tree dies.

Transfer Rate Coefficient from litter to soil (TC_{LiToS})

The transfer rate coefficient from litter to soil is a simple rate constant (in units of 1/y) equal to the litter decomposition rate.

Transfer Rate Coefficient from soil out from the system (TC_{SToOut})

The transfer rate coefficient from soil out from the system is calculated with the following equation:

$$TC^j_{S\,To\,Out}(t) = \frac{P - ET}{h * (\theta + Kd^j * \rho)} \qquad (4)$$

where,

$TC^j_{S\,To\,Out}$ is the transfer rate coefficient of the ***j-th*** radionuclide from *soil* out from the system [y–1],

P is the area normalised precipitation rate [$m^3/m^2/y$],

ET is the area normalised evapotranspiration rate [$m^3/m^2/y$],

Θ is the volumetric water content in soil [m^3/m^3],

Kd^j is the distribution coefficient of the ***j-th*** radionuclide in soil [m^3/kg],

ρ is the soil bulk density [kg/m^3],

h is the thickness of the soil rooting layer [m].

The numerator in Equation 4 is the annual infiltration defined as the total amount of water that reaches surface receiving waters through both immediate surface runoff and sub-surface flow following infiltration of precipitation into the soil. The evapotranspiration is defined as the sum of the interception losses and transpiration:

$$ET = IL + T = P * I + T \qquad (5)$$

where,

ET is the area normalised evapotranspiration rate [$m^3/m^2/y$],

IL is the area normalised evaporation rate of intercepted precipitation [$m^3/m^2/y$],

I is the interception fraction [r.u.],

T is the area normalised transpiration rate [$m^3/m^2/y$],

P is the area normalised precipitation rate [$m^3/m^2/y$].

The annual interception loss is calculated (Equation 5) by multiplying the annual precipitation by a factor (I). The interception fraction (I) has been shown to be reasonably constant at a given annual precipitation value /Roberts 1983/ and exhibits a consistent decrease with increasing annual rainfall total.

The transpiration rate in Equation 5 comprises transpiration from trees, understorey vegetation and forest litter. As it will be explained in epigraph 3.3, the annual transpiration in temperate forests also varies within a narrow interval.

3.2.3 Calculation of radionuclide concentrations

The results of the simulations with the radionuclide transfer model can be used to calculate radionuclide concentrations in different environmental media as specified below.

Radionuclide concentrations in soil

The radionuclide concentrations in soil can be calculated with the following equation:

$$C_S^j = \frac{A_S^j}{\rho * h} \qquad (6)$$

where,

C_S^j is the concentration of the ***j-th*** radionuclide in soil [Bq/kg dw],

A_S^j is the inventory of the ***j-th*** radionuclide in soil [Bq/m^2],

ρ is the soil bulk density [kg dw/m^3],

h is the thickness of the soil rooting layer [m].

Radionuclide concentrations in vegetation

The radionuclide concentrations in tree wood, tree leaves and understorey plants are calculated with the following equations:

$$C_W^j = \frac{A_W^j}{M_W}$$
$$C_L^j = \frac{A_L^j}{M_L} \qquad (7)$$
$$C_U^j = \frac{A_U^j}{M_U}$$

where,

C_W^j is the concentration of the ***j-th*** radionuclide in tree wood [Bq/kg dw],

C_L^j is the concentration of the ***j-th*** radionuclide in tree leaves [Bq/kg dw],

C_U^j is the concentration of the ***j-th*** radionuclide in understorey plants [Bq/kg dw],

A_W^j is the inventory of the ***j-th*** radionuclide in tree wood [Bq/m^2],

A_L^j is the inventory of the ***j-th*** radionuclide in tree leaves [Bq/m^2],

A_U^j is the inventory of the ***j-th*** radionuclide in understorey plants [Bq/m^2],

M_W is the biomass of tree wood [kg dw/m^2],

M_L is the biomass of tree leaves [kg dw/m^2],

M_U is the aboveground biomass of understorey plants [kg dw/m^2].

Radionuclide concentrations in mushrooms

The radionuclide concentrations in mushrooms are calculated by multiplying the radio-nuclide concentrations in soil by a concentration ratio:

$$C_M^j = CR_M^j * C_S^j \qquad (8)$$

where,

C_M^j is the concentration of the ***j-th*** radionuclide in mushrooms [Bq/kg dw]

C_S^j is the concentration of the ***j-th*** radionuclide in soil [Bq/kg dw]

CR_M^j is the concentration ratio from soil to mushrooms of the ***j-th*** radionuclide [–]

Radionuclide concentrations in herbivores

The radionuclide concentration in herbivores is calculated with the help of an allometric relationship relating the radionuclide concentration in the animal diet with the radionuclide concentration in the animal body:

$$C_H^j = C_{diet}^j * CR_H^j$$

$$CR_H^j = f_H^j * a^j * W_H^{b^j} \qquad (9)$$

$$C_{diet}^j = \alpha_W * C_W^j + \alpha_L * C_L^j + \alpha_U * C_U^j + \alpha_M * C_M^j$$

where,

C_H^j is the concentration of the ***j-th*** radionuclide in the herbivore [Bq/kg fresh weight],

C_{diet}^j is the concentration of the ***j-th*** radionuclide in the diet of the herbivore [Bq/kg dry weight],

CR_H^j is the concentration ratio of the ***j-th*** radionuclide between the herbivore diet and the herbivore [Bq/kg fresh weight per Bq/kg dry weight],

C_W^j is the concentration of the ***j-th*** radionuclide in tree wood [Bq/kg dry weight],

C_L^j is the concentration of the ***j-th*** radionuclide in tree leaves [Bq/kg dry weight],

C_U^j is the concentration of the ***j-th*** radionuclide in understorey plants [Bq/kg dry weight],

C_M^j is the concentration of the ***-th*** radionuclide in mushrooms [Bq/kg weight],

α_W is the fraction of tree wood in the diet of the herbivore [–],

α_L is the fraction of tree leaves in the diet of the herbivore [–],

α_U is the fraction of understorey plants in the diet of the herbivore [–],

α_M is the fraction of mushrooms in the diet of the herbivore [–],

f_H^j is the gut uptake fraction of the ***j-th*** radionuclide [–],

a^j is the multiplier in the allometric relationship for the ***j-th*** radionuclide [in appropriate units],

b^j is the exponent in the allometric relationship for the ***j-th*** radionuclide [–].

The allometric relationship for the CR_H in Equation 9 was derived by assuming that:

1. The elimination rate of the radionuclide from the animal body is proportional to the total activity in the animal and an effective elimination rate.
2. The radionuclide concentration in the animal diet remains constant during its lifetime.
3. The dilution due to animal growth can be neglected.
4. Changes with age in the dry matter intake and biological half-life can be neglected.
5. The initial radionuclide concentration, i.e. at birth, in the animal is zero.

In this case the CR_H can be expressed as:

$$CR_H^j = \frac{C_H^j}{C_{diet}^j} = \frac{DMI * f_H^j * T_{eff}^j}{0.6931 * W_H} * (1 - e^{-0.6931*\frac{T_{life}}{T_{eff}^j}})$$

(10)

$$T_{eff}^j = \frac{0.6931}{\lambda_{eff}^j} = \frac{T_b^j * T_{rad}^j}{T_b^j + T_{rad}^j}$$

where,

CR_H^j is the concentration ratio of the ***j-th*** radionuclide between the herbivore diet and the herbivore [Bq/kg fresh weight per Bq/kg dry weight],

C_H^j is the concentration of the ***j-th*** radionuclide in the herbivore [Bq/kg fresh weight],

C_{diet}^j is the concentration of the ***j-th*** radionuclide in the diet of the herbivore [Bq/kg dry weight],

DMI is the daily dry matter intake by the herbivore [kg/d dry weight],

f_H^j is the gut uptake fraction of the ***j-th*** radionuclide for herbivores [–],

T_{rad}^j is the disintegration half time of the ***j-th*** radionuclide [d],

T_b^j is the biological half time of the ***j-th*** radionuclide [d],

W_H is the body weight of the herbivore [kg fresh weight],

T_{life} is the life time of the herbivore [d].

Finally, to obtain the coefficients ***a*** and ***b*** in Equation 9 the *DMI,* T_{life} *and* T_b in Equation 10 were substituted with the following allometric relationships:

$$DMI_H = a1 * W_H^{b1} \quad T_b^j = a2^j * W_H^{b2^j} \quad T_{life} = a3 * W_H^{b3}$$

3.3 Parameter values

The tables below show values for the model parameters compiled from the literature. A nominal value is given to each parameter, which was used in the example simulations that were carried out (results shown in epigraph 4). This value was selected to be as representative as possible of the environmental conditions and forest types in the Forsmark area. Whenever sufficient information was found for a parameter, a minimum and maximum value are also provided, which gives an idea of the variability and uncertainty associated with this parameter. Details regarding the choice of parameter values can be found in footnotes to the tables.

