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Peter Saetre, Sara Nordén, Sven Keesmann
Svensk Kärnbränslehantering AB

Per-Anders Ekström, Facilia AB

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Svensk Kärnbränslehantering AB
Swedish Nuclear Fuel
and Waste Management Co

Box 250, SE-101 24 Stockholm
Phone +46 8 459 84 00

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Update notice

The original report, dated December 2013, was found to contain factual errors which have been corrected in this updated version. The corrected factual errors are presented below.

Updated 2017-09

Location	Original text	Corrected text
Page 63, Equation 6-22.	Wrong equation.	Equation updated.
Page 75, Equation 7-16.	Wrong equation.	Equation updated.
Page 79, explanation Equation 7-29, line $RegoUp_{DM,i}$.	Wrong text.	$RegoUp^{RN}_{DM,i}$ is the initial activity of other radionuclides in the soil plant system, and is used synonymously with $RegoUp^{RN}_{DM}$ [Bq].
Page 79, explanation Equation 7-29.	$area_{cult}$ is missing.	$area_{cult}$ is the surface area of cultivated land (i.e. $support\_area_{DM}$ times N_{group}) [m^2].
Page 84, explanation Equation 7-45.	$RegoUp^{RN}_{i,aver,DM}$ is missing.	$RegoUp^{RN}_{i,aver,DM}$ is the average inventory for crop i, which implies that all right hand terms of Equation 7-44 is scaled by the fraction of area used for each crop respectively ($f_{area,i}$).
Page 101, Equation 8-19.	Wrong equation.	Equation updated.
Page 141, explanation Equation B-2.	I_{gas} [Bq] is the inventory in pore water and	I_{gas} [Bq] is the inventory in the gas phase and
Page 156, Table G-1.	WF_i	q_i
Page 156, Table G-1.	WF_i (m^3 year^{-1})	q_i (m year^{-1})

Abstract

This report presents the *radionuclide model for the biosphere*, which is the final link in the chain of performance assessment models used to demonstrate that the extended SFR repository (SFR 1 and SFR 3) meets regulatory criteria with respect to long-term safety. The model is used for the quantitative calculations of transport and accumulation of radionuclides in surface ecosystems, and for the assessment of radiological consequences to humans and the environment over very long periods of time. For these calculations the model has been parametrised with data that are relevant for Forsmark over the assessment period and combined with appropriate assumptions on release and biosphere conditions.

Radionuclides from the geological repository reach the biosphere primarily through transport with groundwater which is discharged to local near-surface environments (i.e. biosphere objects). Transport and accumulation of radionuclides is simulated in two linked natural ecosystems (i.e. an aquatic and a mire ecosystem) of a biosphere object. The distribution of radionuclides in the landscape is then modelled by linking biosphere objects through surface-water driven fluxes of radionuclides. Dynamic changes in ecosystem structures, caused by for example land rise, lake isolation and mire ingrowth, are all handled as continuous processes in time.

The biosphere model is conceptually very similar to the one used in the latest SKB safety assessment SR-Site. However, the original SR-Site model has been updated and improved in several ways. For example, the resolution of regolith layers has been increased to better capture the fate of radionuclides in organic matter, the representation of gas exchange has been updated, and a new sub-model of the near surface atmosphere has been developed. Dose calculations have also been updated to reflect a number of explicitly defined exposed groups. These groups are credible bounding cases for relevant exposure pathways, and reflect land use and habits that are reasonable and sustainable. The calculations of dose rates to other biota, previously carried out in the ERICA Tool, have also been included in the model code.

This report starts with a general introduction of the SR-PSU project, followed by a brief description of the assessment purpose and its endpoints. The present conditions and the expected development of the climate and the surface ecosystems in Forsmark are presented as a background to the model, as are a brief description of biosphere objects and an overview of identified exposure pathways. Chapter 3 introduces the physical compartments and the radionuclide fluxes that are used to simulate transport and accumulation of radionuclides in surface ecosystems. The next seven chapters make up the core the report. In these chapters equations representing inventories and activity concentrations of compartments and radionuclide fluxes between compartments and biosphere objects are presented, together with the rational for the equations and underlying simplifying assumptions.

The presentation of the mathematical model is organized in thematic chapters, starting with transport by advection and diffusion in water (Chapter 4). The chapter describes vertical and horizontal transport by ground and surface water and hydrological links between ecosystems and biosphere objects. Next follows two chapters (5 and 6) that describe the sub-models for aquatic (sea, lake and stream) and mire ecosystems. These two sub-models are linked, and together they make up the core transport model used to simulate long term accumulation of radionuclides and activity concentrations in regolith and water within a biosphere object. In Chapter 7 the sub-model for agricultural ecosystems is described. The model which is relatively simple describes the layer of cultivated soil only. Initial values for radionuclide source terms and soil concentrations are provided by the coupled lake-mire sub-model. In Chapter 8 the sub-model for transport and exchange of radiocarbon in the near surface atmosphere is described. As the exchange of air between atmosphere layers is a relatively fast processes, the activity concentration in atmospheric layers are solved for steady state conditions, and consequently the atmosphere is treated as a source and a sink in the transport simulations.

Finally equations for dose calculations for human inhabitants and non-human biota are described in Chapters 9 and 10. Annual doses are calculated by combining activity concentrations in environmental media (soil, water and air), provided by the sub-models for aquatic, mire and agricultural ecosystems and the near surface atmosphere, with exposure times corresponding to the land-use associated with an exposed group. Similarly, dose rates to representative organisms are calculated from radionuclide concentrations in relevant habitats, following the ERICA methodology.

Sammanfattning

I denna rapport presenteras radionuklidmodellen för biosfären, som är den sista länken i den kedjan av modeller som används för att visa att det utbyggda SFR-förvaret (SFR 1 och SFR 3) uppfyller regler och kriterier med avseende på den långsiktiga säkerheten. Modellen används för kvantitativa beräkningar av transport och ackumulation av radionuklider i ytnära ekosystem och för att bedöma radiologiska konsekvenser för människor och miljön över mycket långa tidsperioder. För dessa beräkningar har modellen parametriserats med data som är relevanta för Forsmark under beräkningsperioden och kombinerats med rimliga antaganden om utsläpp och ytförhållanden.

Radionuklider från det geologiska förvaret når biosfären främst genom transporter med grundvatten som strömmar ut i lokala ytnära miljöer (dvs biosfärsobjekt). Transport och ackumulation av radionuklider modelleras i två naturliga länkade ekosystem (dvs ett akvatiskt och ett våtmarks- ekosystem) inom ett biosfärsobjekt. Fördelningen av radionuklider i landskapet modelleras därefter genom att länka biosfärsobjekt med ytvattendrivna flöden av radionuklider. Dynamiska förändringar i ekosystemens struktur, som orsakas av t.ex. landhöjningen, sjöisolering och inväxt av våtmarker, hanteras som kontinuerliga processer över tiden.

Biosfärsmodellen är konceptuellt mycket lik den som används i SKB:s senaste säkerhetsanalys SR-Site. Den ursprungliga SR-site modell har emellertid uppdaterats och förbättrats på flera sätt. Till exempel har upplösningen på regolitskikt ökats för att förbättra beskrivningen av ackumulation av radionuklider i organiskt material. Dessutom har representationen av gasutbyten uppdaterats, och en ny delmodell av den ytnära atmosfären har utvecklats. Dosberäkningar har också uppdaterats för att återspegla ett antal tydligt definierade exponeringsgrupper. Dessa grupper kan ses som trovärdiga begränsningsfall för relevanta exponeringsvägar, och reflektera en markanvändning som är rimlig och hållbar i ett historiskt perspektiv. Beräkningarna av dosrater till icke-mänsklig biota, som tidigare utförts i ERICA-verktyget, har nu även inkluderats i beräkningskoden.

Rapporten inleds med en allmän introduktion av SR-PSU-projektet, följt av en kort beskrivning av säkerhetsanalysens syfte och de kvantiteter som används för beräkningar av skadeverkningar. De nuvarande förhållandena i Forsmark och den förväntade utvecklingen av klimatet och de ytnära ekosystemen presenteras som en bakgrund till modellen. Här ges även en kort beskrivning av identifierade biosfärsobjekt och en översikt av relevanta exponeringsvägar. Kapitel 3 presenterar de förråd och flöden av radionuklider som används för att simulera transport och ackumulation i ytnära ekosystem. De kommande sju kapitlen utgör rapportens kärna. I dessa kapitel presenteras de ekvationer som beskriver aktivitetskoncentrationer av radionuklider i miljömedier, samt flödet av radionuklider inom och mellan ytekosystem. Här ges även en motivering till de matematiska beskrivningarna som använts och bakomliggande antaganden och förenklingar.

Presentationen av den matematiska modellen är organiserad i tematiska kapitel. Det inledande kapitlet beskriver transport via advektion och diffusion i vatten (kapitel 4). Kapitlet presenterar ekvationerna som används för vertikal och horisontell transport med grund- och ytvatten, samt redovisar hydrologiska kopplingarna mellan ekosystem och biosfärsobjekt. Därefter följer två kapitel (5 och 6) som beskriver delmodellerna för vatten- (hav, sjö och vattendrag) och våtmarks-ekosystem. Dessa två delmodeller är kopplade, och bildar tillsammans den kärnmodell som används för att simulera långsiktig ackumulation av radionuklider och resulterande aktivitetskoncentrationer i mark och vatten inom ett biosfärsobjekt. I kapitel 7 beskrivs delmodellen för jordbruksekosystem. Modellen är relativt enkel och beskriver enbart det övre uppodlade jordskiktet. Initialvärden för källtermer och markkoncentrationer hämtas från simuleringarna av det kopplade sjön-våtmarksekosystemet. I kapitel 8 beskrivs delmodell för transport och utbyte av kol-14 i den ytnära atmosfären. Eftersom utbyten av luft mellan atmosfärslager sker relativt snabbt, beräknas aktivitetskoncentrationen i atmosfären för stationära förhållanden, och atmosfären behandlas som en källa respektive en sänka för radionuklider i transportsimuleringar.

Slutligen beskrivs ekvationer för dosberäkningar för människor och andra organismer i kapitel 9 och 10. Årliga doser beräknas genom att kombinera aktivitetskoncentrationer i miljön (mark, vatten och luft), vilka beräknats i delmodellerna för vatten-, våtmarks- och jordbruks-ekosystem och den ytnära atmosfären för respektive system, med exponeringstider som motsvarar markanvändningen för en exponerad grupp. På ett liknande sätt beräknas dosraten till representativa organismer från radionuklidkoncentrationer i relevanta livsmiljöer enligt ERICA-metodiken.

Contents

1 Introduction

1.1 The SR-PSU Biosphere Assessment

The final repository for short-lived low and intermediate-level radioactive waste, SFR 1, is located in Forsmark in northern Uppland, in the immediate vicinity of the Forsmark nuclear power plant (Figure 1-1). The SFR 1 repository consists of a set of disposal chambers situated in rock at ca 60 m depth beneath the sea floor, and is built to receive and after closure serve as a passive repository for low- and intermediate-level short-lived radioactive wastes. The radioactive wastes stored and disposed in SFR include operational wastes from Swedish nuclear power plants and from the interim storage facility for spent nuclear fuel, Clab, as well as radioactive wastes from other industries, research institutions and medical establishments.

In order to be able also to store and dispose decommissioning wastes from the Swedish nuclear power plants in SFR, an extension of the repository, referred to as SFR 3 is planned. An SFR repository extension called SFR 2 was included in earlier plans for disposal of reactor core components and internal parts, which are now planned to be disposed of in a separate deep geological repository (SFL).

As a part of the license application for the extension of SFR, the Swedish Nuclear Fuel and Waste Management Company (SKB) has performed the SR-PSU project. The objective of SR-PSU is to assess the long-term radiological safety of the entire future SFR repository, i.e. both the existing SFR 1 and the planned SFR 3. SR-PSU is reported in a series of SKB reports, which includes a main report, referred to as SR-PSU Main report (SKB 2014g), and a set of primary references. These include, among others, the reports denoted Climate report (SKB 2014b), Radionuclide transport report (SKB 2014f), FEP report (SKB 2014c), FHA report (SKB 2014d) and Biosphere synthesis report (SKB 2014a) in the SR-PSU reporting (Table 1-1). In addition to these primary references, the safety assessment is based on a large number of background reports and other references. The present report is a background report for the Biosphere synthesis report (SKB 2014a) summarising the SR-PSU Biosphere project as a sub-project of SR-PSU.

Table 1-1. The SR-PSU Main report and main references to it produced within SR-PSU. FEP stands for features, events and processes and includes FEP for all disciplines in the assessment (e.g. waste, geosphere and climate). FHA is short for future human actions.

Report number	Short name used when referred to in the text	Full title
TR-14-01	SR-PSU Main report	Safety analysis for SFR. Long-term safety. Main report for the safety assessment SR-PSU.
TR-14-02	Initial state report	Initial state report for the safety assessment SR-PSU.
TR-14-03	Waste process report	Waste form and packaging process report for the safety assessment SR-PSU.
TR-14-04	Barrier process report	Engineered barrier process report for the safety assessment SR-PSU.
TR-14-05	Geosphere process report	Geosphere process report for the safety assessment SR-PSU.
TR-14-06	Biosphere synthesis report	Biosphere synthesis report for the safety assessment SR-PSU.
TR-14-07	FEP report	FEP report for the safety assessment SR-PSU.
TR-14-08	FHA report	Handling of future human actions in the safety assessment SR-PSU.
TR-14-09	Radionuclide transport report	Radionuclide transport and dose calculations for the safety assessment SR-PSU.
TR-14-10	Data report	Data report for the safety assessment SR-PSU.
TR-14-11	Model summary report	Model summary report for the safety assessment SR-PSU.
TR-14-12	Input data report	Input data report for the safety assessment SR-PSU.
TR-13-05	Climate report	Climate and climate-related issues for the safety assessment SR-PSU.

Figure 1-1. *The surface part of the SFR facility in Forsmark harbour with the Forsmark nuclear power plant in the background.*

The biosphere is a key part of the system considered in a safety assessment of a nuclear waste repository. This is where the main consequences of potential future radionuclide releases from the repository would arise, and hence near-surface radionuclide transport and dose calculations are performed within the framework of the biosphere assessment. This report belongs to the sub-project of SR-PSU called SR-PSU Biosphere.

SR-PSU Biosphere primarily describes the information needed to calculate impacts on humans and the environment in the case of a radionuclide release from SFR. The calculated impacts are then used to show compliance with regulations related to future repository performance for time spans up to 100,000 years after closure. Because of the uncertainties associated with the prediction of future development of the site in this time frame, a number of calculation cases are analysed to describe a range of possible site developments. For details of the biosphere assessment and associated tasks, methodology, and organisation see the Biosphere synthesis report (SKB 2014a). All reports in SR-PSU Biosphere are listed in Table 1, and an overview of the report structure is presented in Figure 1-2.

Table 1-2. Biosphere background reports produced within SR-PSU Biosphere; FEP stands for features, events and processes.

Report number	Short description and reference in text	Full title
R-12-03	DEM report, Strömgren and Brydsten (2013)	Digital elevation model of Forsmark. SR-PSU Biosphere.
R-13-01	K_d and CR report, Tröjbom et al. (2013)	K_d and CR used for transport calculations in the biosphere in SR-PSU.
R-13-18	Biosphere parameter report, Grolander (2013)	Biosphere parameters used in radionuclide transport modelling and dose calculations in SR-PSU.
R-13-19	Surface hydrology report, Werner et al. (2013a)	Hydrology and near-surface hydrogeology at Forsmark – synthesis for the SR-PSU project. SR-PSU Biosphere.
R-13-20	Hydrological data report, Werner et al. (2013b)	Meteorological, hydrological and hydrogeological monitoring data from Forsmark – compilation and analysis for the SR-PSU project. SR-PSU Biosphere.
R-13-22	RDM report, Sohlenius et al. (2013a)	Depth and stratigraphy of regolith at Forsmark. SR-PSU Biosphere.
R-13-27	RLDM report, Brydsten and Strömgren (2013)	Landscape development in the Forsmark area from the past into the future (8500 BC to 40,000 AD).
R-13-43	Biosphere process definition report, SKB (2013b)	Components, features, processes and interactions in the biosphere.
R-13-46	Biosphere radionuclide model report, Saetre et al. (2013a)	The biosphere model for radionuclide transport and dose assessment in SR-PSU.
R-14-02	Biosphere FEP handling report, SKB (2014)	Handling of biosphere FEPs and recommendations for model development in SR-PSU.

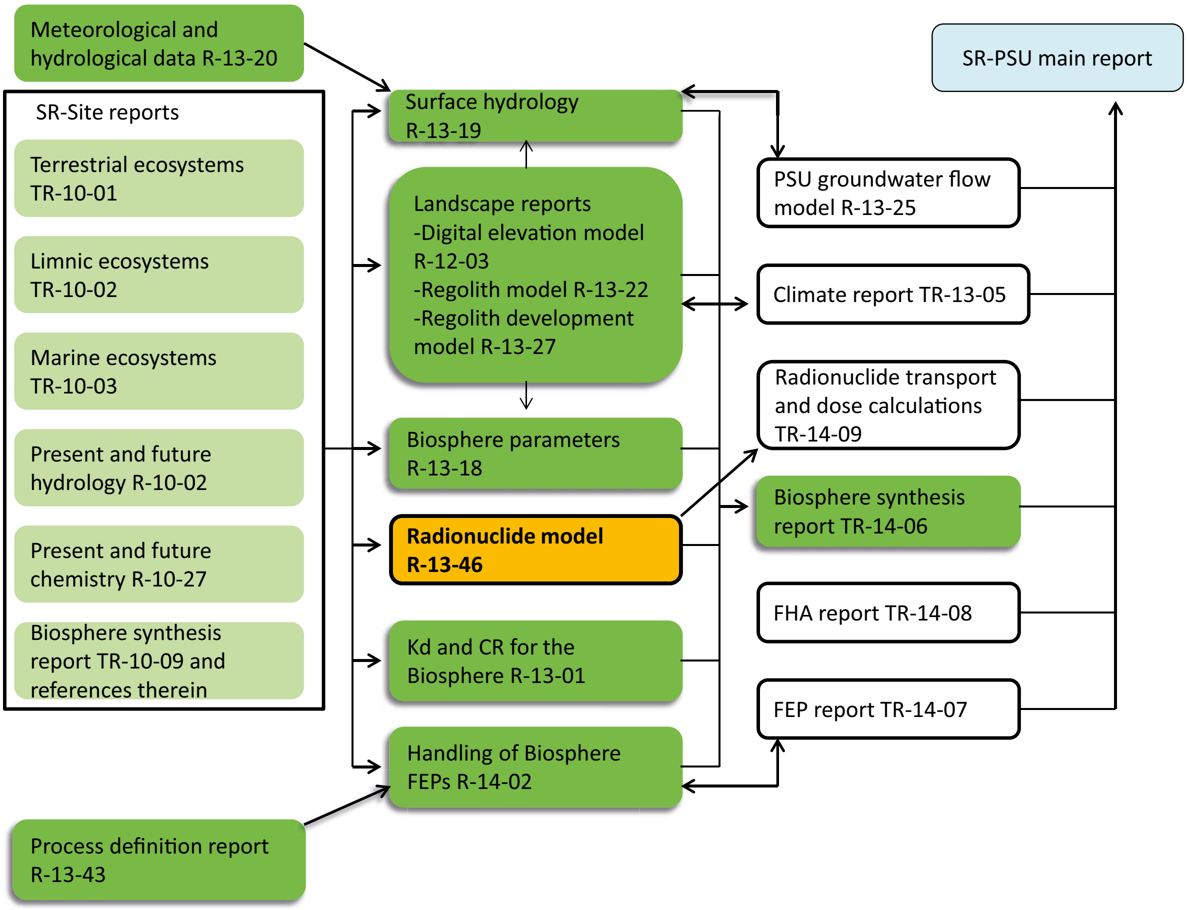

Figure 1-2. Relationship between reports produced in the SR-PSU Biosphere project (dark green boxes). The present report is shown in orange and bold. Supporting documents produced within other biosphere projects at SKB are shown as light green boxes, whereas other reports in the SR-PSU project are shown in white except the SR-PSU Main report (SKB 2014g), which is shown in blue.

1.2 This report

The present report is a background report for SR-PSU Biosphere, documenting the mathematical model for the quantitative assessment of the long-term transport and accumulation of radionuclides in the biosphere and radiological impacts on humans and non-human biota. This model, referred to as *the radionuclide model for the biosphere*, is the final link in the chain of performance assessment models used to demonstrate compliance of the SR-PSU assessment with quantitative regulatory requirements on long-term safety. The other SR-PSU performance assessment models for the near-field and the far-field of the repository are documented in SKB (2014f).

The aim of the radionuclide model for the biosphere is to model radiological consequences of potential releases of radionuclides from wastes disposed of in the SFR repository to the near-surface environment over very long periods of time.

Any releases of repository derived radionuclides from the geosphere to the near-surface environment would be a consequence of processes causing the release of those radionuclides from waste packages and their migration through the engineered barriers of the repository and the geosphere. Therefore, there is a need to describe the consequences of releases of these radionuclides into surface ecosystems in the far future, by which time the landscape will have undergone considerable changes and intrinsically unpredictable societal developments with their impact on human behaviour will have occurred. It is assumed that the future generations who will be exposed to these releases will not be aware of the presence of the repository or of the impacts of those releases on their living environment.

Given the requirements of the safety assessment and the features of the potentially impacted environments, the radionuclide model of the biosphere was required to be capable of (i) incorporating temporal changes in ecosystems driven by e.g. land rise, shoreline displacement, ecosystem succession and climate change, (ii) handling time-dependent releases of radionuclides with groundwater discharges and their distribution in a heterogeneous landscape, (iii) handling transport, accumulation and decay of radionuclides (including progeny) with different geochemical behaviour, (iv) assessing radiation exposures of humans and wildlife, and (v) accounting for transport of radionuclides between different areas of the landscape (driven by e.g. advection or surface runoff).

The model presented in this report builds on those developed for previous safety assessments for existing and planned nuclear waste repositories in Sweden. The SFR repository has been in operation since 1988 and a number of safety assessments have been performed for it since SKB received permission to start building SFR 1 in 1983. These include the previous SFR safety assessments SAFE (Kautsky 2001, Lindgren et al. 2001) and SAR-08 (SKB 2008). In addition, the model described here builds on work undertaken for the SR-Site safety assessment for the planned repository for spent nuclear fuel in Forsmark (SKB 2011). In the most recent of these safety assessments SKB presented a model that simulated the fate of radionuclides in surface ecosystems of potentially affected areas of the Forsmark landscape (Avila et al. 2010, 2013). That model had the above-mentioned capabilities.

In SR-Site, the distribution of radionuclides in the landscape was modelled by linking potentially affected local near-surface environments (the so called biosphere objects) via transfers representing surface-water driven fluxes of radionuclides. It simulated transport and accumulation of radionuclides in two linked ecosystems (i.e.an aquatic and a mire ecosystem) of a biosphere object, and the transitions between ecosystems (i.e. lake isolation and lake ingrowth) were handled as continuous processes in time. The description of the transition in time was derived from extensive landscape modelling of the development of the Forsmark area over a full interglacial period (Lindborg 2010, Lindborg et al. 2013).

The model presented in this report is conceptually very similar to the one used in the SR-Site assessment from which it was derived. However, due to the specific relevance of carbon in biosphere-related processes and of radiocarbon (C-14) in the present assessment, the model has been improved in several ways to better capture the fate of radiocarbon for a more complete estimate of activity concentrations in air, soil and water (see Chapter 3 for an overview of compartments and fluxes). Moreover, the most exposed group has been defined to explicitly reflect the behaviour of self-sustained historical communities, and the calculations of dose rates to non-human biota, previously carried out in the ERICA Tool, have been included in the model code.

The report is divided into chapters and sections that are intended to guide the reader through the radionuclide model for the biosphere used in SR-PSU. In addition, the report is supported by a number of technical appendices.

Chapter 1 – Introduction sketches the context of the sub-project to which this report is contributing and gives background to the radionuclide model for the biosphere.

Chapter 2 – Assessment context and site-specific features starts with a brief description of the assessment purpose and its endpoints. A brief description of the present conditions and the expected development of the climate and the landscape in Forsmark follow. The chapter also presents the Biosphere object, a key-concept in the surface assessment, and identifies exposure pathways.

Chapter 3 – The structure of the radionuclide transport model explains the fundamental concepts and relevant processes considered in the model. These include the physical compartments and the radionuclide fluxes that are used to simulate transport and accumulation of radionuclides in surface ecosystems.

Chapter 4 – Transport by advection and diffusion in water describes transport with ground and surface water in aquatic and mire ecosystems. The chapter describe both vertical and horizontal transport and lists the hydrological links within and between ecosystems and biosphere objects that are used in the transport simulations.

Chapter 5 – Aquatic ecosystems describes the sub-model for sea, lake and stream ecosystems. The chapter is divided into two sections. The first section describes inventories and corresponding activity concentrations of model compartments. The second section describes the rates of radionuclide fluxes between compartments (or into and out of the system).

Chapter 6 – Mire ecosystems describes the sub-model for wetland ecosystems. The chapter is divided into two sections. The first section describes inventories and corresponding activity concentrations of model compartments. The second section describes the rates of radionuclide fluxes between compartments (or into and out of the system).

Chapter 7 – Agricultural Ecosystems describes the sub-models for agricultural ecosystems. The chapter is divided into sections that describe three different cultivation systems, namely Infield-outland cultivation, draining and cultivation of a lake-mire complex, and cultivation of a garden plot. The solutions to the differential equations describing radionuclide accumulation in cultivated soils are relatively simple, and algebraic expressions for activity concentrations in cultivated soils are derived.

Chapter 8 – Surface atmosphere provides a short description of the sub-models used to calculate the activity concentration (and the specific activity) of $^{14}CO_2$ in the near-surface atmosphere. That is, in the atmospheric layers where gas is exchanged with soil and water, where plant uptake occurs and where humans may be exposed to radionuclides in gaseous form due to inhalation.

Chapter 9 – Dose calculations for human inhabitants describes how radiation doses to future inhabitants are calculated from radionuclide concentrations in environmental media and assumptions as to the habits and behaviour of the relevant groups. The land-use variants evaluated in the assessment are: foraging the landscape for food, self-sufficient farming and cultivation of a garden plot.

Chapter 10 – Dose rate calculations for non-human biota describes how dose rates to non-human biota are calculated from radionuclide concentrations in environmental media and assumptions as to organism habitats. The dose rate calculations follow the ERICA methodology.

2 Assessment context and site-specific features

This chapter gives an overview of the assessment context, site-specific features and the analysis of exposure pathways relevant to the biosphere radionuclide model. See the Biosphere synthesis report (SKB 2014a) for a complete presentation of these topics.

2.1 Assessment purpose and endpoints

The SR-PSU safety assessment is undertaken to ensure that human health and the environment are protected from harmful effects of ionising radiation and radioactive material originating from the existing and the planned repositories for low and intermediate waste in Forsmark. The purpose of the SR-PSU assessment is to show compliance with regulatory requirements, and the biosphere radionuclide model is one link in a chain of numerical models to evaluate quantitative aspects of the assessment.

According to the Swedish regulations, a repository for spent nuclear fuel or nuclear waste should be designed so that the annual risk of harmful effects after closure does not exceed 10^{-6} for a representative individual in the group exposed to the greatest risk (SSMFS 2008:37). However, if the exposed group includes only a few individuals, regulations are considered to be complied with if the calculated annual individual risk does not exceed 10^{-5}. The principal quantitative regulatory requirement is the risk criterion for humans. By applying the factor for risk conversion of 7.3 per cent per Sievert (ICRP 1991) the risk constraint of 10^{-6} per year is converted into a dose constraint for effective dose of 14 µSv per year.[1]

The protection of the environment includes the protection of biodiversity and the sustainable use of biological resources (SSMFS 2008:37). However the regulations do not give dose rate (or risk) constraints for the protection of non-human. In the current assessment SKB has applied the screening dose rate of 10 µGy h^{-1} proposed in the ERICA methodology (Beresford et al. 2007, Brown et al. 2008). The screening dose rate represents a no-effect threshold, below which only minor effects on non-human biota are anticipated. It can also be noted that even if dose rates were to exceed this screening level, this would not necessarily imply that significant adverse effects would occur.

Different kinds of dose caused by radionuclides released to the biosphere have to be assessed for humans and non-human biota. However, both depend on radionuclide concentrations in environmental media and on living habits, with exposure models taking account of the latter. The calculation endpoint of the biosphere radionuclide model is individual dose to a representative member of the most exposed group and to reference organisms.

Due to the very long time-scales considered for evaluating safety, uncertainties with respect to the properties of future surface ecosystems into which radionuclides may be released, and the characteristics and habits of future human inhabitants and non-human biota can only be taken into account through the use of simplifying assumptions. Intentionally, these assumptions aim at rather over- than under-estimating radiological consequences without being unrealistic. Consequently the assessment philosophy is to try to be reasonably cautious for a sufficiently robust demonstration of compliance.

SKB has aimed to make the transport model of natural ecosystems as complete as possible, with respect to model structure, primary transport pathways, landscape development at the site and the associated model parameters. However, dose calculations have been based on an analysis of potential exposure pathways, rather than on an attempt to explicitly predict living conditions and habits of generations to come in the area. Thus, exposed populations are to be interpreted as credible bounding cases (with respect to the identified exposure pathways), and SKB recognises that the endpoints of the safety assessment cannot be interpreted as providing realistic estimates of the actual risk to humans or dose rates to populations of non-human biota in the future. Projected doses in the

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[1] The currently newest recommendations of the ICRP, ICRP Publication 103, publish a risk conversion factor of 5.6 per cent per Sievert, from which a dose constraint of 19 µSv would result.

far future should consequently not be used to predict future health effects in individuals, populations or ecosystems. Instead, estimated consequences should be seen as a tool to quantify the protective performance of the existing and planned repositories.

2.2 The Forsmark site

Within the SR-PSU project, extensive site investigations have been carried out to improve the description of the surface ecosystems that are potentially affected by radionuclides released from the existing (SFR1) or the planned (SFR3) repository. These investigations on present-day and future conditions are described in the Biosphere synthesis report (SKB 2014a, Chapters 3 to 5). A short description of the site and its projected development is presented below as a background.

2.2.1 Description of the site

The Forsmark site, at which the SFR repository for low- and intermediate-level radioactive waste is located, is a coastal site near the shoreline of the Baltic Sea (Bothnian Sea) in the northern Uppland region, Sweden. The mean annual air temperature at Forsmark was approximately +7°C during the period 2004–2010 (Werner et al. 2014). The vegetation period (mean air temperature above +5°C) lasts approximately from May to September (Johansson et al. 2005).

The Forsmark area is characterised by a small-scale topography at low altitude (Strömgren and Brydsten 2008) and is located below the highest coastline associated with the last glaciation. The majority of the landscape is covered by a thin regolith layer, which was formed during or after the final phase of the latest glaciation. The mean regolith depth in the Forsmark area is c 4 m in terrestrial areas and c 8 m in marine areas (Sohlenius et al. 2013). Till and glacial clay in Forsmark have high contents of calcium carbonate ($CaCO_3$) that originated from Palaeozoic limestone that outcrops on the sea bottom north of the Forsmark area; and both surface waters and shallow groundwaters at Forsmark are generally slightly alkaline (pH 7–8) and have high concentrations of the major dissolved constituents. The underlying bedrock consists of crystalline rock that formed around 1.9 billion years ago during the Svecokarelian orogeny, and it has been affected by both ductile and brittle deformation (Söderbäck 2008).