3.3.1. Nuclide independent parameters

The values for the nuclide independent parameters are given in Tables 3-2 to 3-4 and 3-6. These parameters measure environmental and ecological characteristics and vary from site to site and with the spatial scale considered. The parameters related to vegetation (Table 3-2), soil (Table 3-3) and fauna (Table 3-6) characteristics can be straightforwardly obtained from site investigations or estimated with simple models. In contrast, the parameters related to hydrological characteristics (Table 3-4), with the exception of the precipitation rate, are more difficult to obtain or estimate. These parameters are discussed below in more detail.

Hydrological parameters

The evapotranspiration is divided in two components: the evaporation of the intercepted water and transpiration processes occurring under wet and dry canopy conditions, respectively. There is a good physical understanding of the interception processes, which have been incorporated into detailed models, such as the "Rutter" model /Rutter et al. 1971/. Experimental studies and model assessments have shown that at least in temperate regions the structure of the forest plays only a minor role in determining the interception fraction when compared to the role played by the frequency and duration of storms /Roberts 1983 and references therein/. This has allowed estimating the interception fraction with acceptable accuracy by simply multiplying the annual precipitation value by a factor, commonly named interception fraction or interception loss ratio. This interception fraction /Calder and Newson 1979/ has been shown to be reasonably constant at a given annual precipitation value and exhibits a consistent decrease with increasing annual rainfall total. There are a considerable number of studies of rainfall interception in temperate deciduous and coniferous forests /see Baird and Wilby 1999 and references therein/. Because of the leafless nature of deciduous forest canopies in winter, the interception fraction of gross rainfall is usually substantially less than for evergreen conifers. Also, interception losses from temperate deciduous forests are far more variable at a given value of precipitation than for conifers. The interval of variation of the interception loss ratio given in Table 3-4 covers both deciduous and coniferous forests. A narrower interval could be proposed if differentiation is made between deciduous and coniferous forests.

Table 3-2. Values of model parameters related to vegetation characteristics.

Parameter	Units	Nominal	Min	Max	Comments
Yearly production of tree wood (WP)	$kg/m^2/y$ dry w.	0.18			(1)
Yearly production of tree leaves (LP)	$kg/m^2/y$ dry w.	0.08	0.05	1.7	(1)
Yearly production of understorey plants (UP)	$kg/m^2/y$ dry w.	0.08	0.02	0.25	(1)
Tree wood biomass (M_W)	kg/m^2 dry w.	5.1	2.2	55	(1) (2)
Tree leaves biomass (M_L)	kg/m^2 dry w.	0.5	0.2	7.0	(1) (2)
Understorey biomass (M_U)	kg/m^2 dry w.	0.08	0.02	0.25	(1) (3)
Yearly fractional loss of tree wood biomass (TC_{WToLi})	1/y	0.004			(4)
Yearly fractional loss of tree leaves biomass (TC_{LToLi}) – coniferous trees	1/y	0.25			(5)
Yearly fractional loss of tree leaves biomass (TC_{LToLi}) – deciduous trees	1/y	1			(6)
Yearly fractional loss of understorey plants biomass (TC_{UToLi})	1/y	1			(7)
Yearly fractional loss of litter biomass (TC_{LiToU})	1/y	0.16			(8)

(1) Nominal values are taken from /SKB 2004/. The values strongly depend on forest age and site-specific conditions.

(2) Minimun and maximun values, covering a wide range of European forests, reported in /Alriksson and Eriksson 1998, Ingerslev and Hallbäcken 1999, Nilsson and Albrektson 1993, Garten 1999, Belli 2000/.

(3) Minimum and maximum values reported in /Alriksson and Eriksson 1998, Garten 1999/.

(4) From /Garten 1999/.

(5) Assuming that needles have a turnover rate of 4 years.

(6) Assuming that all leaves fall each year.

(7) Assuming a yearly turnover of the understorey vegetation.

(8) From /Garten 1999/.

Table 3-3. Values of model parameters related to soil characteristics.

Parameter	Units	Nominal	Min	Max	Comments
Soil bulk density (ρ)	$kg\ dw/m^3$	1,180	700	1,500	(1)
Thickness of the soil rooting layer (h)	m	0.3	0.2	0.5	(2)
Volumetric water content in soil (θ)	m^3/m^3	0.20	0.1	0.5	(3)

(1) Minimum and maximum values based on values reported in /Wall and Heiskanen 2003/ and /Wästerlund 1985/ covering a wide range of forest soils of varying organic matter content and texture. The nominal value was chosen as representative of the Forsmark area /SKB 2004, Lindborg and Kautsky 2004/.

(2) Comprehensive reviews of maximum rooting depths /Canadell et al. 1996, Jackson et al. 1996/ have shown that the boreal forest has between 80–90% of its roots in the upper 20–50 cm of soil. Additionally, most woody vegetation has about 50% of roots in the upper 30 cm of soil.

(3) Minimum and maximum values obtained from studies covering a range of forest types /Alavi 2002, Heiskanen and Mäkitalo 2002, Paul et al. 2003/. The nominal value was chosen as representative of the Forsmark area /SKB 2004, Lindborg and Kautsky 2004/.

Table 3-4. Values of model parameters related to precipitation and evapotranspiration.

Parameter	Units	Nominal	Min	Max	Comments
Transpiration (T)	$m^3/m^2/y$	0.335	0.151	0.455	(1)
Interception fraction (I)	r.u.	0.3	0.1	0.5	(2)
Precipitation rate (P)	$m^3/m^2/y$	0.674	0.588	0.760	(3)

(1) The nominal value is the average value calculated using the values for coniferous forests given in Table 3-5. The minimum and maximum values are the corresponding ones from Table 3-5 considering all forest types.

(2) The interval of variation is based on minimum and maximum values reported in /Baird and Wilby 1999, Ladekarl et al. 2004, Murakami et al. 2000, Grelle et al. 1997, 1999, Perttu et al. 1980, Bringfelt 1982/. The nominal is the value reported in /Grelle et al. 1997/ for a coniferous forest in central Sweden.

(3) The interval of variation is based on minimum and maximum values observed in Forsmark, with higher precipitation in westerly parts and lower in the archipelago /Lindborg and Kautsky 2004/. The nominal value corresponds to the average precipitation at Eckarfjärden catchment reported in /Lindborg and Kautsky 2004/.

The transpiration component of evapotranspiration is influenced by many more factors, such as climate, forest age, species and structure and soil-moisture conditions. In addition, it is much more difficult to obtain data of forest transpiration. A wide range of techniques has, therefore, been used to measure the transpiration process depending on the temporal and spatial scales over which estimates are needed. Table 3-5 lists annual transpiration values from a wide range of studies involving both coniferous and deciduous forests and several different measurement techniques.

Table 3-5. Reported values of annual transpiration from different forest covers, T, including transpiration from trees, understorey and the forest litter.

Forest cover	Age (years)	Location	T (mm/year)	Reference
Ash	45	UK	407	/Roberts and Rosier 1994/
Ash	63	UK	294	/Roberts and Rosier 1994/
Beech	30–90	Belgium	344	/Schnock 1971/
Beech	64	UK	393	/Roberts and Rosier 1994/
Beech	–	France	288	/Chassagneux and Choisnel 1987/
Beech	100	Germany	283	/Kiese 1972/
Sweet chestnut	12	France	275	/Bobay 1990/
Oak (sessile)	18	Germany	342	/Brechtel 1976/
Oak (sessile)	54	Germany	298	/Brechtel 1976/
Oak (sessile)	165	Germany	342	/Brechtel 1976/
Oak	70	Denmark	293	/Rasmussen and Rasmussen 1984/
Oak	32	France	226 (151–301)	/Bréda et al. 1993/
Oak	120	France	288 (241–340)	/Nizinski and Saugier 1989/
Oak	150	Denmark	361 (295–455)	/Ladekarl et al. 2004/
Oak/Beech	100	Netherlands	289 (239–362)	/Bouten et al. 1992/
Cypress/cedar	72–75	Japan	333 (320–345)	/Murakami et al. 2000/
Cypress/cedar	4–7	Japan	294 (248–375)	/Murakami et al. 2000/
Norway spruce	70	Germany	362	/Tajchman 1971/
Norway spruce	–	Germany	279	/Brechtel 1976/
Norway spruce	–	UK	290	/Calder 1977/
Norway spruce	–	UK	340	/Calder 1977/
Norway spruce	–	UK	330	/Calder 1977/

Forest cover	Age (years)	Location	T (mm/year)	Reference
Sitka spruce	28	UK	340	/Law 1956/
Scots pine	–	Germany	327	/Brechtel 1976/
Scots pine	46	UK	353	/Gash and Stewart 1977/
Scots pine	–	UK	427	/Rutter 1968/
Norway spruce/ scots pine	100	Sweden	299	/Grelle et al. 1997/

The observed annual transpiration values are well below the potential rate of transpiration determined by climatic conditions. Furthermore, there is a close similarity in the annual transpiration values with an average value of 322 mm/y and a coefficient of variation of only 14%. Several factors have been suggested that can explain the low and similar transpiration rates /see Roberts 1983, Baird and Wilby 1999/:

(i) The existence of a strong negative correlation between air humidity deficit and stomatal and surface conductance. On days when evaporative demand is high stomata tend to close, whereas on days when demand is low the stomata are open. Consequently, daily transpiration rates remain conservative at below 4 mm/d, while transpiration rates from day to day are quite similar.