The long-term water balance of the area is: 560 $\text{mm}\cdot\text{y}^{-1}$ precipitation, 400–410 $\text{mm}\cdot\text{y}^{-1}$ actual evapotranspiration, and 150–160 $\text{mm}\cdot\text{y}^{-1}$ runoff (Johansson et al. 2005, Werner et al. 2014). Twenty five lake catchments and sub-catchment areas have been delineated (Brunberg et al. 2004, Andersson 2010). Within these catchments, the groundwater table is within one metre of the ground surface and shallow local groundwater flow systems in the regolith overlie larger-scale flow systems in the bedrock.

The marine area at Forsmark consists of the open-ended embayment Öregrundsgrepen, with a wide and deep boundary towards the north and a narrow and shallower strait towards the south. The seabed in the coastal areas is dominated by erosion and transport bottoms with heterogeneous sediments. The area has been divided into 38 basins (Strömgren and Brydsten 2013). Most of the coastal area is shallow (sea depth less than 10 m). The direction of the annual net flow flow through Öregrundsgrepen is directed from north to south (Karlsson et al. 2010).

Brackish conditions, shallow waters, subdued bathymetry, restricted light penetration and upwelling along the coast characterise the marine ecosystems at Forsmark. This results in a high primary production in the near-shore zone (Aquilonius 2010). Faunal species diversity is relatively low, as few organisms are adapted to brackish conditions. Benthic organisms, such as macro- and micro-algae and vascular plants, dominate primary production. However advective movement of water is more important for fluxes of carbon than biotic fluxes, especially in offshore basins (Aquilonius 2010).

Present-day lakes in the Forsmark area are small and shallow (mean depth typically less than 1 m, Figure 3-6). They are characterised as oligotrophic hardwater lakes that have high concentrations of calcium and low concentrations of nutrients (Andersson 2010). Lake bottoms are covered by dense stands of macroalgae and a thick layer of microphytobenthos, and primary producers dominate the biomass. All lakes are surrounded by reed belts. The streams in Forsmark are small and many stream stretches are dry during summer. Primary production exceeds respiration in many lakes, and, for some larger lakes, the primary production involves large fluxes of carbon compared with the amounts received from the surrounding catchment area (Andersson 2010).

The terrestrial vegetation is affected by topography, regolith characteristics and human land use. A calcareous influence is typical of the north-eastern part of Uppland and has resulted in an abundance of rich fens. Wetlands cover 10–20% of the area in larger catchments and up to 25–35% in some local catchments (Johansson 2008). A major part of the wetlands is a mix of coniferous forest swamps and open mires. The woodland is dominated by coniferous trees. The largest carbon flux in terrestrial ecosystems is the uptake of carbon by primary producers, and vegetation at all the investigated localities acts as a carbon sink today (Löfgren 2010). Forestry has a long history in the area and clear cuts are frequent. Arable land and pastures are found close to earlier settlements. Pastures were once intensively used, but are today part of the abandoned farmland (Eriksson et al. 2002, Löfgren 2010).

There are holiday houses and barns but no permanent residents in a 20 km^2 area around SFR (Miliander et al. 2004). The only agricultural activity at present is situated at Storskäret. It is focused on meat production and the cattle graze outdoors during the vegetation period.

Post-glacial uplift, in combination with the flat topography, results in rapid shoreline displacement in the area. As a result, contemporary land ecosystems are young (the first parts emerged from the sea around 500 BC). New areas of the sea floor are continuously emerging above the wave base, and new lakes and wetlands are being formed (see landscape development below).

2.2.2 Future Climate

The current knowledge on past and future projected climate evolution in the Forsmark area is reviewed in the Climate report (SKB 2014b). Considering both the influence of enhanced atmospheric greenhouse-gas concentrations and the projected low-amplitude variability in insolation over tens of millennia, four cases describing the future climate in Forsmark were defined (see below). The first two of these were used to define the main safety assessment scenario in SR-PSU.

- the global warming climate case,
- the early periglacial climate case,
- the extended global warming climate case,
- the Weichselian glacial cycle climate case.

The first three cases are defined to span the range in future climate evolution associated with low (early periglacial climate case), medium (global warming climate case) and high (extended global warming climate case) human carbon emissions. However it should be noted that even with low human carbon emissions, the early periglacial climate case is unlikely to occur, and it has been included to ensure that this possibility is not neglected. The Weichselian glacial cycle climate case represents a repetition of conditions reconstructed for the last glacial cycle.

The climate cases are described in terms of climate domains (Table 2-1). In short, the temperate climate domain describes climates dominated by temperate climate conditions in a broad sense, including both situations with cold winters and either cold or warm summers, as well as warmer and wetter climates affected by global warming. The periglacial climate domain is defined strictly as pertaining to regions that have discontinuous or continuous permafrost present without the presence of an ice sheet. The periglacial climate domain has a climate colder than the temperate climate domain. The glacial climate domain is defined as regions that are covered by glaciers or ice sheets. The glacial climate domain has the coldest climate of the three climate domains.

Table 2-1. Summary of the climate cases considered in the SR-PSU safety assessment (ka is short for kilo annum, i.e. thousands of years, AP stands for after present). Note that periods of submerged conditions are included in the temperate domain and not reported separately. The Weichselian glacial cycle climate case is defined for a period of 120 ka, rather than for 100 ka as the other climate cases.

Climate case	Description	Climate domains
Global warming	Temperate conditions until 50 ka AP followed by natural variability and cooling of climate until 100 ka	Temperate: 69% Periglacial: 31%
Early periglacial	Same as the global warming case except for a 3 ka period of periglacial conditions centred at 17 ka AP	Temperate: 66% Periglacial: 34%
Extended global warming	Temperate conditions until 100 ka AP	Temperate: 100%
Weichselian glacial cycle	Repetition of reconstructed last glacial cycle conditions	Temperate: 42% Periglacial: 34% Glacial: 24%

2.2.3 Landscape development

At the end of the latest deglaciation the Forsmark area was covered by approximately 150 m of water and the nearest shoreline was situated some 100 km west of Forsmark (Söderbäck 2008). Thereafter, the isostatic rebound rate has decreased from c 3.5 m/100 years to a present rate of c 0.6 m/100 years, and is predicted to become insignificant around 30,000 AD.

As the present sea floor is uplifted, marine sediments are exposed to wave erosion, and sea basins are isolated. The stratigraphy and thickness of regolith layers will consequently change with time. Fine-grained sediments will be redistributed in the coastal area. New lakes will become successively shallower as sediments accumulate, and peat-covered areas will increase as shallow lakes are colonised by sedges and mire vegetation (Chapter 4 in SKB 2014a).

In SR-PSU the continuous changes in the geometry of the landscape are described by the Regolith Lake development model (RLDM, Brydsten and Strömgren 2013). The RLDM is divided into a marine module and a lake module. The marine module starts at the time when the area has recently been deglaciated, with initial conditions generated from the Regolith depth and stratigraphy model (RDM, Sohlenius et al. 2013). Postglacial sediments then accumulate, or are resuspended, depending on the local environment which changes over time. The lake module starts once a sea basin has been isolated, and describes the accumulation of clay-gyttja and the expansion of mire vegetation into the lake (and the subsequent accumulation of peat). The outputs from the marine and lake modules are combined to produce three-dimensional projections of surface geology for successive steps in time. From these projections, maps showing, for example, water depth, present shoreline and lake basin position, and the thicknesses of the surface regolith layers and the vegetation types present can be generated (see Landscape Development Model in Chapter 5 in SKB 2014a).

The present and future hydrology and near-surface hydrogeology will also be affected by landscape development and climate changes (Werner et al. 2014). For example, exchanges of water between sea basins will decrease as the land rises and straits narrow. Infilling of lakes will affect both evapotranspiration (decrease) and runoff (increase) for a given area. Shoreline displacement influences the hydraulic gradients in the upper part of the rock during the transition from sea to land, and the distance to the shoreline influences the depth to the groundwater table. However, climate change will have an even stronger influence on surface hydrology, and a wet and warm climate, or a cold periglacial climate, are expected to have larger effects on hydrological fluxes than those associated with other landscape-scale processes.

The above results were derived from water-flow models for marine basins and land areas, developed with the MIKE 3 FM and the MIKE SHE modelling tool respectively (Werner et al. 2014). More specific, local water-flow models were developed for 2000 AD (present conditions), 3000 AD, 5000 AD and 11,000 AD, using meteorological data representing the site and a detailed description of regolith depth and stratigraphy (Sohlenius et al. 2013, Brydsten and Strömgren 2013). The influence of a wet and warm climate on surface hydrology was investigated for the year 5000 AD, whereas the influence of periglacial conditions was discussed based on previous SKB studies of the site (Bosson et al. 2010, 2013).

Landscape development and climate changes affect both the structure and functions of surface ecosystems in several ways (Chapter 4 in SKB 2014a). For example, changes in the bathymetry of sea and lake basins (driven by e.g. shore-line displacement and sediment accumulation) affect the amount of biomass, the rate of primary production, and the relative contributions of benthic and pelagic primary producers to these ecosystem properties. Mire ecosystems develop from rich fens to peat bogs, and the accumulation of peat will continue until the decomposition rate balances the input of organic matter from wetland vegetation. The influence of the calcium-rich deposits is likely to be reduced over time, and may result in a decreasing pH of both mire and freshwater ecosystems. Moreover, a significantly warmer and wetter climate will probably result in higher biomass and productivity of both natural and cultivated terrestrial ecosystems. In contrast, the productivity in both freshwater and mire ecosystems is likely to be reduced in a cold climate, and the possibilities for cultivation will be very limited in the presence of permafrost.

2.3 Biosphere objects

According to the hydrogeological modelling, most groundwater from the repository is expected to be discharged into a relatively limited area in the sea, close to the shoreline, in the first thousand years after repository closure (Odén et al. 2014). From the bedrock, radionuclides are expected to pass through deep sediments before reaching biologically active soft bottoms and the sea water in the basin. From the primary basin radionuclides will then spread through water exchange with nearby basins, and eventually radionuclides will be diluted in the Baltic Sea. As the land rises, most groundwater from the repository will be discharged to roughly the same area (Odén et al. 2014), which by this time will have developed into a mire complex. Radionuclides from the bedrock will have to pass through deeper sediments (including deep peat) before reaching oxygenated and biologically active surface peat. From the primary discharge area, radionuclides can then reach down-stream lakes or wetland areas through transport via surface and sub-surface water fluxes.

A biosphere object is defined as an area in the present and/or future landscape that potentially, at any time during the assessment period, could receive radionuclide-containing groundwater or surface water originating from the repositories. The biosphere objects are thus the spatial unit used to quantitatively evaluate the fate of radionuclides reaching the surface ecosystems in the future Forsmark landscape. An extensive description of the concept of biosphere objects and the methods used to outline the objects in the SR-PSU assessment are provided in Chapter 6 in SKB (2014a).

In short, seven biosphere objects were identified and delineated in the Forsmark landscape for releases of radionuclides from SFR during temperate conditions (Figure 2-1). The succession in time of the biosphere objects was described using the Landscape development model (and time-dependent data on relevant object properties were extracted to be used as input parameters for the radionuclide transport model). All identified objects start out as sea basins, and end up as terrestrial areas (see landscape development above). During the land period, most of the objects go through a succession from a lake-mire complex to a mire stage, but two objects develop into the mire stage without passing through a lake stage. The transition from the sea stage into lake/mire stage lasts about 500 years. However, the time it takes for the lake to be replaced by a mire, by infilling of sediments and ingrowth of wetland vegetation, depends on the size and depth of the lake basin of each object. During permafrost conditions, one additional biosphere object was identified, namely a deep lake in the north-east of the area. This object was only considered to be significant in the radionuclide transport context if permafrost develops.

Time series of parameter values, describing each biosphere object with respect to, for example, the area of the aquatic and terrestrial ecosystems, thickness of regolith layers, average water depth, sedimentation rate, and rate of vegetation ingrowth were extracted from the Landscape model. Similarly, parameter values describing water flux between sea basins, between regolith layers within biosphere objects, and stream water fluxes were derived from MIKE 3 FM and MIKE SHE simulations.

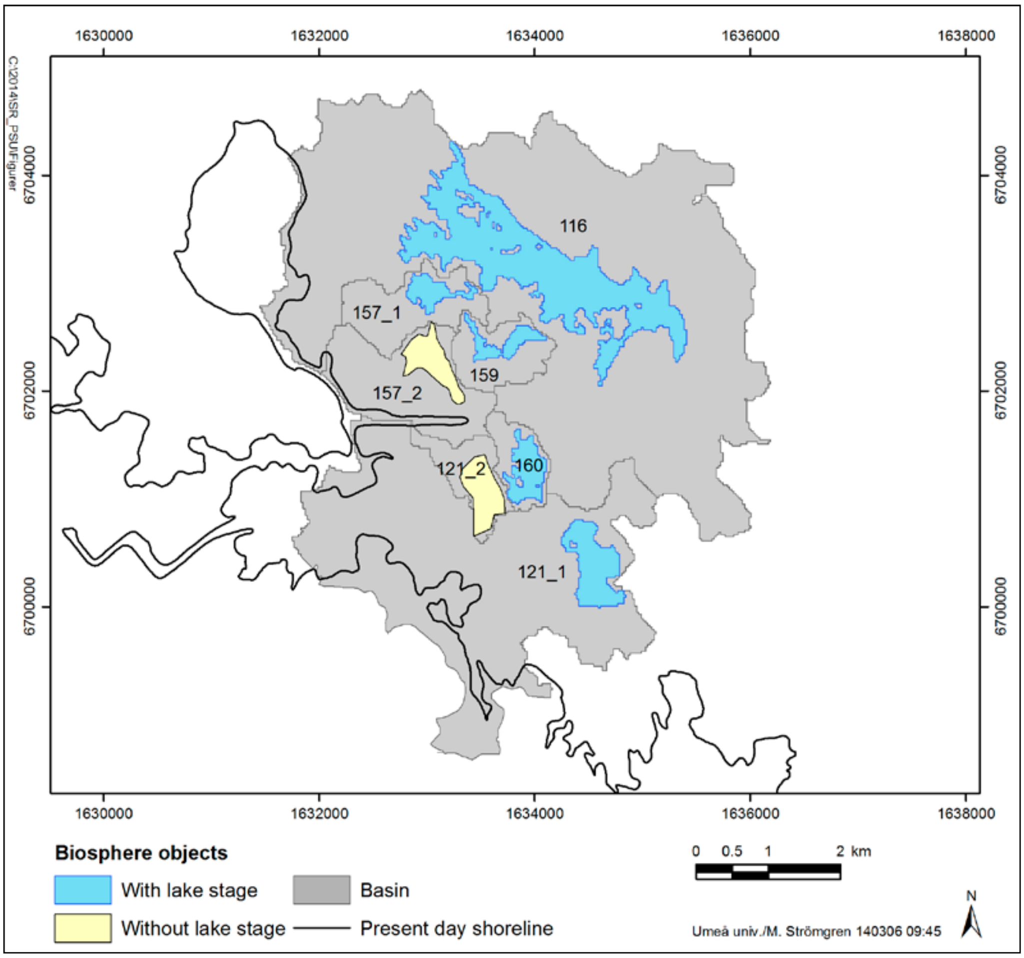

Figure 2-1. *The biosphere objects (blue or yellow) and their associated original sea basins (grey). During the sea period, the basin sets the limits for the biosphere object, whereas the initial lake (blue) or wetland area (yellow) is used to outline the biosphere object during the land period. Figure from SKB (2014a).*

2.4 Exposure pathways

An exposure pathway is the course radionuclides take from the point of release in the geosphere via the surface ecosystem to the exposed organisms, i. e. humans and non-human biota. Elements in an exposure pathway, includes: 1) the radionuclide source or release, 2) environmental transport and fate in the surface system, 3) exposure area, 4) exposure route or the way a radiation enters an organism, i.e. either external exposure to radionuclides in environmental media or internal exposure from radionuclides taken up by an organism, and 5) potentially exposed populations (SKB 2014e). These five elements largely determine to what extent exposure may occur in the future at a particular site, and need to be considered together when exposure pathways are evaluated.