(ii) The contribution of understorey to the annual transpiration leads to equalization of transpiration between stands, when the lower transpiration rate of one tree species is a consequence of having less dense foliage, which permits greater levels of radiation to reach the forest floor. Higher radiation levels at the forest floor could lead to greater understorey transpiration because there would be more understorey plant growth and consequently more transpiration area; the light levels below the tree canopy tend to be in the range that still sets a limit to stomatal opening and therefore transpiration and the greater radiation levels below a dense tree canopy provide extra available energy for transpiration. To a lesser extent, this equalizing role can also be filled by the litter layer below trees. The storage capacity for water in litter may be high (up to 10 mm), but the energy available to promote evaporation is low, and therefore the litter evaporation is usually small (between 1 and 5% of gross rainfall).

(iii) Because of the likely modest daily transpiration rates in the studies in Table 3-5, it is probable that limiting soil moisture deficits are reached only rarely and therefore play a minor role in generating differences between sites.

Table 3-6. Values of model parameters related to fauna characteristics.

Parameter	Units	Nominal	Min	Max	Comments
Fraction of tree wood in the diet of moose (αLi)	%	1.6	0.7	2.4	(1) (3)
Fraction of tree wood in the diet of roe deer (αLi)	%	0.9	0.3	1.0	(2) (3)
Fraction of tree leaves in the diet of moose (αL)	%	54	38	55	(1) (3)
Fraction of tree leaves in the diet of roe deer (αL)	%	8.4	3.2	17.6	(2) (3)
Fraction of understorey plants in the diet of moose (αU)	%	43.5	28	47	(1) (3)
Fraction of understorey plants in the diet of roe deer (αU)	%	77	45	94	(2) (3)
Fraction of mushrooms in the diet of moose (αM)	%	0.9	0.25	1.1	(3) (4)
Fraction of mushrooms in the diet of roe deer (αM)	%	13.7	4.5	20	(3) (4)
Body weight of a moose (WH)	Kg FW	279	260	296	(5)
Body weight of a roe deer (WH)	Kg FW	21.3	18.6	24.0	(6)

(1) Based on values reported in /Cerderlund et al. 1980/ for August, September and October.

(2) Based on values reported in /Cerderlund et al. 1980/ for July, August and October.

(3) Nominal values were obtained by normalising the average values in the corresponding period by the sum of the contribution from all diet components.

(4) Based in values reported in /Avila 1998/ and /Avila et al. 1999a/.

(5) Based in values reported for the Forsmark area in /Lindborg and Kautsky 2004/.

(6) Based in values reported in /Lindborg and Kautsky 2004/. The maximum relates to adults and the minimum to calves.

Parameters related to fauna characteristics

The nuclide independent parameters related to fauna (herbivore) characteristics, and required by the model, are the weight of the animal and the relative contribution of different feeds to the diet (Table 3-6). Other needed radionuclide independent parameters, discussed in epigraph 3.3.2, are the total dry-matter intake and the life span since they were used to estimate the coefficients ***a*** and ***b*** in Equation 9. The values in Table 3-6 are given for roe deer and moose, which are among the most abundant herbivores in the Forsmark area and are commonly consumed by man. The values of the relative contribution of different feeds to the diet given in Table 3-6 are representative for the summer-autumn period, when most of the roe deer and moose are hunted.

3.3.2 Nuclide dependent parameters

The values for the nuclide dependent parameters are given in Tables 9 to 14. These are parameters related to processes responsible for differences among radionuclides in their distribution in forest compartments and in their degree of export from the system. It should be noted that in these tables a nominal value is provided for all parameters, even though for many of them no data were found in the literature. In general, there is a large disparity in the degree of confidence in the values, as shown in Table 3-7, where all nuclide-dependent parameters are classified by the degree of confidence in the values provided in Tables 3-8 to 3-13.

Table 3-7. Subjective classification of the nuclide-dependent parameters by the degree of confidence in the values given in Tables 3-8 to 3-13 (G- good, M-medium and P-poor).

Nuclide Am	CRU Table 3-8 M	CRL Table 3-9 P	CRW Table 3-10 P	CRM Table 3-11 P	Kd Table 3-12 M	a and b Table 3-13 M	fH Table 3-13 M
Cl	G	G	G	G	M	G	G
Cs	G	G	G	P	G	G	G
I	P	P	P	P	M	G	G
Ni	M	P	P	P	M	P	M
Np	M	P	P	P	M	P	P
Pu	M	P	P	P	M	G	P
Ra	M	P	P	P	M	M	M
Sr	G	G	G	M	M	M	M
Tc	M	P	P	P	M	M	M
Th	M	P	P	P	P	M	P
U	M	P	P	P	M	M	M

Three confidence categories were defined in Table 3-7:

(i) Good confidence – when sufficient empirical data, specific for forest ecosystems, were found and it was possible to identify and parameterise probability distributions.

(ii) Medium confidence – when empirical data were found, although not necessarily for forests. For this category it is deemed likely that the provided interval of variation will encompass the representative values for forests.

(iii) Poor confidence – when few or no data were found and the values (often only a nominal value) were estimated from other parameters.

Concentration ratios from soil to plants and mushrooms

Caesium is the only nuclide for which a good confidence is given to all values of concentration ratios. A good confidence is also given to strontium and chlorine concentration ratios for understorey plants, tree leaves and tree wood. For other nuclides the degree of confidence in the values for understorey plants was categorised as medium. It was considered that since the reported data cover a wide range of plant types and environmental conditions, it is likely that the values for understorey plants at a given site will fall within the provided interval. This holds for caesium, strontium and chlorine, for which values reported for forest plants fall within the range given for plants in /IAEA 1994/. Most of the values for tree leaves are either assumed equal to the values for understorey plants or derived from limited literature data. The situation for tree wood is similar, but the nominal values were set at one tenth of the values for tree leaves. This is based on the general observation that element concentrations in stems are 10% to 20% those in foliage /Perry 1994/.

Distribution coefficients

For most considered nuclides distribution coefficient values were generally found either for forest soils or for organic soils. The values show rather wide intervals of variation, which are likely to include values that are representative of boreal forests. For some elements, like caesium /Sanchez et al. 2002/ and uranium /Echevarria et al. 2001/, even quantitative relationships between soil properties and the distribution coefficients have been proposed. These relationships could be used to further reducing the uncertainty in the values.

Table 3-8. Concentration ratio of nuclides from soil to understorey plants (Bq/kg per Bq/kg dry weight).

Nuclide	Nominal	Minimum	Maximum	Comments
Am	1.3E–3	1.5E–7	7.7E–1	(10)
Cl	2.8E+1	3.0E+0	1.7E+2	(1)
Cs	7.0E+0	1.0E–1	1.0E+2	(2)
	2.3E+0	1.0E–2	2.4E+2	(3)
I	6.0E–1	1.0E–3	1.5E+0	(4)
Ni	1.3E–1	1.0E–2	4.7E+0	(5)
Np	7.0E–2	2.3E–5	5.7E–1	(10)
Pu	2.0E–3	5.0E–5	5.0E–2	(6)
Ra	2.7E+0	6.0E–1	7.6E+0	(7)
Sr	7.0E–1	2.0E–1	3.6E+0	(3)
	1.1E+0	6.0E–2	1.1E+2	(9)

Nuclide	Nominal	Minimum	Maximum	Comments
Tc	1.0E+0	5.0E–1	2.0E+1	(8)
Th	9.0E–2	3.0E–3	2.0E–1	(9)
U	1.4E–1	6.0E–3	7.5E–1	(9)

(1) The nominal value is the geometric mean calculated from values reported for native herbaceous plants in /Sheppard et al. 1999a/. The minimum and maximum are based on values reported in the same paper for native herbaceous plants.

(2) The nominal value is the geometric mean calculated from values reported for understorey plants in /Fesenko et al. 2001a/. The minimum and maximum values were obtained from values reported for shrubs in /Fesenko et al. 2001a, FASSET 2003 and references therein/.

(3) Based on values reported for grasses and herbs in /FASSET 2003 and references therein/. Values reported in /Yoshida and Muramatsu 1997/ fall within the given range.

(4) The nominal value is the geometric mean reported by /Robens et al. 1988/ for forage plants. The minimum and maximum values are based on values reported in /Schüttelkopf and Pimpl 1982, Robens et al. 1988/.

(5) The nominal value is the geometric mean calculated from values reported for blueberries over a wide range of soil types in /Sheppard and Evenden 1990/. The minimum and maximum are the 1 and 99 percentiles, respectively, obtained from values reported in /Denys et al. 2002/ and /Sheppard and Evenden 1990/ assuming a lognormal distribution. Values reported in /Yoshida and Muramatsu 1997/ fall within this range.