2.4.1 Human inhabitants

In SR-PSU, a comprehensive range of potential exposure pathways for humans have been analysed (SKB 2014e). From the initial evaluation, a limited number of exposure pathways were considered to be quantitatively relevant for the SR-PSU safety assessment. For each of these pathways, the combinations of environmental media and exposure routes were mapped to one (or more) exposed populations (Table 2-1, SKB 2014e). In total, four distinct exposed populations were identified, each representing a unique use of resources from the biosphere object or objects (Table 2-2). For most of the relevant pathways the exposure area is situated within a single biosphere object. However, exposure associated with cultivation of soil, where the source of radionuclides originates from fertilisation or irrigation, could also occur outside the boundaries of a biosphere object. This is also true for inhalation of combustion gases and fly ash that results from burning of bio-fuel.

The four land-use variants that served as credible bounding cases for exposure through a combination of pathways were:

Hunting and gathering (HG) – represents a community using natural ecosystems in the landscape for living space and food. The major exposure pathways are from foraging the landscape (fishing, hunting, and collecting berries and mushrooms), and from drinking surface water (from streams or lakes).

Infield–outland farming (IO) – represents self-sustained agriculture where infield farming of crops and livestock breeding are dependent on hay from wetlands (outland), used as fodder and organic fertiliser (manure). The major exposure pathways result from haymaking, from consumption of crops, meat and dairy products and from drinking water from either a dug well or from surface water in the biosphere object.

Draining and cultivating a mire (DM) – represents self-sustained agriculture in which wetlands are drained and used for agriculture (both crop and fodder production). The major exposure pathways are from growing food on land where radionuclides have accumulated for an extended period, and from drinking water from either a well (dug or drilled) or from surface water in the biosphere object.

Garden plot household (GP) – represents a household that is self-sustained with respect to vegetables and root crops produced through small-scale horticulture. The major exposure pathways result from fertilisation and food production on the garden plot, and from using either a well (dug or drilled) or surface water for drinking and irrigation purposes.

Table 2-2. Exposure route cases included in a safety assessment and the corresponding potentially most exposed groups (HG = hunter-gatherers, IO = infield-outland farmers, DM = drained-mire farmers, GP = garden-plot households).

Environmental medium	Exposure route case	Most exposed group	Comment
Atmosphere – outdoor	Inhalation of gases during breathing	HG, IO, DM, GP	Occurs during outdoor activities.
	Inhalation of dust during breathing	HG, IO, DM, GP	Occurs during outdoor activities.
Regolith – soil (inorganic)	Inadvertent soil ingestion[2]	GP	Occurs as a consequence of outdoor activities.
	External irradiation from the ground when staying outdoors	IO, GP	Occurs during all outdoor activities where there is soil.
	External irradiation from sediments[2]	HG	Occurs during outdoor activities such as fishing or bathing.
Regolith – peat	Inadvertent peat ingestion[2]	DM	Occurs as a consequence of outdoor activities.
	External irradiation from the ground when staying outdoors	HG, IO, DM	Occurs during all outdoor activities where there are peat soils.
	Indirect[1] contamination of the environment	GP	Inhalation of gas and dust originating from the burning of peat as biofuel.
	Indirect[1] contamination of soil	GP	Burning of peat as biofuel, and depositing ashes as fertilisers.
Well water	Ingestion of well water	IO, DM, GP	Drinking water from a dug well or a bedrock well.
	Indirect[1] contamination of crops and soils	GP	Irrigation of small-scale horticulture.
	Indirect[1] contamination of fauna that drinks water	IO, DM	Drinking water for livestock.
Surface water	Ingestion of water	HG, IO, DM, GP	Drinking water from a stream or a lake.
	Indirect[1] contamination of crops and soils	GP	Use of surface water for irrigation.
	Indirect[1] contamination of fauna that drinks water	IO, DM	Livestock drink or are watered with surface water.
Flora	Ingestion of crops, berries, mushrooms	HG, IO, DM, GP	Fundamental source of foodstuffs.
	External irradiation outdoors from vegetation[2]	HG	Exposure to radionuclides in the standing vegetation.
	Indirect[1] contamination of the environment from burning wood fuel	GP	Inhalation of gas and dust originating from the burning of wood as biofuel.
	Indirect[1] contamination of the environment by ash and manure	IO, GP	Burning of wood as biofuel, and depositing ashes as fertilisers. Depositing manure as fertiliser.
	Indirect[1] contamination of soils by fertilisation with aquatic flora	GP	Fertilisation with sea weed.
Fauna	Ingestion of terrestrial food	HG, IO, DM	Fundamental source of foodstuffs.
	Ingestion of aquatic food	HG	Eating fish.

[1] Indirect exposure routes are those that can lead to transfer of radionuclides from one environmental medium to another as a result of human activities.

[2] The exposure route case was evaluated with assessment-specific data and did not contribute significantly to dose (see Section 10.10 in SKB 2014a).

2.4.2 Non-human biota

Exposure pathways were also identified for non-human biota (SKB 2014e). As radionuclides are expected to be discharged into the sea, lakes or wetlands, environmental media at the exposure locations in natural ecosystems were used in the assessment (Table 2-3). Agricultural ecosystems were not considered relevant for the assessment, as the small exposed areas were expected to provide marginal habitats for domestic and/or immigrating species, with insignificant importance for the proliferation of populations in the area (see further discussion in SKB 2014e). Three different exposure routes were identified as relevant for the assessment; namely, exposure by ingestion, inhalation and external irradiation. However as exposure via ingestion and inhalation is not separated in the ERICA methodology, these two routes were combined in characterising the internal exposure route.

For each ecosystem, a set of organisms likely to get the highest exposure was identified. In most cases, ERICA reference organisms were found to represent species at the site reasonably well. However, a few site-specific organisms were also added to the set of exposed populations (e.g. freshwater microphytobethos and European otter). The selected organism types were:

Pelagic marine organisms living (or feeding[2]) in the sea water column (Wading bird, Mammal, Pelagic fish, Phytoplankton, Zooplankton, European otter and Ruddy turnstone).

Benthic marine organisms living on top of surface sediments (Benthic fish, Crustacean, Macroalgae and Vascular plant), or in marine surface sediments (Benthic mollusc and Polychaete worm).

Pelagic limnic organisms living in the water column (Amphibian, Bird, Mammal, Pelagic fish, Phytoplankton, Zooplankton) for lake and/or stream ecosystems.

Benthic limnic organisms living on top of surface sediments (Benthic fish, Gastropod, Vascular plant and Microphytobenthos), or in surface sediments (Bivalve mollusc, Crustacean and Insect larvae) in lake and/or stream ecosystems.

Terrestrial organisms living on top of surface peat (Amphibian, Bird, Bird egg, Flying insects, Gastropod, Grasses & herbs, Lichen & bryophytes, Mammal [Deer], Reptile, Shrub, Tree, European otter, and Ruddy turnstone), or in surface peat (Detritivorous invertebrate, Mammal [Rat], Soil Invertebrate [worm]) in mire ecosystems.

Table 2-3. Exposure pathways from source to potentially exposed populations of non-human biota. Pathways have been organised with respect to the natural surface ecosystem in which the exposure occurs.

Ecosyst.	Source		Exposure point (environmental medium)	Exposure route		Exp. population (Repr. Organsim.)
	Type	From		Internal irradiation	External irradiation	
Sea	Groundwater discharge or surface water	Bedrock or nearby basin	Sea water	x	x	Pelagic and benthic marine organisms
			Surface sediments		x	Benthic marine organisms
Lake/stream	Groundwater discharge or surface water	Bedrock or upstream area	Lake/stream water	x	x	Pelagic and benthic limnic organisms
			Surface sediments		x	Benthic limnic organisms
Mire	Groundwater discharge or surface water	Bedrock or upstream area	Surface peat	x	x	Terrestrial organisms

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[2] European otter and Rudy Turnstone were assessed for external exposure from living in the mire ecosystem and for internal exposure from feeding in the sea ecosystem.

3 The structure of the radionuclide transport model

The radionuclide model of the biosphere calculates the distribution (transport and accumulation) of radionuclides released from the geosphere to potentially effected surface ecosystems and the potential radiological consequences to humans and non-human biota in terms of radiological dose resulting from the accumulation of radionuclides in environmental media. The part of the model describing the distribution of radionuclides from the biosphere-geosphere interface to the different (abiotic) media of the surface ecosystems is here referred to as the "radionuclide transport model".

The mathematical approach used in the radionuclide model of the biosphere is that of compartment modelling. This approach has been described in detail by e.g. Sheppard (1962), Jacquez (1972) and Andersson et al. (1983). It assumes that a system can be sufficiently represented by a finite number of compartments or pools, each of which is homogeneous and connected to other compartments. In the radionuclide model of the biosphere described here, each compartment represents a radionuclide inventory associated with a physical or biological component in the modelled surface ecosystems. The dynamic change of each compartment is the result of radionuclide fluxes associated with mass fluxes of water, solids (including organic matter) or gas, or with diffusion.

As the end points of the radionuclide transport model are the concentrations of radionuclides in environmental media in affected areas of the landscape, which are subject to significant environmental change in time, the identification of these areas and their (current and future) properties is a prerequisite for the model (see the description of the Forsmark site and of Biosphere objects in the previous chapter). Radionuclide concentrations are then used to calculate doses to humans and dose rates to non-human biota that are exposed to radionuclides in the environment (see Chapters 9 and 10).

There are many biophysical, geo-chemical and physiological processes that affect transport of radionuclides in surface ecosystems, and these processes occur over a wide range of spatial and temporal scales. Due to the complexity of the surface ecosystems and the intrinsic uncertainties with regard to their evolution over large spatial and temporal scales, a mechanistic process-oriented modelling approach is not feasible in the context of a long-term safety assessment. Instead we have chosen to represent system complexity by a relatively small number of compartments and parameters, to describe the outcome of the underlying processes at larger scales. That is, discharge of radionuclides into the biosphere is assessed over thousands of years, the assessed area is considered at the scale of a coastal basin, a lake mire complex, or an agricultural field, and exposure and dose are calculated over a lifetime of future inhabitants. At these scales, it is assumed that most biogeochemical interactions can be approximated by equilibrium or steady-state conditions. Thus, ecosystem states are represented by average conditions, and fluxes of water, solid matter and gas are described as functions of aggregated empirical parameters, that are assumed to capture the outcome of the underlying processes.

The coarse spatial resolution of the compartmental structure introduces dispersion of radionuclide in the biosphere. This introduced dispersion can be assumed to be larger than the real physical dispersion within the regolith layers represented by the compartments. However, the introduced numerical dispersion is likely to be cautious, as it will result in radionuclides reaching the surface (where humans and non-human biota may be exposed) more rapidly than in a more finely resolved model. The effects of the horizontal and vertical resolution on transport, accumulation and potential exposure are discussed in SKB (2014a, Section 11.2.3).

In the following sections of this chapter we derive the physical compartments (Section 3.1) and the radionuclide fluxes (Section 3.2) that are used to simulate transport and accumulation of radionuclides in surface ecosystems. Figure 3-1 shows the compartments and radionuclide fluxes identified for a coupled lake and mire ecosystem. The complete model consists of a set of interconnected ecosystem models located at possible discharge locations for radionuclides released from the repository (discharge areas) and which evolve in time due to landscape development and climatic change. The links between aquatic and terrestrial ecosystems (within a local catchment) and the links between sea basins and lake-mire systems in different catchments are described in the next chapter (Chapter 4). The details, including the mathematical description of the radionuclide transport model and dose calculations are given in the following chapters.

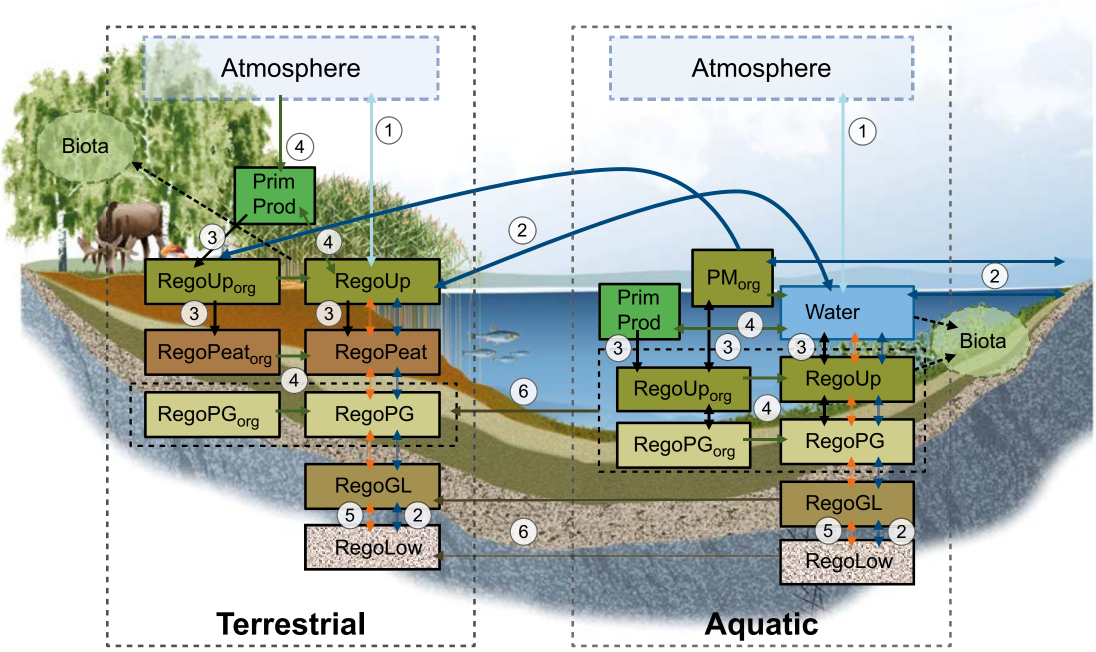

Figure 3-1. *A graphical representation of the radionuclide transport model used to simulate transport and accumulation in a discharge area with two natural ecosystems (black dotted boxes). Each box corresponds to a radionuclide inventory associated with a physical compartment. Arrows represents radionuclide fluxes between compartments and fluxes into and out of the system. Radionuclide fluxes are linked to mass fluxes of gas (1, light blue), water (2, dark blue) and solid matter (3, black), to transitions between inorganic and organic forms of radionuclides (4, green), to diffusion in soil pore water (5, orange), and to ingrowth of wetland vegetation (6, green). Dashed arrows represent concentration calculations not included in the dynamic model. The atmosphere serves as a source and sink of radionuclides.*

3.1 Model compartments

The conceptual models for marine, fresh water and terrestrial ecosystems (Aquilonius 2010, Andersson 2010, Löfgren 2010), the description of the Forsmark landscape and its development (Lindborg 2010), and the biosphere components identified to be important in safety assessments for radioactive waste disposal by SKB (2013), are the basis for the definition of ecosystem sub-models and the selection of compartments for the modelling of radionuclide transport. From the previous safety assessment of SFR 1, radiocarbon (C-14) was expected to be the major dose contributing radionuclide (SKB 2008). Therefore, specific attention was paid to the cycling of carbon in the system and its impact on transport of C-14 was considered in the identification of model compartments.

In practice, general biosphere components (SKB 2013) were interpreted in the context of ecosystems in the Forsmark landscape (Lindborg et al. 2013), which are potentially affected by radionuclides released from the repository. These included sea, lake and wetland ecosystem in biosphere objects, but excluded terrestrial upland ecosystems in higher terrain. Moreover, we aimed to define compartments that were meaningful and homogenous units with respect to physical, chemical and biological properties influencing radionuclide transport and accumulation. The initial analysis frequently resulted in aggregated biosphere components being split into several separate components (e.g. regolith was split into separate geological layers such as till, glacial clay, clay-gyttja and peat). On the other hand, components or processes that were considered to have small or insignificant radionuclide inventories were excluded from the model (e.g. consumption). Components with a fast turnover of radionuclides with respect to the time resolution of the model (e.g. the liquid phase) were in many cases combined with components having slower dynamics (e.g. the solid phase) applying equilibrium conditions for radionuclide exchange between them.

A minimum requirement for the final set of model compartments was that it included all the environmental media having a potential to expose future inhabitants and wildlife due to the release of radionuclides from the repository (i.e. surface water, surface sediments, air and the upper soil layer in wetlands and arable land, see Section 2.4 exposure pathways).

3.1.1 Aquatic ecosystems

Two major types of aquatic ecosystems are recognised as relevant; sea ecosystems (Aquilonius 2010) and freshwater or limnic ecosystems (Andersson 2010). For the purpose of the biosphere assessment, these two systems have been regarded as structurally and functionally similar and consequently a common set of abiotic and biotic components for all aquatic ecosystems was presented in Anderson (2013) (Figure 3-2).

All of the aquatic organism compartments identified by SKB (2013) were considered to have fast dynamics (i.e. a turnover within or over a few years). Elemental distributions from the sea and lake ecosystems at the site (Andersson 2010, Aquilonius 2010) suggest that pools in consumers (filter feeders, herbivores, carnivores and decomposers) are small, or very small, in relation to pools in sediments and water. Thus, simulated dynamics and concentrations of radionuclide inventories in water and surface sediments are not likely to be affected by the inclusion or exclusion of consumer compartments. Moreover, when effects of long-term release are to be evaluated, it is reasonable to use an equilibrium approach to calculate radionuclide concentrations in these organism groups. Therefore, filter feeders, herbivores, carnivores and decomposers were not included as model compartments. Instead radionuclide concentrations in consumer organisms were calculated from the concentrations in environmental media using equilibrium concentration ratios (CR), see Sections 10.2.1 and 11.2.2.

For a few elements (e.g. Mn in the sea and Cl in lakes) the inventory in primary producers at Forsmark made up a substantial fraction of that in water (Aquilonius 2010, Andersson 2010). Primary production is also the main source of organic carbon (and C-14) in sediments, and thus it was found appropriate to include aquatic primary producers (*PrimProd*) as an explicit model compartment.

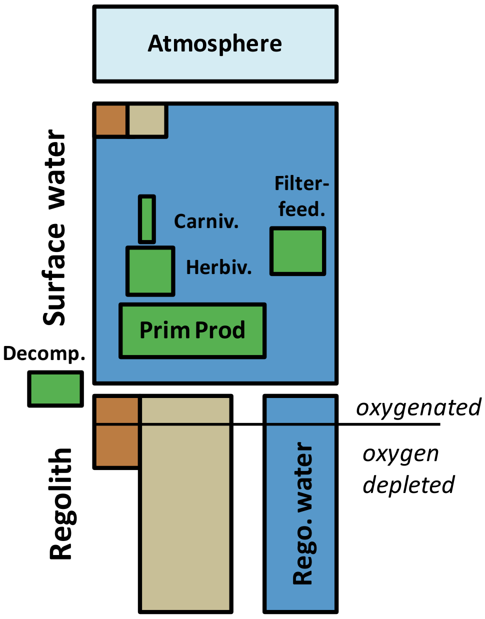

***Figure 3-2.** Biosphere components identified as important for the transport and fate of radionuclides in aquatic ecosystems (SKB 2013). Green indicate radionuclide inventories in organisms, blue indicates inventories of dissolved radionuclides. Grey and brown indicate radionuclide inventories associated with regolith solid states, i.e. adsorbed radionuclides (grey) and radionuclides stored in organic matter (brown). The horizontal line indicates the boundary between oxygenated and oxygen depleted aquatic sediments (regolith).*

SKB (2013) did not separate radionuclides in solution from radionuclides adsorbed to suspended particulate matter in the water column. To get a proper representation of the radionuclide flux associated with sedimentation, the radionuclide concentration on suspended particulate matter has to be considered. However, as the time resolution of the model is in the order of a year, it is reasonable to represent dissolved and adsorbed radionuclides with one compartment (*Water*), partitioning radionuclides between the inventories by assuming equilibrium. On the other hand, the radionuclide concentration in refractory organic particulate matter is not necessarily in equilibrium with the surface water (e.g. during periods of wave erosion and pronounced re-suspension). Thus, a compartment representing radionuclides stored in particulate organic matter in the water column was added as a model compartment (PM_{org}).

With radiocarbon in mind, a single compartment was not sufficient to represent storage (and subsequent release) of radionuclides in aquatic sediments. More specifically, compartments representing radionuclides stored in organic matter were identified as essential to describe the accumulation of radionuclides in aquatic sediments. The storage of carbon in sediments is a function of organic matter input and mineralisation (Andersson 2010). Mineralisation rate is highly dependent on oxygen, and in oxygen-depleted environments even decomposable (labile) organic matter may accumulate. Thus, the need for a vertical stratification of the aquatic sediments into oxygenated surface sediments and oxygen-depleted deep sediments was recognised (Figure 3-2). Given the time resolution of the model (year) it was assumed that radionuclides in pore water are in equilibrium with radionuclides adsorbed on solids, and the two components regolith and water in regolith were described with one compartment and an equilibrium partition coefficient to assign a fraction of the compartmental radionuclide inventory to an inventory dissolved in water and another to an inventory adsorbed on regolith.

The oxygenated top sediments were represented by two compartments, one representing radionuclides sorbed on solids/dissolved in pore water (*RegoUp*), and one representing radionuclides in organic matter ($RegoUp_{org}$). The sum of these two inventories reflects radionuclides in the biologically active aquatic sediment, where organisms living in these sediments can be exposed to ionising radiations.

Below the oxygenated layer, decomposition rates slow down and organic matter builds up in aquatic sediments. These regolith layers are recognised as marine or lacustrine gyttja, and the build-up of these sediments in Forsmark started after the last glaciation (i.e. they are post glacial sediments). As with the oxygenated layer (above), radionuclides stored in post glacial sediments were represented with two compartments, one representing radionuclides adsorbed on solids/dissolved in pore water (*RegoPG*), and one representing radionuclides in organic matter ($RegoPG_{org}$).

The older sediments buried under post glacial gyttja at Forsmark are depleted in organic matter (Andersson 2010), and, for the purpose of this assessment, the amount of radionuclides stored in organic matter in these layers was considered to be quantitatively insignificant. Older deposits at the site consist primarily of two distinctive classes; till (unsorted minerogenic material which rests on bedrock material), and glacial clay (fine-grained sorted minerogenic material which has sedimented on top of the till). The developmental history and properties (e.g. density, porosity, grain sizes, surface area, and hydraulic conductivity) are very different between these two geological layers, and therefore minerogenic old sediments were split into two compartments, namely glacial clay (*RegoGL)* and underlying till (*RegoLow)*. Both compartments represent radionuclides adsorbed on solids and radionuclides dissolved in pore water (as discussed above).

The last physical biosphere component that is identified by SKB (2013) is the atmosphere. As part of the SR-PSU safety assessment, SKB has developed a new conceptual and numerical model of the atmosphere, recognising a need to distinguish between near-surface and higher layers of the atmosphere (see Chapter 8). However, the dynamics in the atmosphere are fast (in the order of hours) compared with the dynamics in water (weeks/months) or regolith (years or more), and the exchange with the upper atmospheric layers is much greater than the exchange with surface waters. Thus, for modelling purposes, the atmosphere can be treated as a sink (or source) in the radionuclide transport model, and the concentration of volatile radionuclides in the atmosphere can be calculated using a steady state solution given the input from surface water.

3.1.2 Terrestrial ecosystems

Two major types of terrestrial ecosystems which are likely to be affected by radionuclides from a geological repository are recognised, namely mire ecosystems and agricultural ecosystems (Löfgren 2010). SKB has identified several biosphere components that should be considered in a safety assessment of a geological repository (SKB 2013). Figure 3-3 displays the listed biosphere components; high-lighting that humans and their livestock are major consumers in agricultural ecosystems.

Elemental distributions in terrestrial ecosystems in Forsmark suggest that pools in consumers (herbivores and carnivores) are small in relation to pools in organic soils, and very small compared with pools in mineral soil (Löfgren 2010). The amounts of elements stored in decomposers have not been quantified in Forsmark, but on similar sites these pools are small compared with the total soil pools, and the dynamics of soil animals and microbiota are typically very fast (days to months, Löfgren 2010). Thus, simulated dynamics and concentrations of radionuclides in regolith layers are not likely to be affected by inclusion or exclusion of organism compartments. Moreover, when effects of long-term release are to be evaluated, it is reasonable to calculate radionuclide concentrations of these organism groups with an equilibrium approach. For simplicity, terrestrial herbivores (including humans and livestock), carnivores (including humans) and decomposers were not included as model compartments in the radionuclide transport model, and radionuclide concentrations in consumer organisms were calculated from the concentrations in environmental media using equilibrium concentration ratios (CR, see Sections 9.2.1 and 10.2.2).

For a few elements (e.g. non-metals like Br and Cl) the inventory in terrestrial primary producers makes up a substantial fraction of that in soils in natural terrestrial ecosystems in Forsmark (Löfgren 2010). Primary production is also the main source of organic carbon in wetland peat, and, therefore, we found it appropriate to include terrestrial primary producers (*PrimProd*) as an explicit model compartment in the sub-model of a mire ecosystem.

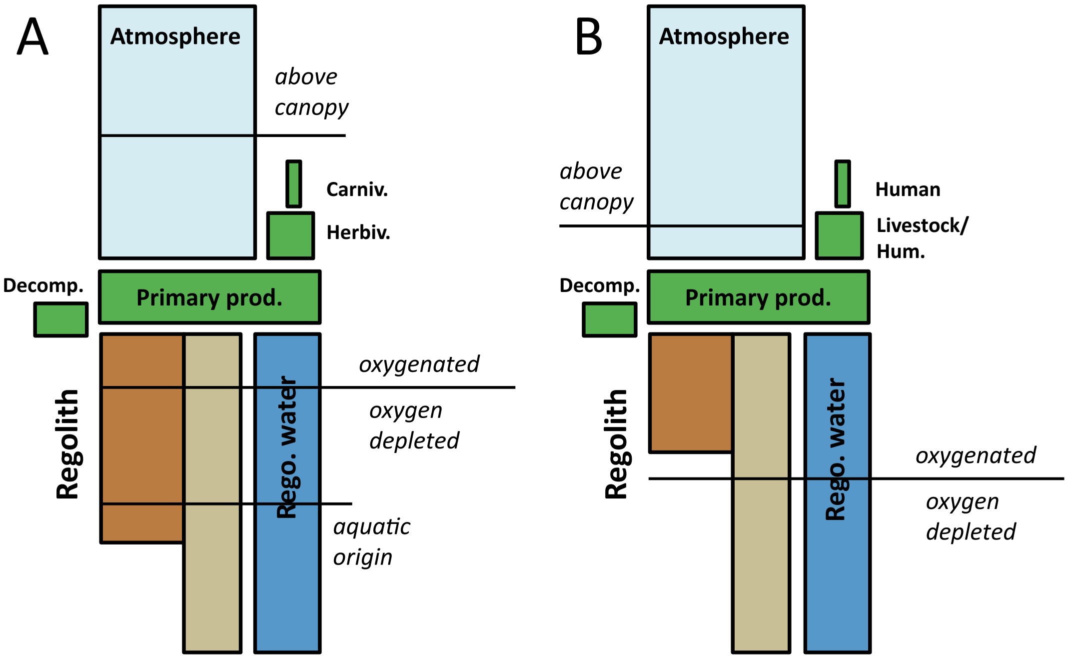

Figure 3-3. *Biosphere components identified as important for the fate of radionuclides in terrestrial ecosystems (SKB 2013). A) Components of a mire ecosystem. B) Components of an agricultural ecosystem. Green indicate radionuclide inventories in organisms, and blue indicates inventories in groundwater. Grey and brown indicate radionuclide inventories associated with solid state regolith, i.e. adsorbed on soil particles (grey) or stored in organic matter (brown).*

With radiocarbon (C-14) in mind we identified a need for separate compartments reflecting storage (and release) of radionuclides in organic matter. The accumulation of carbon in peat is a function of organic matter input and mineralisation. Mire peats have strong vertical gradients with respect to e.g. oxygen content, hydraulic conductivity and organic matter quality, and thus we recognised the need for separate compartments of radionuclides in acrotelm (oxygenated top layer) and catotelm (oxygen-depleted deep layer) peat. Radionuclides in pore water and radionuclides adsorbed on solids were assumed to be in equilibrium, and the two inventories were described with one compartment and an equilibrium partitioning coefficient for assigning fractions of radionuclide inventory as dissolved in pore water and adsorbed on solids.

The oxygenated top soil was represented by two compartments, one representing radionuclides adsorbed on solids/dissolved in pore water (*RegoUp*), and one representing radionuclides in organic matter ($RegoUp_{org}$). The sum of these two inventories reflects radionuclides in the biologically active peat, where roots take up elements and wetland dwelling organisms can be exposed to ionising radiation. Below the oxygenated layer, decomposition rates slow down and organic matter accumulates in peat that may grow to several metres depth below the groundwater table given sufficient time. As with the oxygenated layer, we can represent radionuclides stored in post-glacial sediments with two compartments, one representing radionuclides adsorbed on solids/dissolved in pore water (*RegoPeat*), and one representing radionuclides in organic matter ($RegoPeat_{org}$).

Peat in terrestrial wetlands is formed on top of geological layers that were deposited when the discharge area was covered by water (i.e. when the area was a sea basin or a lake). In order not to lose account of radionuclides that have accumulated in earlier successional stages, (or have accumulated during transport through these layers), we recognised the need to represent radionuclides stored in organic matter of marine and lacustrine sediments ($RegoPG_{org}$) and radionuclides adsorbed on solids/dissolved in pore water of the same layer (*RegoPG*), and the underlying minerogenic layers of glacial clay (*RegoGL*) and till (*RegoLow*).

Agricultural ecosystems are simulated to calculate potential exposures of future human inhabitants. Long-term accumulation of radionuclides in agricultural ecosystems over thousands of years is not considered likely at Forsmark, as organic soils in discharge areas cannot be sustainably cultivated for such time spans (Lindborg 2010). Instead, we have selected three scenarios to illustrate the exposure that could result 1) from draining an undisturbed lake-mire ecosystem (where radionuclides could have accumulated over thousands of years), 2) from using hay collected in the biosphere object as winter fodder, and manure as organic fertiliser, in an infield-outland agricultural system, and 3) from irrigating vegetables and potatoes in a garden plot with water over a generation (see Section 9). For these three highly stylised exposure scenarios we argue that it is sufficient to represent the dynamics of radionuclides in the soil-plant system with two regolith compartments, representing radionuclides in the upper biological active soil layers of cultivated soil (see below).

That is, elemental pools in decomposers and consumers (herbivores and carnivores) are expected to be small in relation to the pools in the uppermost layers of cultivated soil. Moreover the turnover of soil animals and microbiota is typically very fast. Also, the majority of plant biomass is expected to have a relatively fast turnover, as we are primarily considering annual crops (cereals, root crops and vegetables). Thus, for simplicity, organism components were not included as separate compartments in the agricultural ecosystem sub-models, and consequently radionuclide concentrations in crop, livestock and dairy products were calculated assuming they were in equilibrium with soil or fodder concentrations. However, a few non-metals (including plant nutrients) and rare elements are known to bio-accumulate, and we explicitly accounted for immobilisation in the plant component for transport calculations (Appendix C).

The oxygenated and unsaturated layer of cultivated soil was represented by two compartments; one representing radionuclides sorbed on solids, dissolved in pore water or bound in crop (*RegoUp*), and one representing radionuclides in soil organic matter ($RegoUp_{org}$). The sum of these inventories (less the pool stored in crop) reflects radionuclides in the biologically active layer of cultivated soil, where roots take up elements and soil organisms can be exposed to ionising radiation.

Some fractions of crop biomass can be considered refractory, (and have the potential to accumulate radionuclides in organic matter). However, we argue that the consequence of long-term radionuclide accumulation in refractory organic matter is already sufficiently handled in the wetland draining scenario (which has an initial inventory as a result of long-term accumulation). Thus, we did not explicitly represent the dynamics of organic matter with a slow or very slow turnover.