(6) The nominal value is the value reported in /Garten et al. 1978/ and the minimum and maximum values are based on values reported in /IAEA 1994/ for all types of plants.

(7) Based on values reported for shrubs in /FASSET 2003 and references therein/.

(8) The nominal value is the value recommended by /IAEA 1994/ for pastures. The minimum and maximum values are those reported in /Garten et al. 1986/.

(9) The nominal value is the value given in /FASSET 2003/ for shrubs. The mimimun and maximum values are based on values reported in /FASSET 2003/ and in /Yoshida and Muramatsu 1997/.

(10) The nominal value is the best estimate value suggested in /IAEA 1994/ for grasses and the minimum and maximum values are based on values reported in the same reference for all types of plants.

Table 3-9. Concentration ratio of nuclides from soil to tree leaves (Bq/kg per Bq/kg dry weight).

Nuclide	Nominal	Minimum	Maximum	Comments
Am	1.3E–3			(10)
Cl	1.0E+1	8.0E–1	2.8E+1	(1)
Cs	3.4E+0	8.0E–1	1.4E+1	(2)
	5.8E+0	4.0E–1	9.1E+2	(3)
I	6.0E–1	1.5E–1	2.3E+0	(7) (10)
Ni	1.3E–1	9.0E–2	2.3E–1	(9) (10)
Np	7.0E–2			(10)
Pu	4.6E–5			(4)
Ra	2.7E+0			(10)
Sr	5.0E–1	3.0E–2	2.4E+1	(5)
	7.0E+0	3.0E–1	1.1E+1	(6)
Tc	1.0E+0			(10)
Th	9.0E–2	1.0E–3	1.7E–2	(9) (10)
U	1.4E–1	3.0E–3	1.0E+0	(8) (9) (10)

(1) The nominal value is the geometric mean calculated from values reported for leaves of native woody plants in /Sheppard et al. 1999a/. The minimum and maximum are based on values reported in the same paper for leaves of native woody plants.

(2) The nominal value is the geometric mean of the values reported in /Fesenko et al. 2001b/ for needles of all ages. The minimum and maximum values are the 1 and 99 percentiles, respectively, obtained by assuming a lognormal distribution with a geometric standard deviation (GSD) of 1.9 derived from reported values for needles in the same reference.

(3) The values were derived in the same way as for coniferous trees, comment (2), but using data for deciduous trees reported in the same reference. The GSD for leaves of deciduous trees derived from these data is 3.3.

(4) The nominal value is the value reported in /Garten et al. 1978/.

(5) The nominal value is the average value reported in /Shcheglov et al. 2001/ for pine trees. The minimum and maximum values are those reported in the same reference for coniferous trees.

(6) The nominal value is the average value reported in /Shcheglov et al. 2001/ for birch. The minimum and maximum are the minimum and maximum values reported in the same reference for deciduous trees.

(7) Minimum and maximum values based on values reported in /Baes et al. 1984, Coughtrey et al. 1985, Sheard 1985, Sheppard and Evenden 1990/.

(8) Based on values reported in /Baes et al. 1984/.

(9) Based on values reported in /Yoshida and Muramatsu 1997/.

(10) Due to the lack or insufficiency of the data, the nominal values were assumed equal to those given in Table 3-8 for understorey plants.

Table 3-10. Concentration ratio of nuclides from soil to tree wood (Bq/kg per Bq/kg dry weight).

Nuclide	Nominal	Minimum	Maximum	Comments
Am	1.3E–4			(7)
Cl	3.0E+0	8.0E–1	1.1E+1	(1)
Cs	8.0E–1	1.0E–1	5.8E+0	(2)
	7.0E–1	2.0E–1	2.5E+0	(3)
I	6.0E–2			(7)
Ni	1.3E–2			(7)
Np	7.0E–3			(7)
Pu	4.6E–5			(4)
Ra	2.7E–1			(7)
Sr	2.0E–1	1.0E–2	1.2E+0	(5)
	8.0E–1	7.0E–3	4.3E+0	(6)
Tc	1.0E–1			(7)
Th	9.0E–3			(7)
U	1.4E–2			(7)

(1) The nominal value is the geometric mean calculated from values reported for stems of native woody plants in /Sheppard et al. 1999a/. The minimum and maximum are based on values reported in the same paper for stem of native woody plants.

(2) The nominal value is the geometric mean of the values reported in /Fesenko et al. 2001b/ for wood of coniferous trees. The minimum and maximum values are the 1 and 99 percentiles, respectively, obtained by assuming a lognormal distribution with a geometric standard deviation (GSD) of 2.3 derived from reported values for wood in the same reference.

(3) The values were derived in the same way as for coniferous trees, comment (2), but using data for deciduous trees reported in the same reference. The GSD for wood of deciduous trees derived from these data is 1.7.

(4) The nominal value is the one reported in /Garten et al. 1978/.

(5) The nominal value is the average value reported in /Shcheglov et al. 2001/ for pine. The minimum and maximum are those values reported in the same reference for coniferous trees.

(6) The nominal value is the average value reported in /Shcheglov et al. 2001/ for birch. The minimum and maximum are those values reported in the same reference for deciduous trees.

(7) Due to lack or insufficient data, the nominal values were assumed equal to one tenth of the values given in Table 3-8 for understorey plants.

Table 3-11. Concentration ratio of nuclides from soil to mushrooms (Bq/kg per Bq/kg dry weight).

Nuclide	Nominal	Minimum	Maximum	Comments
Am	1.3E–3			(3)
Cl	2.8E+1			(3)
Cs	1.2E+2	2.7E–1	6.2E+2	(2)
I	6.0E–1			(3)
Ni	1.3E–1	1.0E–1	2.0E–1	(1) (3)
Np	7.0E–2			(3)
Pu	2.0E–3			(3)
Ra	2.7E+0			(3)
Sr	7.0E–1	1.0E–2	5.0E–2	(1) (3)
Tc	1.0E+0			(3)
Th	9.0E–2	2.0E–3	1.1E–2	(1) (3)
U	1.4E–1	1.1E–2	3.4E–2	(1) (3)

(1) Based on values reported in /Yoshida and Muramatsu 1997/.

(2) The nominal value is the geometric mean of the values reported in /Fesenko et al. 2001a/. The minimum and maximum values are based on values reported in /Yoshida and Muramatsu 1997, Fesenko et al. 2001a/.

(3) Due to the lack or insufficiency of the data, the nominal values were assumed equal to those given in Table 3-8 for understorey plants.

Table 3-12. Distribution coefficients (K_d) of different nuclides in soil (m^3/kg).

Nuclide	Nominal	Minimum	Maximum	Comments
Am	1.0E+2	1.0E+1	1.0E+3	(3)
Cl	1.0E–2	1.0E–3	1.0E–1	(4)
Cs	8.0E–1	4.0E–1	5.3E+1	(1)
I	3.0E–2	8.0E–4	3.0E–1	(2)
Ni	1.0E+0	2.0 E–1	7.0E+0	(3)
Np	1.0E+0	5.0E–1	3.0E+0	(3)
Pu	2.0E+0	2.0E–1	2.0E+2	(3)
Ra	2.0E+0	2.0E–1	2.0E+1	(3)
Sr	2.0E–1	4.0E–3	6.0E+1	(3)
Tc	3.0E–3	4.5E–5	7.0E–2	(6)
Th	1.0E+1	1.0E+0	1.0E+2	(5)
U	4.0E–1	2.0E–2	4.0E+0	(7)

(1) Based on values reported in /Sanchez et al. 2002/ for soils with organic matter content above 20%.

(2) The nominal value is the best estimate value given in /IAEA 1994/ for organic soils. The minimum and maximum values are based on values reported in /Sheppard and Thibault 1991, IAEA 1994, Sheppard 2003/.

(3) Based on values reported in /IAEA 1994/ for organic soils.

(4) The nominal value is the geometric mean reported in /Sheppard et al. 1996/ for organic soils. The minimum and maximum values are based on values reported in /IAEA 1994/ for organic soils.

(5) Based on values reported in /Jiskra 1985/.

(6) The nominal value is the geometric mean of values reported in /Sheppard et al. 1990/ for 27 organic soils. The minimum and maximum values are based on values reported in /Sheppard et al. 1990/ and /IAEA 1994/ for organic soils.

(7) The nominal value is the best estimate value given in /IAEA 1994/ for organic soils. The minimum and maximum values are based on values reported in /IAEA 1994, Echevarria et al. 2001/.

Table 3-13. Parameters related to fauna characteristics: *a* is the multiplier in the allometric relationship for the concentration ratio diet-animal body (in appropriated units), *b* is the exponent in the allometric relationship for the concentration ratio diet-animal body (r.u.) and f_H is the gut uptake fraction (r.u.).