SKB (2013) identified an atmosphere compartment as important to consider for the assessment of radionuclides in terrestrial surface ecosystems. The dynamics in the atmosphere are fast (hours), and the exchange with upper atmospheric layers is much greater than the exchange with the upper regolith layers. For modelling purposes the atmosphere was treated as a sink (or source), and the concentrations of volatile radionuclides in the canopy atmosphere (and above) were calculated using a steady state solution (Section 8).

3.2 Processes affecting radionuclide transport

To identify processes that are important for transport and accumulation of radionuclides in natural and agricultural ecosystems in Forsmark, we first considered the conceptual model of the Forsmark landscape (Lindborg 2010), and the conceptual models presented for marine, freshwater and terrestrial ecosystems (Aquilonius 2010, Andersson 2010, Löfgren 2010). In all three types of ecosystem, fluxes of radionuclides were primarily associated with mass fluxes (of water, solid matter and gas), or with transitions between inorganic and organic forms of radionuclides (i.e. primary production and mineralisation of organic matter). The exchange of radionuclides between solid and liquid states was also highlighted as a key process for radionuclide transfer in all types of ecosystems.

Next, we considered which of these processes may be of quantitative importance for the flux of radionuclides between the identified model compartments (above), accounting for the spatial configurations of the system components. That is, attention was focused on fluxes between compartments that were in proximity to (or in physical contact with) each other. The identified processes were checked against the processes in SKB's FEP list (SKB 2013) to make sure that any processes of importance for a safety assessment of a geological repository of radioactive waste had not been overlooked. After a few iterations we converged on a set of processes and a compartmental topology that was reasonably simple, and that arguably captures the main mass fluxes of the ecosystem, and includes the transformation of radionuclides between organic and inorganic forms.

Thirteen processes representing fluxes of radionuclide between model compartments *within* aquatic, mire or agricultural ecosystems were identified (Table 3-1). We then added two processes that would reflect the transfer of radionuclides *between* natural ecosystems (within a biosphere object), or between biosphere objects. Finally we added three processes associated with human transformation and use of land, and included the sink/source associated with radioactive decay and ingrowth of progeny radionuclides.

Below we briefly describe the radionuclide fluxes included in the ecosystem models (sorted in categories according to Table 3-1). The audit process for these fluxes with respect to the SKB FEP-list is briefly summarised in Appendix E).

3.2.1 Bioturbation

Bioturbation refers to translocation of radionuclides by mixing of sediment and soil particles in aquatic and terrestrial regolith layers. Mixing of aquatic sediments, by, for example, sediment dwelling organisms, leads to improved oxygenation. When the depth of the upper sediment layer is reduced during episodes of resuspension, bioturbation causes the oxygen front to penetrate into underlying sediments, resulting in an apparent translocation of radionuclides from anoxic regolith layers (i.e. $RegoPG$, $RegoPG_{org}$) to the upper, biologically active, sediment layer ($RegoUp$, $RegoUp_{org}$). In mires, bioturbation is considered to be absent, mainly due to the water-saturated, anoxic conditions. This is seen as undisturbed peat profiles reflecting the successive incorporation of organic material of different dominating species and decomposability.

The combined effects of plowing and soil animal activity (e.g. earthworms burrowing) mix cultivated soil. When drained mire and lacustrine sediments are cultivated, radionuclides from deeper regolith layers (*RegoPeat*, $RegoPeat_{org}$, *RegoPG*, $RegoPG_{org}$, and *RegoGL*) are redistributed and combined with radionuclides in the upper soil horison (*RegoUp*, $RegoUp_{org}$) into one biologically active soil layer ($RegoUp_{DM}$, $RegoUp_{org,DM}$).

3.2.2 Plant uptake

Plant uptake is fixation of inorganic radiocarbon and other radionuclides in plant biomass, as a result of net primary production. Uptake can be either active (metabolic) or passive (non-metabolic), and includes incorporation into plant biomass and adsorption onto plant tissue. In aquatic ecosystems, plants (*PrimProd*) take up dissolved radionuclides from the surrounding water (*Water*). In terrestrial (mire and agricultural) ecosystems, plants take up radionuclides from the upper soil layers (*RegoUp*) through root uptake, and from the canopy atmosphere (*Atmosphere*) through leaf uptake. Atmospheric dry deposition on plant leaves is implicitly included in root uptake, as radionuclide concentrations in dust are similar to those in top soil in the biosphere object (i.e. the upper soil layer is the source term for dust). Effects of plant uptake on radionuclide distribution in agricultural systems are accounted for by assuming equilibrium between plant and soil (see Appendix C)

3.2.3 Litter respiration/release

When organic matter is metabolised by decomposers and grazers, labile organic carbon is transformed to inorganic carbon (CO_2 under aerobic conditions). Litter respiration/release refers to the release of radionuclides from fast (within year) mineralisation of aquatic and wetland net primary production. Through this process radionuclides stored in plant biomass (*PrimProd*) are released to the water column (*Water*) and to the upper peat layer (*RegoUp*), in aquatic and mire ecosystems respectively.

3.2.4 Litter production

Most of the biomass produced annually is metabolised by grazers and decomposers (see litter respiration above). However, a part of the biomass production is made up of refractory organic matter. This decomposes much more slowly, and consequently a fraction of the biomass production will contribute to the build-up of organic matter in the upper regolith layer. Through the process of litter production, radionuclides stored in plant biomass (*PrimProd*) are transferred to the upper regolith layers ($RegoUp_{org}$), in aquatic, mire and agricultural ecosystems.

3.2.5 Regolith Mineralisation

When refractory organic matter is metabolised by decomposers, carbon is transformed to inorganic carbon (CO_2 under aerobic conditions and CO_2 or CH_4 under anaerobic conditions). Regolith mineralisation refers to the release of radionuclides from slow decomposition of organic matter in soil and sediments. Through this process, radionuclides stored in organic regolith layers ($RegoUp_{org}$, $RegoPeat_{org}$, $PegoPG_{org}$) are released to the pore water of the corresponding regolith layer (*RegoUp, RegoPeat, PegoPG*) in aquatic, mire and agricultural ecosystems.

Table 3-1. Overview of processes explicitly included as radionuclide fluxes in the transport model. Processes have been categorised with respect to the driving forces, namely: mass-flux of solid (MS), water (MW) or gas (MG), and incorporation into or release from organic matter due to photosynthesis (PP) and mineralisation (Min). Y indicates inclusion as a dynamic flux (between compartments) in the aquatic, mire or agricultural sub-model. * indicates that the process is accounted for assuming that a steady-state has been reached.

	Type	Radionuclide flux			SKB biosphere FEP (process)
Process		**Aqua**	**Mire**	**Agri**	
Biological					
Bioturbation	MS	Y		Y	1) Bioturbation
Plant (root) uptake	PP	Y	Y	*	15) Uptake 13) Primary Production
Litter respiration / release	Min	Y	Y		2) Consumption, 3) Death 4) Decomposition
Litter production	PP	Y	Y	Y^{1}	3) Death 5) Excretion, 12) Particle release/trapping
Regolith Mineralisation	Min	Y	Y	Y	4) Decomposition
Vegetation ingrowth^{BTW}	MS	Y	Y		13) Primary production 33) Covering
Water bound					
Advectiv Horizontal^{BTW}	MW	Y	Y		32) Convection
Advective vertical	MW	Y	Y	Y^{2}	32) Convection
Diffusion (vertical)		Y	Y		32) Convection
Water uptake	MW			Y	36) Import
Solid liquid phase dissociation		*	*	*	26) Reactions 27) Sorption/desorption
Sediment-bound					
Sedimentation	MS	Y			34) Deposition
Resuspension	MS	Y			39) Resuspension
Burrial	MS	Y	Y		34) Deposition
Gas transport					
Degassing	MG	Y	Y	Y	24) Phase transition, 32) Convection
Gas uptake	MG	Y	Y		24) Phase transition
Human behaviour					
Drain/cultivation^{BTW,INIT}				Y	
Fertilisation^{BTW}	MS			Y	16) Ant. Rel. 38) Relocation
Irriggation	MW			Y	16) Anthropogenic release
Radiological					
Radionuclide decay and ingrowth		Y	Y	Y	41) Decay

BTW = flux between ecosystems or biosphere objects; INIT = affects initial inventory (rather than flux); 1 = flux from mire ecosystem included as source term in IO agriculture; 2 = processes named leaching in text.

3.2.6 Vegetation ingrowth

Wetland vegetation will colonise sea and lake shores and bottoms, and the establishment of mosses, emergent and floating-leaved vegetation is associated with soft bottoms, where organic matter from refractory litter accumulates. The cover of vegetation gradually develops from the edges towards the middle of a lake, ultimately covering the lacustrine sediments. The process vegetation ingrowth thus represents the translocation of radionuclides in aquatic regolith layers (*RegoUp, RegoUp*$_{org}$*, RegoPG, RegoPG*$_{org}$*, RegoGL, RegoLow*) to the corresponding mire regolith layer (*RegoPG, RegoPG*$_{org}$*, RegoGL, RegoLow*), caused by peatland succession.

3.2.7 Advective horizontal transport

This process represents advective transport of radionuclides by lateral bulk movement of water. Horizontal transport of radionuclides by advection occurs through water exchange between adjacent sea basins (*Water*), and through stream and diffuse water flow from lakes and mires (*Water*) to recipients of surface waters (*Water*).

Within biosphere objects advective horizontal transports primarily occurs from lake/stream water (*water*) to the upper peat layer (*RegoUp*) of the adjacent mire, (through flooding and diffuse fluxes), and in the opposite direction from transient accumulations of surface water and peat pore-water to the adjacent lake/stream, (by diffuse fluxes of water). Similarly, in biosphere objects that lack persistent surface water bodies (i.e. a lake or a stream), diffuse flux of water from the upper peat layer (*RegoUp*) transports radionuclides to lake-mire complexes down-stream (*Water/RegoUp*).

The horizontal transport of water in sub-surface layers, (i.e. in consolidated peat, lacustrine sediments, glacial clay and till), is expected to be small compared with vertical fluxes between regolith layers (occuring over distances that are orders of magnitude shorter) in future lakes and wetlands in Forsmark (Werner et al. 2014). For simplicity, horizontal transport below the top regolith layer between ecosystems or between biosphere objects is not explicitly represented in the radionuclide transport model.

3.2.8 Advective vertical transport

This process represents transport of radionuclides by vertical bulk movement of water. Groundwater movement in soil pores, and fluxes of water across the sediment-water boundary, are driven by hydraulic pressure gradients and flow pathways are determined by regolith properties. This process is considered to be the quantitatively most important mechanism for vertical transport of radionuclides between pore water in adjacent regolith layers (i.e. between *RegoUp, RegoPeat, RegoPG, RegoGL* and *RegoLow* within aquatic and mire ecosystems), and between aquatic sediments (*RegoUp*) and the overlying water column (*Water*).

3.2.9 Leaching

Leaching refers to the export of radionuclides associated with downward flux (percolation) of meteoric water from unsaturated top soil in agricultural ecosystems (*RegoUp*). Percolation is the difference between precipitation and evapotranspiration, and it is affected by climate (precipitation and temperature), crop system (e.g. primary production, water use efficiency, and leaf area index) and soil properties (e.g. infiltration rate, field capacity, and hydraulic conductivity).

3.2.10 Water uptake

During dry periods of the year, diffusion and capillary rise cause an upward flow of groundwater to unsaturated soil layers. In discharge areas of deep groundwater, water uptake will transport radionuclids in groundwater from deep soil layers (e.g. *RegoGL* or *RegoLow*) to the overlying cultivated and biologically active layer (*RegoUp*), where they can be taken up by fodder and food crops. This process is only considered to be relevant for agricultural soil that has been created by draining and cultivating the uppermost peat and sediment layers in areas where groundwater is discharged.

3.2.11 Diffusion (vertical)

Diffusion refers to the transport of radionuclides due to random movement of molecules in solution or gas. Diffusion results in a net movement of radionuclides from a region of high concentration to a region of low concentration. As the rate of diffusion is strongly dependent on distance, diffusion is only considered to be a quantitatively important at scales below metres. Consequently, this process was only considered a relevant mechanism for vertical transport of radionuclides between adjacent regolith layers with large (>100,000 m^2) contact areas. That is the radionuclide flux by diffusion was represented between *RegoUp, RegoPeat, RegoPG, RegoGL* and *RegoLow* within aquatic and mire ecosystems, and between the upper aquatic regolith layer (*RegoUp*) and surface water (*Water*). (Diffusion of radionuclides in gas is included in the process degassing, see below).

3.2.12 Solid-liquid phase dissociation

In regolith layers, and in the water column, there will be an exchange of elements (including radionuclides) between the solid state (the soil/sediment matrix and suspended particulate matter), and the liquid state (pore water or the water column). This exchange is driven by several different processes, including ion exchange, complex formation, and precipitation/dissolution equilibrium. The exchange of radionuclides between the solid and liquid state is considered to be fast (days) as compared to the model resolution (year), and thus this process is described by an equilibrium partitioning coefficient, rather than by dynamic fluxes. The process is represented in all regolith layers (i.e. *RegoUp*, *RegoPeat*, *RegoPG*, *RegoGL* and *RegoLow* within aquatic and mire ecosystems) and in the water column (*Water*).

A solid/liquid distribution coefficient K_d is commonly used to express the ratio describing the distribution of a radionuclide between a solid and an aqueous phase:

$$K_d = \frac{AC_{solid}}{AC_{solution}} \tag{3-1}$$

where

K_d is the solid/liquid distribution coefficient, [$\text{m}^3\ \text{kg}_{\text{DW}}^{-1}$],

AC_{solid} is the activity concentration of a radionuclide in the solid (solid phase) [$\text{Bq kg}_{\text{DW}}^{-1}$], and

$AC_{solution}$ is the corresponding activity concentration in the solute (aqueous phase) [Bq m^{-3}].

3.2.13 Sedimentation

Particles suspended in the water column will be pulled downwards by gravity. Consequently there is a tendency for particles in suspension to settle out on the top of the uppermost sediment layer in aquatic ecosystems. The process sedimentation refers to the flux of radionuclide on the surface of suspended particulate matter (*Water*) or stored in suspended particulate organic matter (PM_{org}) to the corresponding inventory in the upper aquatic sediments (*RegoUp*, $RegoUp_{org}$).

3.2.14 Resuspension

Sediment particles will be stirred by turbulent water movement, resulting in an upward movement of sediment particles from the surface sediments. The process resuspension refers to the flux of a radionuclide in aquatic surface sediments (*RegoUp*, $RegoUp_{org}$) to particulate organic matter (PM_{org}), and to the surfaces of suspended particulate matter in the water column (*Water*).

3.2.15 Burial

If the sedimentation of particulate matter is larger than the resuspension, or if the production of litter is larger than the rate of mineralisation, there is a net accumulation of sediments, and the total depth of the upper regolith layer will grow. As the depth of the upper biological active zone is limited by oxygen supply from above, the deepest part of this zone will become anoxic as a consequence of burial. Radionuclides associated with buried sediments will consequently be translocated from the upper sediments (*RegoUp*, $RegoUp_{org}$) to the oxygen-depleted sediment layers below (*RegoPG*/$RegoPG_{org}$ for aquatic systems and *RegoPeat*/$RegoPeat_{org}$ for mire ecosystems).

3.2.16 Degassing

There will be a gas exchange between the surfaces of open water and soil pore water, on the one hand, and the overlying atmosphere, on the other hand. The net rate of water-air gas exchange depends on the concentration gradient over the water-air boundary layer and a combination of turbulence and diffusion properties of the two layers. Degassing refers to the radionuclide flux caused by the gross exchange of dissolved gas in the water column (*Water*), or in the soil pore water of surface peat (*RegoUp*), with the atmosphere (sink), for aquatic and mire ecosystems, respectively. For agricultural ecosystems the soil (*RegoUp*) is unsaturated, and degassing is assumed to be limited by gas diffusion through air-filled soil pores.

3.2.17 Gas uptake

There will be a gas exchange between the water/soil pore water and the overlying atmosphere (see above). Gas uptake refers to the radionuclide flux caused by the gross exchange of gas between the atmosphere (source) and gas dissolved in the water column (*Water*), or in the soil pore water of surface peat (*RegoUp*), for aquatic and mire ecosystems, respectively. Gas uptake is not included in the simplified representation of agricultural ecosystems.

3.2.18 Drain/cultivation

Draining lakes and wetlands will lower the groundwater table, and organic regolith layers above the water table will subside due to compaction and decomposition. The soil properties of the unsaturated layer (e.g. the porosity and density) will change as a consequence of subsidence and mixing of the parent material (see bioturbation above). The process drain/cultivation thus refers to the transfer of radionuclides from wetland regolith layers (*RegoUp*, $RegoUp_{org}$, *RegoPeat*, $RegoPeat_{org}$, *RegoPG*, $RegoPG_{org}$, and *RegoGL*) to the unsaturated layer of cultivated soil ($RegoUp_{DM}$, $RegoUp_{org, DM}$) in an agricultural ecosystem. This transfer or radionuclides is handled as an initial condition for radionuclide inventories in the cultivated soil, and the process does not affect radionuclide inventories in natural ecosystems. That is, the drain and cultivate land-use variant evaluates the consequences of cultivation at any point in time (when this is possible), under the condition that the natural ecosystems have been undisturbed to this point (Section 7.2).

3.2.19 Fertilisation

The key principle behind the agrarian infield–outland system is to fertilise arable land with the nutrients from meadows and pastures, by the use of animal manure as organic fertiliser. The process fertilise represents the input of radionuclides from mire vegetation (source) to the organic and inorganic component of infield soil ($RegoUp_{IO}$, $RegoUp_{org,IO}$), originating from wetland hay used as animal fodder. This transfer of radionuclides acts as a source term for cultivation of otherwise unaffected land (see Section 7.1 for details).

Fertilisation may also result in an entry of radionuclides in seaweed from the sea ecosystem, or of radionuclides in ash from combustion of deep peat or biomass fuel harvested in the biosphere object. The process represents the input of radionuclides from seaweed, mire vegetation or deep peat (source) to the organic and inorganic component of cultivated soil ($RegoUp_{GP}$, $RegoUp_{org,GP}$). This transfer of radionuclides acts as a source term for cultivation of otherwise unaffected land (see Section 7.3 for details).

3.2.20 Irrigation

Radionuclides in water can be transferred to soil and vegetation through irrigation of arable land. The need for irrigation will primarily depend on the balance between precipitation, crop water need and groundwater supply during the vegetation season. Under the current climatic conditions, irrigation of potatoes and vegetables cultivated outdoors are plausible in a future Forsmark area. Thus, irrigation refers to the input of radionuclides from well water (source) to cultivated soil ($RegoUp_{GP}$). For simplicity, the removal of radionuclides by irrigation water is cautiously ignored as a sink for radionuclides discharging into natural ecosystems.

3.2.21 Radionuclide decay/ingrowth

Radioactive decay is the process by which a nucleus of an unstable atom loses energy by emitting particles (alpha or beta) or gamma rays of ionising radiation. This decay will cause an atom with one type of nucleus, (called the parent radionuclide), to transform into an atom with a nucleus in a different state, or to a different nucleus containing different numbers of protons and neutrons. Either of these products is named the progeny nuclide. Radioactive decay is a stochastic process and the chance that a given atom will decay is constant over time. Radioactive decay is treated as a sink term in all model compartments. For decay that results in a radioactive progeny nuclide, the radioactive decay is matched by a source term representing ingrowth. Ecolego, the software used for dynamic simulation of transport and accumulation, has the capability of handling chains of decay and ingrowth.

4 Transport by advection and diffusion in water

Radionuclides from a geological repository reach the surface ecosystems by discharge of deep groundwater, and the flux of water is the major process for transport of radionuclides within and between ecosystems. Hydrological and oceanographic modelling contributed in four major ways to the biosphere transport modelling: 1) bedrock hydrogeology and surface hydrology models were used in combination to identify discharge areas for groundwater from the SFR repositories, and to outline biosphere objects, 2) vertical and horizontal water fluxes between compartments within each biosphere object were derived from detailed hydrological modelling, 3) oceanographic models were used to determine the exchange of water between sea basins in the submerged period, and 4) the stream network connecting biosphere objects in the land period was determined based on surface hydrological modelling. The methods and results from hydrological and oceanographic modelling are described in detail in Werner et al. (2014) and the parameters derived are presented in Grolander (2013). The identification of discharge areas and biosphere objects are described in chapter 6 in SKB (2014a).

This chapter describes how water-borne transport by advection and diffusion is represented in mathematical terms with respect to aquatic and mire ecosystems in the biosphere radionuclide model. The chapter also shows the structure of the model with respect to the flux of water within and between ecosystems and biosphere objects. Export of radionuclides with percolating water in cultivated soils (i.e. leaching) is described in Chapter 7 (Agricultural ecosystems).

4.1 Advective transport within ecosystems

The vertical fluxes of groundwater for each biosphere object were described using water-balances for three points in time, representing submerged conditions (3000 AD), conditions in newly emerged lake-mire systems (5000 AD) and fully terrestrial conditions (11,000 AD). The original water balances described the vertical flux of water across boundaries of MIKE SHE calculation layers, as well as the horizontal flux across the boundary of the biosphere object for each calculation layer. During the lake-mire stage the biosphere object was partitioned into two parts, a lake and a mire surrounding the lake (Figure 4-1), and for this stage the water balances included horizontal fluxes across the lake-mire boundaries within the object (see Werner et al. 2014).

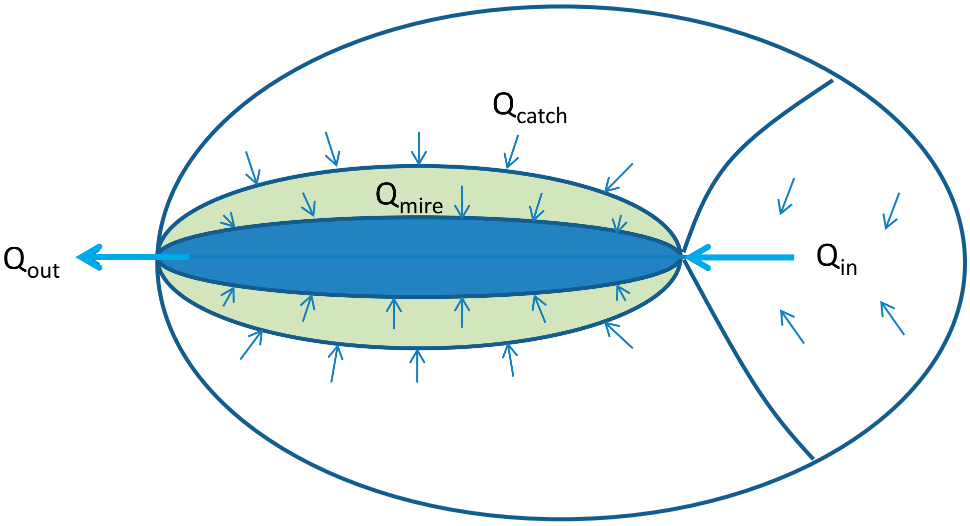

Figure 4-1. *Conceptual model underpinning water balances used to describe the surface-water flows in biosphere objects in the lake-mire stage. Thin blue arrows represent the net water flux into and through the mire component (green area). Thick blue arrows represent stream water entering and exiting the lake (dark blue area).* Q_{in}*,* Q_{out} *and* Q_{catch} *corresponds to* $WF_{stream,}$ $WF_{downstream}$*, and* WF_{catch} *in Figure 4-2. Similarly,* Q_{mire}*, corresponds to the difference between .* $WF_{up,wat,ter}$ *and* $WF_{wat,up,ter}$*, as defined in Figure 4-2. Figure from Werner et al. (2014).*

The geological (and biological) boundaries of regolith compartments did not exactly match the limits of MIKE SHE calculation layers, and water fluxes derived with MIKE SHE were mapped to compartments in the radionuclide model maintaining the water balance in each compartment. The horizontal extent of biosphere objects was on the scale of hundreds of metres, whereas the depth of regolith layers was on the metre scale. Consequently, the transfer rate of water in the vertical direction (water flux divided by travel length) was typically two orders of magnitude larger than the transfer rate in the horizontal direction, for all but the top MIKE SHE calculation layer (see Section 7.6 in Werner et al. 2014). For simplicity, horizontal advective transport (within or between biosphere objects) was only represented for the surface compartments (i.e. surface peat and surface water, Figure 4-2). Transport from the lowest regolith layer back to the bedrock was cautiously ignored. Excluding sub-surface horizontal transport out of the biosphere object was also considered a cautious simplification, as this resulted in all radionuclides released from the bedrock being transported towards the upper regolith layers (and surface water), where humans and biota may be exposed. The detailed water balance results, and the methodology used to calculate water fluxes for compartments of the biosphere radionuclide model, are described in Werner et al. (2014).

In total, eight vertical fluxes of radionuclides are included in the radionuclide transport model for each of the aquatic and mire ecosystems (Figure 4-2). For any point in time, the flux of radionuclides from compartment *i* to *j*, $Adv_{i,j}$ [Bq y^{-1}], is calculated as the product of the advective flux of water from *i* to *j* and the pore water activity concentration in compartment *i:*

$$Adv_{i,j} = WF_{i,j}\, AC_i^D \tag{4-1}$$

where

$WF_{i,j}$	is the advective flux of water between compartment *i* and *j* [m^3 y^{-1}],
AC_i^D	is concentration of dissolved radionuclides in water in compartment *i* [Bq m^{-3}], and
i and *j*	indicate adjacent stacked regolith compartments (see Figure 4-2).

The vertical advective flux of water from compartment i to compartment j, $WF_{i,j}$ [m^3 y^{-1}], is calculated by multiplying the surface area of the regolith stack in the aquatic or mire ecosystem of the biosphere object by the corresponding area-specific water flux. This is done separatly for four combinations of ecosystem types and development stages (see below):

$$WF_{i,j,k} = q_{i,j,k}\, area_{obj,k} \tag{4-2}$$

where

$q_{i,j,k}$	is the area specific water flux from compartment *i* to *j* [m^3 m^{-2} y^{-1}],
$area_{obj,k}$	is the area of the aquatic sediments[3] or the terrestrial part of the object [m^{-2}], and
k	indicates the period and ecosystem: sea, lake at isolation, terrestrial at isolation and terrestrial at mire completion.

Thus four sets of area-specific water fluxes are estimated for each biosphere object, namely for submerged aquatic conditions ($q_{i,j,sea}$ sea ecosystem only), for the lake-mire complex in the early land period ($q_{i,j,lake/ter,iso}$, both lake and mire ecosystems), and for fully terrestrial conditions ($q_{i,j,ter,end}$, mire ecosystem only) (Werner et al. 2014). For the transport calculations, it is assumed that vertical water fluxes are stable in the submerged period, before land starts to emerge ($WF_{i,j,sea}$), and in the late land period, when all land has emerged and the mire expansion has been completed ($WF_{i,j,ter,iso}$). Furthermore, it is assumed that water fluxes are intermediate in transition periods between fully submerged and terrestrial conditions. Thus, point estimates are used for water fluxes in the submerged and the late land period, whereas fluxes are estimated by linear interpolation in transition periods, i.e. during the sea-lake transition, sea-mire transition, and mire expansion (Table 4-1, Appendix G for details).

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[3] When the landscape is submerged, accumulation of sediments only occurs in the deeper central parts of the sea basin (see Chapter 5). For simplicity accumulation of radionuclides in aquatic regolith was only modelled for the central part of the basin in the submerged period, and the size of this area was approximated as being the same as the future lake basin or the terrestrial object (for objects that did not form a lake) (dark blue areas in Figure 4-4a).

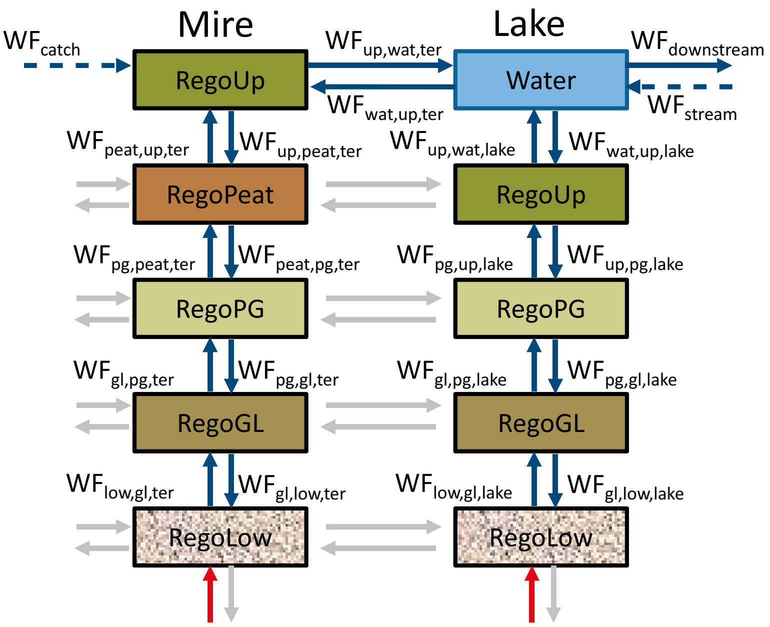

Figure 4-2. *Water fluxes (solid blue arrows) used to calculate advective transport of radionuclides within and out of a biosphere object at the lake-mire stage. All biosphere objects start as sea basins (structure as for the right regolith stack) and end as a mire (structure as for the left stack) with or without a stream (right stack). Red arrows indicate radionuclide fluxes from the bedrock (source term in primary object). Blue dashed arrows indicate water entering the object. If the object is downstream an area receiving radionuclides these fluxes represents radionuclide source terms (see Section 4.2.2). Grey arrows represent water fluxes that were not represented in the radionuclide transport model.*

In lakes the area specific flux ($q_{i,j,lake}$) is assumed to be stable throughout the life time of the lake/stream ecosystem, and the vertical water flux in the lake, $WF_{i,j}$ [$m^3\ y^{-1}$], decreases as mire vegetation expanded into the lake basin, and the area of open water is reduced:

$$WF_{i,j,lake} = q_{i,j,lake}\, area_{obj,aqu}\,(t) \tag{4-3}$$

where

$q_{i,j,lake}$	is the area specific water flux from compartment i to j [$m^3\ m^{-2}\ y^{-1}$], and
$area_{obj,aqu}$	is the surface area of the lake/stream ecosystem, which decreases with time [m^{-2}].

Table 4-1. Vertical groundwater fluxes in transition periods. During each transition period the vertical flux in the ecosystem (aquatic or mire) changes linearly with time between the values at the start and the end of the period. 1 = biosphere object developing through a lake period. All individual parameters are listed in Appendix G.

Ecosyst	Transition	Transition Period Start	End	Water flux ($m^3\ y^{-1}$) Start	End
Aquatic	Sea to Lake[1]	threshold_start	threshold_isolation	$WF_{ij,sea}$	$WF_{ij,lake,iso}$
Aquatic & Mire	Sea to Mire	threshold_start	threshold_isolation	$WF_{ij,sea}$	$WF_{ij,ter,iso}$
Mire	Mire expansion	threshold_isolation	threshold_end	$WF_{ij,ter,iso}$	$WF_{ij,ter,end}$

4.3 Advective transport between ecosystems and biosphere objects

4.3.1 Transport between sea basins

During the submerged period, most groundwater from the repository is expected to be discharged into a relatively limited area in the sea, north of SFR (Odén et al. 2014). Radionuclides from the bedrock will have to pass through deep sediments before reaching surface sediments and water in a relatively small sea basin. This basin (157_2) is presently connected to several small neighbouring basins (north and south of the pier), and to the larger basin 116, which in turn is connected to basins further away in Öregrundsgrepen (Figure 4-3b). Thus, through the exchange of surface water, radionuclides discharged to the surface during submerged conditions will first reach nearby basins, then Öregrundsgrepen, and ultimately end up in the Baltic Sea.