Nuclide	a	b	fH	Comments
Am	2.0E+1	1.4E–1	5.0E–4	(1)
Cl	2.1E–1	1.1E–2	1.0E+0	(2)
Cs	1.2E+0	0.0E+0	1.0E+0	(1)
I	1.4E+0	–1.1E–1	1.0E+0	(3)
Ni	2.2E+1	1.1E–1	5.0E–2	(4)
Np	2.2E+1	1.1E–1	1.0E–3	(4)
Pu	2.0E+1	1.4E–1	5.0E–4	(1)
Ra	1.4E+1	8.0E–2	2.0E–1	(5)
Sr	1.8E+1	8.2E–2	2.0E–1	(1)
Tc	4.2E–1	1.6E–1	1.0E–1	(2)
Th	1.8E+1	1.3E–1	2.0E–4	(5)
U	4.8E–1	4.1E–2	5.0E–2	(5)

The values of ***a*** and ***b*** were obtained as described in epigraph 3.2.3.

All comments below refer to f_H

(1) /Coughtrey et al. 1985, Beresford et al. 2000/.

(2) /Bishop et al. 1989/.

(3) /Beresford et al. 2000/.

(4) /Coughtrey et al. 1985/.

(5) /FASSET 2003/.

Parameters related to fauna characteristics

The coefficients ***a*** and ***b*** of the allometric relationship, included in the expression for the radionuclide concentration ratio between the animal body and its diet (CR_H), were calculated using the following allometric coefficients (see 3.2.3): for the dry matter intake, in kg/d (6.6E–2 for the multiplicand and 6.3E–1 for the exponent, /Nagy 2001/); for animal lifetime, in days (3.7E+2 the for the multiplicand and 3.5E–1 for the exponent, /Calder 1984/); and for the biological half-life the values in Table 3-14. In the case of Np and Ni there were no data of biological half-lifes. For Np the values of ***a*** and ***b*** were assumed equal to the values for Pu, while for Ni these were obtained by assuming an infinite half-life, which is a conservative assumption.

The available data of gut uptake fractions were very limited, although for readily assimilated nuclides as Cs, Cl and I this parameter will not vary substantially. For poorly assimilated nuclides a higher variability can be expected, for example values between 2E–6 and 2E–2 have been reported for Pu /Coughtrey and Thorne 1983/.

Table 3-14. Multiplier (a3) and Exponent (b3) in the allometric relationship of the biological half- life, in days, of different elements in the animal body.

Nuclide	a3	b3	Comments
Am	1.1E+3	7.3E–1	(1)
Cl	2.4E+0	2.5E–1	(2)
Cs	1.3E+1	2.4E–1	(3)
I	1.7E+1	1.3E–1	(4)
Pu	1.1E+3	7.3E–1	(1)
Ra	2.8E+2	2.8E–1	(5)
Sr	6.4E+2	2.6E–1	(4)
Tc	4.8E+0	4.0E–1	(6)
Th	8.9E+2	8.0E–1	(5)
U	5.5E+0	2.8E–1	(5)

(1) From /FASSET 2003/ derived from data presented in /Coughtrey and Thorne 1983, ICRP 1979/.

(2) From /FASSET 2003/ derived from data presented in /Coughtrey and Thorne 1983, Bishop et al. 1989/.

(3) /Beresford et al. 2003/.

(4) /Highley et al. 2003/.

(5) /FASSET 2003/.

(6) /USDoE 2002/.

4 Examples of model simulations

The model was implemented in Matlab/Simulink /MATHWORKS 2003/. A simulation was run for a constant radionuclide input of 1 Bq/m^2/y into the soil layer with all parameters set at nominal values. The Simulink numerical solver ode15 was used for integration of the model. The values of radionuclide concentrations in different compartments at year 10000 after the start of the simulation are shown in Table 4-1. These values can be used to calculate doses to both man and biota.

The results of the model calculations indicate that the time needed for achieving a constant concentration in soil, as well as the equilibrium concentration, vary among the considered radionuclides. The higher the distribution coefficient (Kd) is, the longer the time needed to achieve equilibrium and the higher the equilibrium concentrations. This is illustrated in Figure 4-1 where model predictions are shown for long-lived radionuclides with contrasting Kd values varying from 3.0E–3 to 1.0E+1. Since the model assumes proportionality between the concentrations of radionuclides in soil and other compartments, the same pattern is observed for the radionuclide concentrations in understorey plants, tree leaves, tree wood, mushrooms and herbivores.

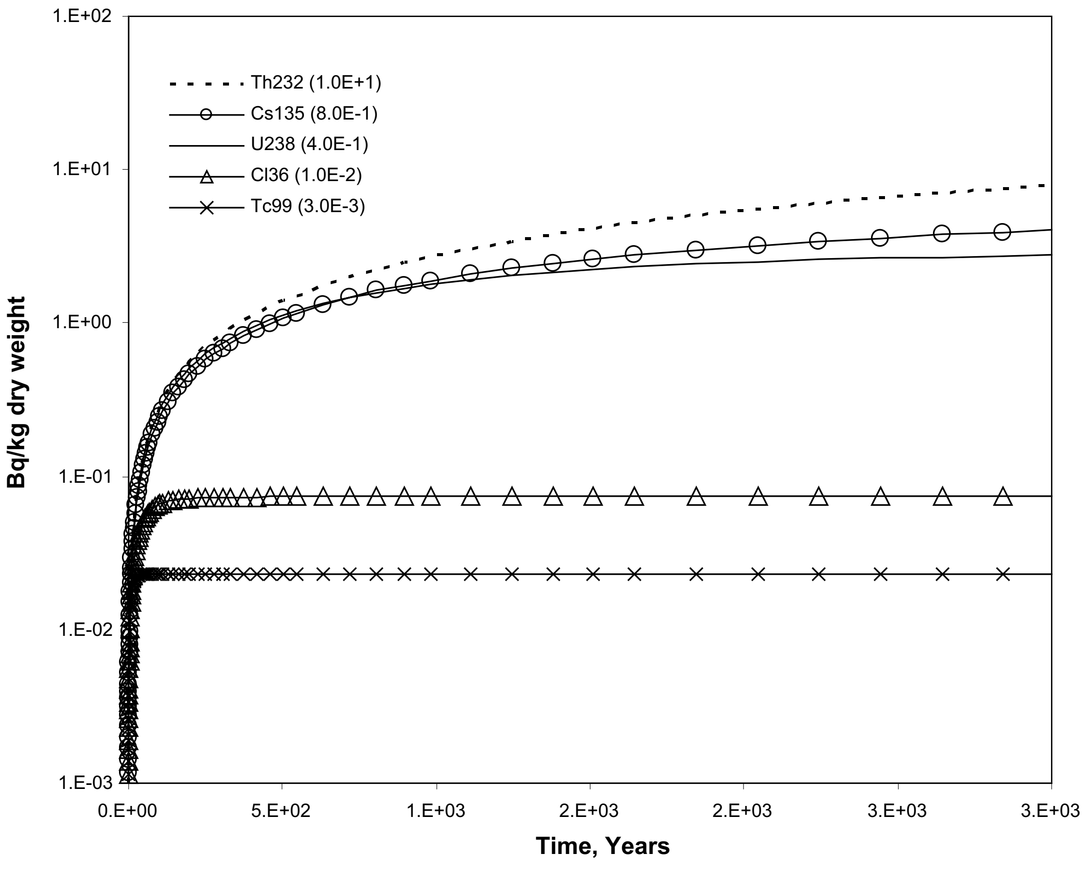

Figure 4-1. *Predicted activity concentrations in soil (Bq/kg dry weight) for radionuclides with contrasting distribution coefficients (value shown within parentheses in the legend) for a constant input into soil of 1 Bq/m*^{2}*/y.*

Table 4-1. Predicted activity concentration in different forest compartments at year 10000 after the start of a continuous input into the soil of 1 Bq/m²/y. Values are given in Bq/kg dry weight for all compartments except for roe deer and moose, which values are given in Bq/kg fresh weight.

	Soil	Understorey plants	Leaves	Wood	Mushrooms	Roe deer	Moose
Am241	1.7E+00	2.3E–03	1.4E–03	6.1E–04	2.3E–03	3.2E–05	3.7E–05
Cl36	7.4E–02	2.1E+00	4.8E–01	6.5E–01	2.1E+00	4.1E–01	2.6E–01
Cs135	5.8E+00	4.0E+01	1.2E+01	1.3E+01	6.9E+02	1.5E+02	3.5E+01
Cs137	1.2E–01	7.9E–01	2.3E–01	1.1E–01	1.4E+01	2.9E+00	6.9E–01
I129	2.2E–01	1.3E–01	8.5E–02	3.8E–02	1.3E–01	1.3E–01	8.2E–02
Ni59	7.0E+00	9.1E–01	5.8E–01	2.6E–01	9.1E–01	1.4E+00	1.5E+00
Np237	7.2E+00	5.0E–01	3.2E–01	1.5E–01	5.0E–01	1.5E–02	1.6E–02
Pu239	1.1E+01	2.3E–02	3.3E–04	1.5E–03	2.3E–02	3.1E–04	2.2E–04
Pu242	1.2E+01	2.5E–02	3.7E–04	1.7E–03	2.5E–02	3.4E–04	2.4E–04
Ra226	4.4E+00	1.2E+01	7.5E+00	3.4E+00	1.2E+01	4.2E+01	4.3E+01
Sr90	1.1E–01	7.4E–02	3.2E–02	2.6E–02	7.6E–02	3.3E–01	2.9E–01
Tc99	2.3E–02	2.3E–02	1.5E–02	6.7E–03	2.3E–02	1.5E–03	1.9E–03
Th232	2.3E+01	2.1E+00	1.3E+00	6.1E–01	2.1E+00	1.1E–02	1.3E–02
U238	2.9E+00	4.1E–01	2.6E–01	1.2E–01	4.1E–01	1.1E–02	9.8E–03

The radionuclide concentrations in the soil solution show a different pattern (Figure 4-2). As for the total concentrations, for radionuclides with high Kd it does take longer time to achieve an equilibrium value. However, the soil solution concentrations of all radionuclides converge to the same value. This means that the model predicts that concentrations at equilibrium are independent of the Kd values. The same observation was made by /Sheppard et al. 1999b/ in studies of heavy metals emissions from the mining industry.

In order to test the model, predictions of the transfer factor (TF) from soil to herbivores (expressed in Bq/kg fresh weight per Bq/kg dry weight) were compared with empirical values found in the literature /IAEA 1994, FASSET 2003/. The virtues of this test reside in that predictions are compared with empirical values that are not used for the model parameterisation and predicted levels in herbivores are influenced by all modelled processes, since in this model herbivores are at the end of the food chain. As a limitation, it can be mentioned that it does not permit testing the predictions of soil concentrations. For Cs, and Sr the predicted *TF*s were within the range of empirical observations (see Table 4-2). The model predictions were slightly higher for Ra and U and slightly lower for Th. It should be, however, noted that only few data were available for these three elements (49 values for Ra, 8 for Th and only 4 for U).

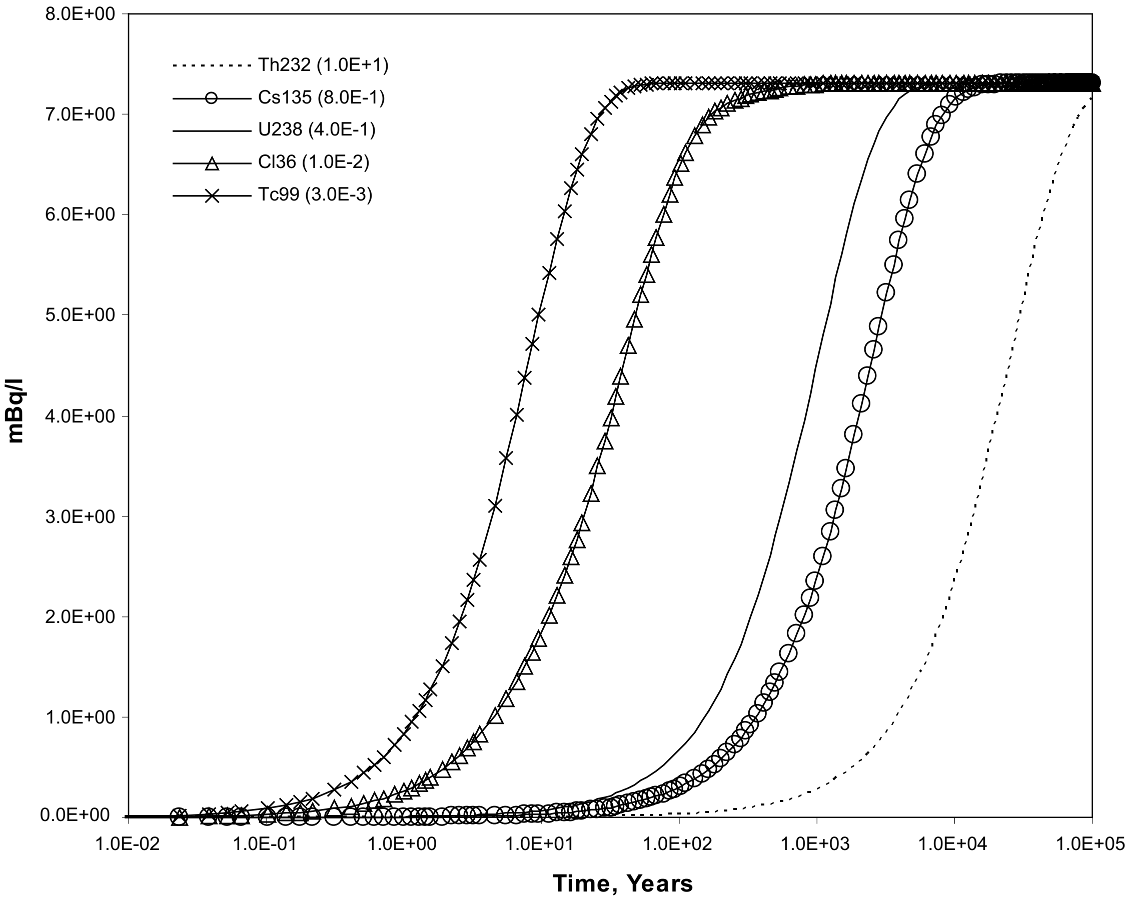

Figure 4-2. *Predicted activity concentrations in the soil solution (mBq/l) for radionuclides with contrasting distribution coefficients (value shown within parentheses in the legend) for a constant input in soil of 1 Bq/m²/y.*

Table 4-2. Model predictions and empirical values found in the literature /FASSET 2003, IAEA 1994/ of Transfer Factors to herbivores (expressed in Bq/kg fresh weight in the herbivore per Bq/kg dry weight in soil).

Nuclide	**Prediction Roe deer**	**Prediction Moose**	**FASSET* Herbivores**	**IAEA* Roe deer**	**IAEA* Moose**
Cs	2.5E+1	6.1E+0	1.9E–2 1.4E+2	3.5E–1 7.1E+1	2.1E+00 1.1E+01
Ra	9.6E+0	9.7E+0	2.1E–3 2.0E–1		
Sr	3.0E+0	2.7E+0	4.5E–3 1.4E+1		
Th	4.7E–4	5.4E–4	2.1E–3 4.7E–1		
U	3.7E–3	3.4E–3	1.2E–4 2.8E–3		

* Minimum and maximum values shown

5 Discussion

The radionuclide independent processes, and their corresponding parameters in the model, can relatively easily be obtained from the on-going site investigation programme. The discussion below will therefore mainly focus on the radionuclide-dependent processes and parameters.

5.1 Soil to plant transfer

The soil to plant concentration ratio is one of the model parameters for which there are substantial data gaps and therefore poor confidence in the values. Some of the data gaps could be filled with data obtained from on-going site investigations. However, the experimental data will always be representative for a limited set of environmental conditions. At the same time, taking into account that the model will be used for very long-term predictions, it is important that it can deal with potential environmental changes. Hence, it is desirable to describe the plant uptake processes in a mechanistic way that relies on parameters for which it is possible to assess the potential variability. Below we will discuss two approaches to solve this problem.

The first approach is based on the assumption that some elements are taken-up passively with the transpiration flux. For them the total flux from soil to plants can be expressed as:

$$
\begin{aligned}
F^{j}_{S\,To\,Tree}(t) &= \sigma^{j} * T_{tree} * C^{j}_{pw} \\
F^{j}_{S\,To\,U}(t) &= \sigma^{j} * T_{U} * C^{j}_{pw}
\end{aligned}
\qquad (11)
$$

where,

$F^{j}_{S\,To\,Tree}(t)$ is the flux of the ***j-th*** radionuclide from ***soil*** to the whole ***tree*** including roots [Bq/m^2/y]

$F^{j}_{S\,To\,U}(t)$ is the flux of the ***j-th*** radionuclide from ***soil*** to ***understorey plants*** including roots [Bq/m^2/y]

σ^{j} is the permeability of the root to the ***j-th*** radionuclide [–]

T_{tree} is the transpiration rate of trees [m^3/m^2/y]

T_U is the transpiration rate of understorey plants [m^3/m^2/y]

C^{j}_{pw} is the concentration of the ***j-th*** radionuclide in the soil pore water [Bq/m^3].

By dividing the above fluxes by the total production of trees and understorey plants, and using a simple linear model (such as Kd) to express the concentration in pore water as a function of the total concentration in soil, the following expression for the soil-to-plant concentration ratios can be obtained:

$$CR_{Tree}^{j} = \sigma^{j} * \frac{T_{tree}}{TP} * \frac{\theta}{Kd^{j} * \rho + \theta}$$
$$CR_{TU}^{j} = \sigma^{j} * \frac{T_{U}}{TUP} * \frac{\theta}{Kd^{j} * \rho + \theta} \qquad (12)$$

where,

CR_{Tree}^{j} is the total concentration ratio from soil to trees of the ***j-th*** radionuclide [–],

CR_{TU}^{j} is the total concentration ratio from soil to understorey plants of the ***j-th*** radionuclide [–],

σ^{j} is the permeability of roots to the ***j-th*** radionuclide [–],

T_{tree} is the transpiration rate of trees [$m^3/m^2/y$],

T_U is the transpiration rate of understorey plants [$m^3/m^2/y$],

TP is the total biomass production of trees [$kg/m^2/y$],

TUP is the total biomass production of understorey plants [$kg/m^2/y$],

Θ is volumetric water content in soil [m^3/m^3],

Kd^{j} is the distribution coefficient of the ***j-th*** radionuclide in soil [m^3/kg],

ρ is the soil bulk density [kg/m^3].

Note that the *CR*s and biomass production in Equation 12 refer to the whole plant, including roots, while in Equation 2 they refer to specific aboveground parts of the plants. The *CR* for specific plant parts can be obtained from the total *CR* if the ratios between *CR*s and biomass of different plant parts are known. For example the *CR* for tree leaves can be estimated from the total *CR*s as follows:

$$CR_{L}^{j} = \frac{CR_{Tree}^{j}}{\frac{M_L}{M_{Tree}} + \frac{M_W}{M_{Tree}} * \frac{CR_W^{j}}{CR_L^{j}} + \frac{M_R}{M_{Tree}} * \frac{CR_R^{j}}{CR_L^{j}}} \qquad (13)$$

where,

CR_{L}^{j} is the concentration ratio from soil to tree leaves of the ***j-th*** nuclide [–],

CR_{Tree}^{j} is the total concentration ratio from soil to trees of the ***j-th*** nuclide [–],

CR_{W}^{j} is the concentration ratio from soil to tree wood of the ***j-th*** nuclide [–],

CR_{R}^{j} is the concentration ratio from soil to tree wood of the ***j-th*** nuclide [–],

M_L is the biomass of tree leaves [kg/m^2],

M_W is the biomass of tree wood [kg/m^2],

M_R is the biomass of tree roots [kg/m^2],

M_{Tree} is the total biomass of trees [kg/m^2].

It should be noted that to apply Equation 13 it is not necessary to know the values of the specific *CR*s, but it is sufficient to know the ratios between them, which are less variable. Equally, absolute values of biomass of different plant parts are not needed, but rather their contribution to the total biomass.

The concentration ratios from soil to understorey plants of Cl, Cs, I, Ni, Pu, Ra, Sr and Tc, as well as the concentration ratios from soil to tree leaves and tree wood of Cl, Cs and Pu, were estimated by performing probabilistic simulations with Equations 12 and 13. It was assumed that roots have the same permeability for radionuclides and water, i.e. the permeability coefficient σ was set equal to one for all radionuclides considered. The values and distributions assigned to other parameters are detailed below:

- A probability distribution was obtained for the transpiration rate of understorey plants by multiplying the total transpiration rate (T) by a coefficient representing the contribution of understorey to the total transpiration. A triangular distribution was assigned to the total transpiration rate with mode, minimum and maximum values equal to the nominal, minimum and maximum values, respectively, given in Table 3-4. The contribution of the understorey to the total transpiration was also assigned a triangular distribution with a mode of 0.15 (value reported in /Grelle et al. 1997/ for a coniferous forest in central Sweden) and minimum and maximum values of 0.06 and 0.65 /Black and Kelliher, 1989/, respectively.
- The values of total biomass production and the contribution of different parts to the total biomass were derived from data in /Garten et al. 1978/.

  Total biomass production of trees (*TP)*: 1.6 $kg/m^2/y$.

  Total biomass production of understorey plants (*TUP)*: 0.22 $kg/m^2/y$.

  Contribution of leaves to total tree biomass (M_L/M_{Tree}): 0.03.

  Contribution of wood to total tree biomass (M_W/M_{Tree}): 0.75.

  Contribution of roots to total tree biomass (M_W/M_{Tree}): 0.22.

  Contribution of aboveground parts to total understorey biomass: 0.5.

  Contribution of roots to total understorey biomass: 0.5.
- Triangular probability distributions were assigned to the soil water content (θ), the soil density (ρ) and distribution coefficients (Kd) with mode, minimum and maximum values equal to the nominal, minimum and maximum values, respectively, given in Tables 3-3 and 3-11.
- From data reported in /Garten et al. 1978/ the ratio of Pu *CR*s between tree wood and tree leaves and between tree roots and tree leaves were estimated as 1 and 1300, respectively. The ratio of Pu *CR*s between understorey roots and aboveground parts were estimated as 27. For Cl the ratio of *CR*s between tree wood and tree leaves were estimated as 0.3 from data reported in /Sheppard et al. 1999a/. For Cs the ratio of *CR*s between tree wood and tree leaves, 0.12, reported in /Fesenko et al. 2001a/ was used. All other ratios of *CR*s were assumed to be equal to 1.

The calculated concentration ratios for different plant parts are presented in Table 5-1. In Figure 5-1 the calculated values for understorey plants are compared with values reported in the literature (see Table 3-8). The intervals of the calculated and reported *CR*s overlap for all nuclides, although with a varying degree of agreement. It should be taken into account, that the literature data used in the comparison were obtained at different sites and using different methods. A better agreement could be achieved if the model parameters and the empirical data of *CR*s are obtained for the same site. Hence, it can be concluded that this approach has good perspectives, in particular for radionuclides that are not analogues of plant nutrients, such as the actinides, and also some analogues of plant micronutrients, for example Cl36 and Ni59.

Table 5-1. Calculated values of the concentration ratios from soil to understorey plants CR_U), to tree leaves (CR_L) and tree wood (CR_W) expressed in units of Bq/kg per Bq/kg dry weight.

Nuclide	CRU	CRL	CRW
Cl	1.1E+1	8.7E+0	2.6E+0
	1.8E+0 – 1.0E+2	2.2E+0 – 7.2E+1	6.7E–1 – 2.2E+1
Cs	2.5E–2	2.6E–2	3.1E–3
	3.6E–3 – 7.0E–1	6.0E–3 – 6.2 E–1	7.2E–4 – 7.4 E–2
I	3.9E+0		
	6.5E–1 – 4.3E+1		
Ni	1.5E–1		
	2.7E–2 – 1.2E+0		
Pu	4.1E–3	7.3E–5	7.3E–5
	6.7E–4 – 4.3E–2	1.8E–5 – 7.0E–4	1.8E–5 – 7.0E–4
Ra	5.8E–2		
	9.7E–3 – 5.8E–1		
Sr	2.0E–1		
	3.0E–2 – 3.9E+0		
Tc	1.8E+1		
	2.6E+0 – 2.8E+2		

Note: The values given are the median, the 1 percentile (lower value of the interval) and the 99 percentile (higher value of the interval) of the simulated probability distributions.

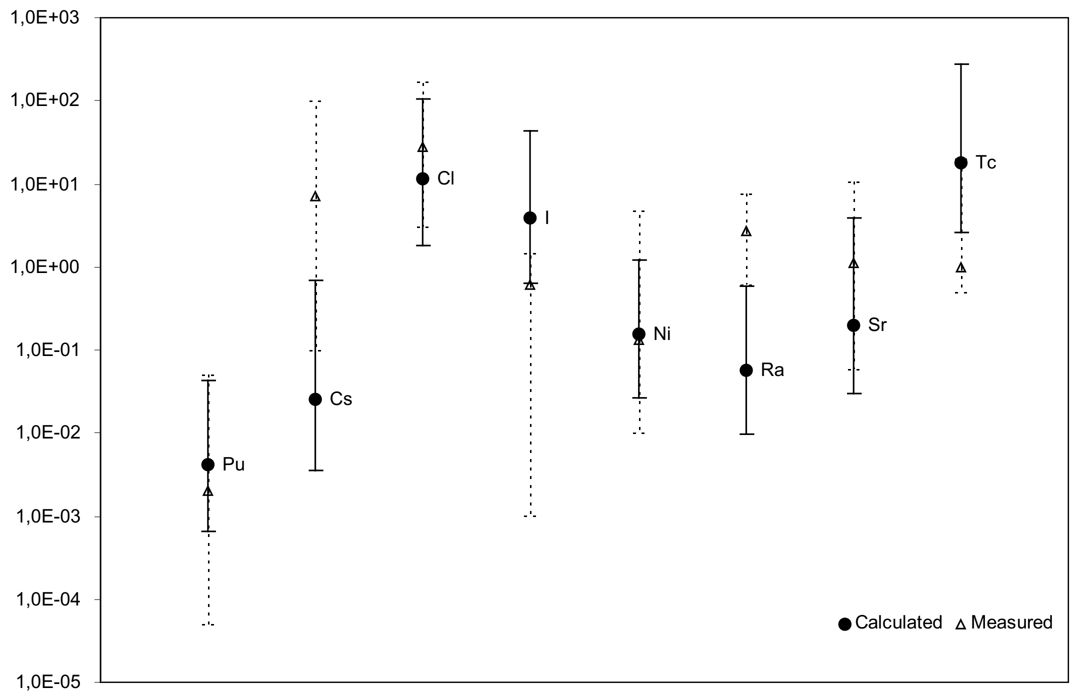

Figure 5-1. *Calculated (Table 5-1) and empirical values (Table 3-7) of the concentration ratios from soil to understorey plants.*

For some of the studied radionuclides, this approach seems to give an underestimation (Cs and Ra) or overestimation (I and Tc) of the *CR*s. The overestimation could be partly explained by the fact that the permeability coefficients σ were set equal to 1, when they are most likely lower than one. The discrepancies might be also a result of the parameter uncertainty. However, in some cases the differences might abide more fundamental reasons, for instance that the implicit assumption of linear proportionality, of the radionuclide uptake rates to the transpiration rate and the radionuclide concentration in the soil solution, might be wrong. For example, experimental studies /Sheppard and Evenden 1988/ have shown that for some nuclides, like Se and U, the concentrations in plants are proportional to the concentration in the soil pore water, while for others, like Cs, they are proportional to the total nuclide concentration in soil.

In general, the uptake rate of some plant nutrients, like K (Cs analogue), is not limited by transpiration rates, but rather by diffusion in soil and accumulation-depletion processes near the roots /Marschner 1995/. For these radionuclides, and in particular for analogues of plant macronutrients, an alternative approach has been proposed based on the assumption that their uptake by plants is modulated by the plant uptake of the nutrient /Casadesus et al. 2001/. This means that the radionuclide and its corresponding analogue nutrient are taken up by plants in an identical manner via the same carrier molecules. Assuming that only ions in the soil solution near the roots, where the radionuclide concentrations are much lower than analogue concentrations, are available for transition into the roots, the transition of radionuclides from soil to plants can be represented as an independent Poisson process. It follows from this, that the uptake rate of the radionuclide will be proportional to the uptake rate of the analogue nutrient and that the concentration of the radionuclide in the soil solution near the roots and inversely proportional to the analogue concentration near the roots. Assuming that the uptake rate of the nutrient is proportional to the nutrient demand (excluding excess uptake of nutrients) and that the later is proportional to the biomass production, the following general equation can be proposed for soil to plant transfer rate coefficients:

$$TC^{j}_{S\,To\,P}(t) = \frac{Sc^{j}}{\rho * h} * \frac{[A]^{j}_{P}}{[A]^{j}_{s}} * \frac{dM_{P}}{dt} * \frac{B^{j}}{BA^{j}} \tag{14}$$

where,

$TC^{j}_{S\,To\,P}(t)$ is the transfer rate coefficient of the ***j-th*** radionuclide from ***soil*** to ***plant*** [y–1],

$\frac{dM_{P}}{dt}$ is the yearly psroduction of plant biomass [kg/m^2/y],

Sc^{j} is the selectivity coefficient between the ***j-th*** radionuclide and its analogue nutrient [–],

$[A]^{j}_{P}$ is the concentration in plant of the analogue nutrient to the ***j-th*** radionuclide [mol/kg],

$[A]^{j}_{s}$ is the concentration in soil of the analogue nutrient of the ***j-th*** radionuclide [mol/kg],

B^{j} is the bioavailability factor of the ***j-th*** radionuclide [–],

BA^{j} is the bioavailability factor of the analogue nutrient to the ***j-th*** radionuclide [–],

ρ is the soil bulk density [kg/m^3],

h is the thickness of the soil root layer [m].

The selectivity coefficient, *Sc*, in Equation 14 is a value between 0 and 1, with 0 indicating that the plant completely distinguishes and rejects the radionuclide, and 1 indicating the inability of the plant to distinguish between the two ions. The bioavailability factors, *B* (*BA)*, represent the fraction of the total radionuclide (analogue) content in the soil that is available for uptake by plants, i.e. the fraction in the soil solution near the roots, or that can be transferred there within a given time interval. Details on the definition of bioavailability factors and their estimation from experimental data, or with the help of mechanistic models, can be found in /Avila et al. 2001b, Norden et al. 2004, Gonze et al. 2004/.

5.2 Sorption in soils

The sorption and fixation processes in soil influence the mobility and bioavailability of radionuclides and are therefore important for estimating the radionuclide leaching from the system and their uptake by plants. The model describes the radionuclides sorption with a simplified model, based on the Langmuir and Freundlich equations, with the distribution coefficient, Kd, corresponding to the slope of a linear isotherm. Values of Kds have been reported for all considered radionuclides, although these have been measured in different environments and can vary within a range of more than two orders of magnitude. Also, in a given soil, the Kds exhibit both vertical and spatial variability and can vary with time due to fixation and remobilisation processes.

For some nuclides it might be even necessary to substitute, or at least modify, the Kd model in order to properly describe the processes determining their mobility and bioavailability. One example is Tc, which mobility in anaerobic soils, especially those with high organic matter, is strongly retarded, probably as TcO_2 /Bennett and Willey, 2003 and references therein/ and its bioavailability to plants is lower. TcO_2 can be absorbed to the soil solid-phase in reducing conditions by complexing with organic matter and where the complexes are re-oxidised very slowly. Oxidation of Tc compounds is probably so slow that accumulation of Tc is possible in soils that are subject to periodic water logging /Yanagisawa and Muramatsu 1993, 1995, Tagami and Uchida 1996/. Low redox potentials can also cause TcO_4^- to react with hydrogen sulphide to form Tc_2S_7, which is not available to plants /Brookins 1988, Tagami and Uchida 1996, 1997/.

Despite the limitations of the Kd model, this is the most convenient and practical method available to account for sorption processes. There have been proposed models that describe the kinetics of the processes in more detail, but their use is limited due to lack of data for their parameterisation. Hence, it seems that efforts should focus on determining Kd values that are representative for forests and for the temporal and spatial domains relevant to the model. Reported empirical relationships between Kd and soil properties, as those proposed for Cs /Sanchez et al. 2002/ and U /Echevarria et al. 2001/, could be also used to derive appropriate values. As it was shown above, the Kds are used in the equations of several transfer rate coefficients included in the model. If the values of several of these transfer rate coefficients are estimated from field investigations, then it should be possible to derive appropriate Kd values.

5.3 Transfer to animals

The model of transfer to animals requires further testing, which could be achieved by performing measurements of element concentrations in animals and their feed. The same type of data could be used for estimating missing allometric coefficients for some of the radionuclides of interest, for example Se. The model currently considers only the transfer to herbivores via food ingestion. The same kinetic-allometric approach could be applied for carnivores and for considering other transfer pathways, such as water ingestion and inhalation. The model could be also further improved by describing food intake rates as a function of metabolic rates, as proposed in /Whicker and Shultz 1982/. This would facilitate estimating the diet composition under the constraints set by biomass/energy production in the system. At the moment, only total body concentrations can be calculated with the model, whereas some nuclides are known to accumulate in specific organs, e.g. I in the thyroids, Sr in bones, actinides in kidneys and liver. Concentrations in specific organs could be estimated from empirical measured ratios between concentrations in organs and in the whole body /Coughtrey et al. 1985/ or with the help of more detailed kinetic models. For nuclides that are poorly assimilated in the animal body, such as the actinides, the fractional gut uptake, f, is one of the parameters that contributes the most to the uncertainty in estimation of transfer to animals /Avila et al. 2004/. Satisfactory methods for disaggregating this parameter have not been reported and thus a better estimation will have to rely on the availability of more empirical data.

5.4 Missing processes

The most obvious missing process in the model is the radionuclide vertical redistribution in soil and corresponding distribution of plant and tree roots. The model describes only one soil layer, but it can be conveniently scaled to couple with several layers, for example in a similar way as in the Coup model /Jansson and Karlberg 2004/. The number of needed layers will depend on the hydrological characteristics of the system modelled, such as the depth of the groundwater table. Taking into account the long-term time scope of the assessments, it is probably sufficient to consider an equilibrium water-content profile that will vertically transmit to the water table a constant flux of water equal to the local climatic-average rate of recharge. The equilibrium profile is useful for portraying the typical effects of climate, soil type and water table depth on the hydraulic conditions of the soil and can be computed as described by /Salvucci and Entekhabi 1994/. The uptake of radionuclides from different soil layers can be accounted for by using the approaches described in epigraph 5.1.

Other missing processes, some mentioned in epigraph 3.1.1, are the re-suspension and volatilisation of radionuclides from soil, transfer to litter by weathering processes, inhalation of radionuclides by animals. Although these processes are probably of less importance, they should be included in the model for completeness. Also, a process like volatilisation might be important for the vertical transport of some nuclides, like I /Johanson 2000/. Moreover, if other radionuclides, like H3 and C14 (non-considered here), are included in the model, then other processes may emerge requiring consideration.

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