As land emerges from the sea both the coastal landscape and the flow paths change (see SKB 2014a Figure 6-9). For example, a thousand years from now there is no direct connection between basin 157_2 and basin 116 or to basins south of the pier (Figure 4-3a). After 3000 AD coastal basins become significantly shallower, straits narrow up and several basins successively become isolated (Werner et al. 2014, Grolander 2013).

The horizontal advective transport of radionuclides out of a sea basin depends on the horizontal water flux and the total activity concentration in sea water (including radionuclides dissolved in water, adsorbed to particulate matter or stored in organic matter, see Chapter 5 Aquatic ecosystem). The total export of radionuclides from a sea basin, $Adv_{water,out}$ [Bq year^{-1}], is expressed as the product of the activity concentration in water and the flux of water to the adjacent basin, summed over all basins in contact with the biosphere object:

$$\begin{aligned} Adv_{Water,out} &= \sum_i WF_{out,i} AC_{Water}, \; i = adjacent\ sea\ basins \\ Adv_{Water,org,out} &= \sum_i WF_{out,i} AC_{Water,org}, \; i = adjacent\ sea\ basins \end{aligned} \tag{4-4}$$

where

$WF_{out,i}$ is the advective flow of water from the basin to the adjacent basin i [m^3 y^{-1}].

AC_{Water} is the radionuclide concentration in water, including dissolved and adsorbed radionuclides [Bq m^{-3}], and

$AC_{Water,org}$ is the concentration of radionuclides in organic particulate matter [Bq m^{-3}].

Similarly, the total flux of waterborne radionuclides into a sea basin during submerged conditions is given by summing over all transport directed towards the object from the adjacent basins. As the landscape emerges out of the sea, sea basins may also receive radionuclides from stream water or as diffusive fluxes / runoff from terrestrial areas (see below).

The exchange of water between marine basins ($WF_{out,i}$) was estimated with MIKE 3 every thousand years. These parameter estimates were combined into time series for each sea basin (see Grolander 2013). The rate of water exchange between these point estimates was calculated using linear interpolation.

4.3.2 Transport between objects in the land period

Once the landscape has emerged out of the sea, the transport of radionuclides with surface water between biosphere objects is determined by the topography and the stream network. This means that transport between biosphere objects in the land period is unidirectional, from an upstream object to a downstream object (Figure 4-4).

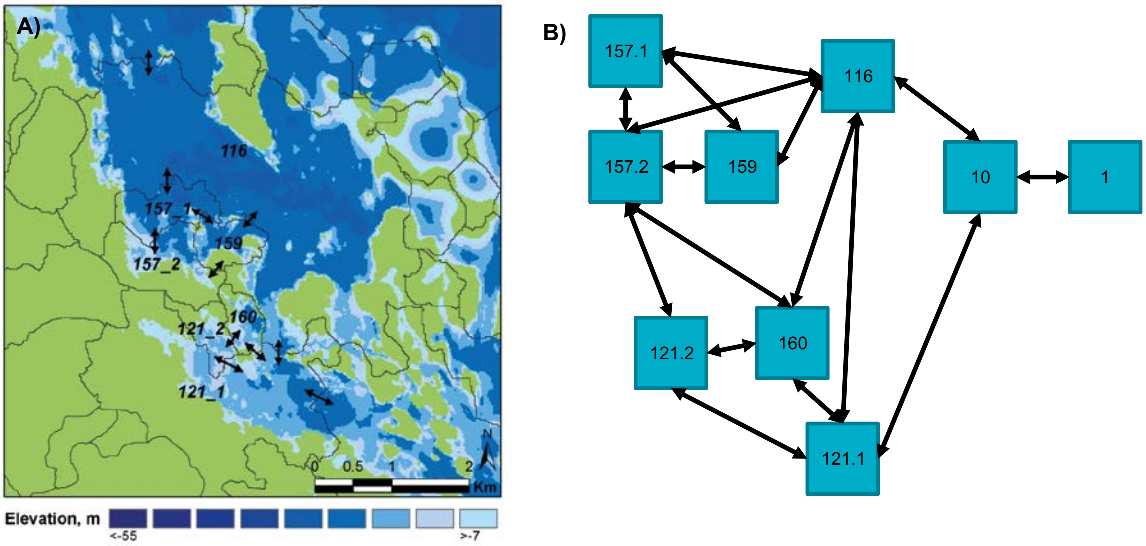

Figure 4-3. *Flow pathways for exchange of surface water between sea basins. A) Flow-paths (black arrows) superimposed on bathymetric colour map from 3000 AD (green represents land) B) Schematic diagram of flow-paths during present conditions at 2000 AD. The basins closest to the repository were identified as biosphere objects (indicated by numbers). For simplicity all other basins in Öregrundsgrepen have been combined in one object (object 10), which exchanges water with the rest of the Baltic Sea (object 1).*

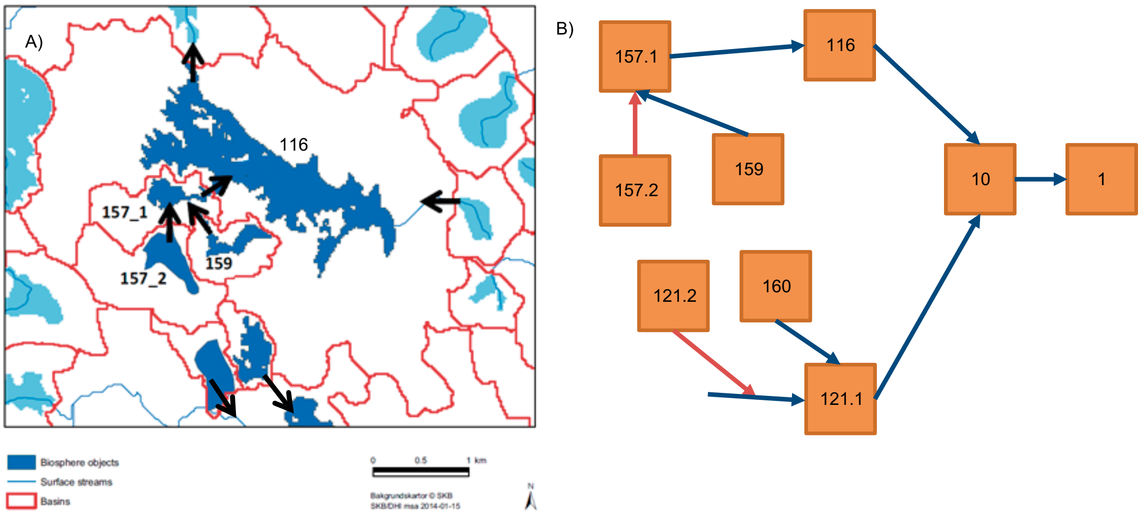

Figure 4-4. *Surface water flow-paths between biosphere objects in the land period. A) Flow-paths (black arrows) superimposed on a landscape map showing the extent of the biosphere objects in the land period (dark blue) and of the original sea basin (red). Figure modified from Werner et al. (2014). B) Schematic diagram with surface water connections. Boxes represents biosphere objects, and arrows represents fluxes of stream water (blue arrows) and diffuse fluxes of overland water (brown arrows).*

The advective transport from the water body of an upstream biosphere object (a lake or a stream) to the water body of downstream biosphere object, $Adv_{downstream}$ [Bq year^{-1}], is expressed as the product of the yearly total flow of water downstream and the radionuclide concentration in water:

$$Adv_{downstream} = WF_{downstream}\, AC_{Water} \tag{4-5}$$

$$Adv_{downstream,org} = WF_{downstream}\, AC_{Water,org}$$

Where

AC_{Water}	is the radionuclide concentration in water, including dissolved and adsorbed radionuclides [Bq m^{-3}],
$AC_{Water,org}$	is the concentration of radionuclides in organic particulate matter [Bq m^{-3}].
$Adv_{downstream}$	is the advective flow of water from the current to the downstream object [m^3 y^{-1}] defined as:

$$Adv_{downstream} = q_{downstream,k}\, area_{obj} \tag{4-6}$$

Where

$q_{downstream,k}$	is the area specific water flux from an upstream to a downstream object [m^3 m^{-2} y^{-1}],
$area_{obj}$	is the surface area of the biosphere object (lake and mire) in the land period [m^2], and
k	is the point in time (at isolation or at mire completion, see text).

Similarly, for the objects that developed into a mire without a lake, the transport of radionuclides from shallow pools and surface peat to the downstream biosphere object, $Adv_{downstream}$ [Bq year^{-1}], is expressed as the product of the yearly diffusive flow of overland water out of the object and the activity concentration in peat pore water.

$$Adv_{downstream} = WF_{downstream}\, AC^{D}_{Up,ter} \tag{4-7a}$$

Where

$AC^{D}_{Up,ter}$	is the pore water concentration of radionuclides in surface peat [Bq m^{-3}].

Estimates for area-specific downstream water fluxes (q_i) were available for two conditions, namely for the early land period shortly after lake isolation/land emergence ($q_{downstream,iso}$) and for the late land period when the mire expansion is complete ($q_{downstream,end}$). For the transport calculations, it is assumed that the downstream flux of water is stable in the late land stage, whereas the fluxes in the period after isolation/emergence are assumed to change approximately linearly until stable conditions are reached (see Appendix G for details).

The analysis of surface water flow paths during the land period suggests that radionuclides from biosphere objects will reach downstream objects primarily through stream water (Werner et al. 2014). Moreover for transport calculations it is assumed that most stream water will reach the water body (lake or stream) of the downstream object (Q_{in} in Figure 4-1), before dispersing to the mire areas by flooding (see below). The only exception in the landscape above the repository is object 157_2, where radionuclides are expected to reach object 157_1 in a diffuse flow of overland water (i.e. a flow of surface runoff and near-surface groundwater) (Werner et al. 2014, Figure 10-3).

Given the uncertainties of future diffuse flow paths from the upstream mire in 157_2 into the lake-mire complex of object 157_1, it is assumed that both parts of the downstream object will receive radionuclides in the period when mire vegetation is expanding into the lake basin. It is further assumed that radionuclides will reach the aquatic and the terrestrial parts of the object in proportional to the surface area of each ecosystem in the downstream object[4]. However, in the late

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[4] As the wetland vegetation expands from the shores of the lake, the water body is presumably surrounded by a zone of wetland vegetation, and considerable amounts of C-14 would potentially be lost by degassing before reaching the water column of the down-stream lake. Thus, allowing a fraction of the radionuclides to reach the water body directly (by diffuse fluxes of overland water) is a cautious assumption with respect to the C-14 activity concentration of the water column (and in aquatic organisms). Due to horizontal mixing of water (Equations 4-8 and 4-9), the original partitioning of the influx of non-volatile radionuclides between the water body and the mire is presumably of limited importance.

land period when the mire is fully developed (Time > threshold_end), all radionuclides from the upstream object 157_2 are expected to reach the mire (before reaching the central stream).

The total flux of radionuclides from 157_2 to 157_1 can be divided into two parts, one directed to the mire and one to the lake of the downstream object:

$$Adv_{downstream,ter} = WF_{downstream}\, AC^{D}_{regoUp,ter}\, f_{ter} \tag{4-7b}$$

$$Adv_{downstream,aqu} = WF_{downstream}\, AC^{D}_{regoUp,ter}\, (1 - f\_ter)$$

Where f_{ter} is the fraction of the object that is mire in the downstream object (157_1).

4.3.3 Transport between lake and mire ecosystems within biosphere objects

In the land period there is horizontal advective transport of radionuclides between the mire and the lake/stream ecosystems within a biosphere object due to the exchange of water. The exchange of water primarily occurs through a diffuse flux of overland water from surface peat to the lake/stream, and in the opposite direction by flooding of the lake/stream. The advective transport of radionuclides from surface peat to the lake/stream, $Adv_{Up,ter,Water}$ [Bq y^{-1}], is expressed as the product of the advective flow of water and the pore water concentration of radionuclides in the upper regolith:

$$Adv_{Up,ter,Water} = WF_{Up,ter,Water}\, AC^{D}_{Up,ter} \tag{4-8}$$

where

$WF_{Up,ter,Water}$ is the advective flow of water between surface peat and open water [m^3 y^{-1}], and

$AC^{D}_{Up,ter}$ is the pore water activity concentration in surface peat [Bq m^{-3}].

Similarly, the advective transport of radionuclides from the water body in to the mire due to flooding, $Adv_{Water,Up,ter}$ [Bq y^{-1}], is expressed as the product of the advective flow of water and the total activity concentration in the water (includes dissolved radionuclides and radionuclides in or on particulate matter):

$$Adv_{Water,Up,ter} = WF_{Water,Up,ter}\, AC_{Water} \tag{4-9}$$

$$Adv_{Water,Up,ter,org} = WF_{Water,Up,ter}\, AC_{Water,org}$$

where

AC_{Water} is the concentration of radionuclides in water [Bq m^{-3}],

$AC_{Water,org}$ is the concentration of radionuclides in water (organic form) [Bq m^{-3}], and

$WF_{Water,Up,ter}$ is the advective flow of water between the water body and surface peat [m^3 y^{-1}].

The horizontal flux of water between the mire and the stream, $WF_{i,k}$ [m^3 y^{-1}], is the product of the area-specific water flux and the total area of the object:

$$WF_{i,k} = q_{i,k}\, area_{obj} \tag{4-10}$$

where

$q_{i,k}$ is the area-specific water flux between the mire and the aquatic parts of the object [m^3 m^{-2} y^{-1}],

$area_{obj}$ is the total surface area of the biosphere object (mire and lake) in the land period [m^2],

i is the direction of the flux (from $RegoUp_{ter}$ to *Water*, or the opposite direction due to flooding), and

k is the point in time (at isolation or at mire completion).

Estimates for area-specific water fluxes (q_i) were available for two different stages, namely for the early land period shortly after lake isolation/land emergence ($q_{i,iso}$) and for the late land period when the mire expansion was complete ($q_{i,end}$). For the transport calculations it is assumed that the downstream flux of water is stable in the late land stage, whereas the flux in the period after isolation/emergence is assumed to change approximately linearly until stable conditions are reached (see Appendix G for details).

4.4 Transport of radionuclides by diffusion

There will be an exchange of radionuclides between sediment layers next to each other due to diffusion in the water-saturated pore space. Diffusion is the net flux of radionuclides down a concentration gradient due to random movement. Under steady-state conditions, i.e. when the concentration within the diffusion volume does not change over time; the diffusive transport in one dimension in free water can be described by Fick's first law. According to Fick's first law: the diffusion flux density, J [Bq m^{-2} y^{-1}], is directly proportional to the steepness of the concentration gradient:

$$J = -D_w \frac{\partial C_w}{\partial z} \tag{4-11}$$

where

J is the diffusion flux of radionuclides at a given point [Bq m^{-2} y^{-1}],

D_w is the diffusion coefficient for water-based radionuclide concentration [m^2 y^{-1}],

C_w is the radionuclide concentration in water [Bq m^{-3}], and

z is the position in space [m].

Fick's first law applies to diffusion in bulk water. In a porous medium the diffusion process is inhibited and the diffusion coefficient must be reduced. This is because the available cross-section for transport decreases in the porous medium. Moreover, the transport paths get longer (accounted for by the tortuosity) and there may be significant hindrance in narrow passages (accounted for by the constrictivity). Therefore, the diffusive flux in porous media can be described by using an effective diffusivity, or as a function of the diffusivity in water and properties of the porous medium (Ohlsson and Neretnieks 1997):

$$J = -D_e \frac{\partial C_w}{\partial z} = -D_p \theta \frac{\partial C_w}{\partial z} = -D_w \frac{\partial}{\tau} \theta \frac{\partial C_w}{\partial z} \tag{4-12}$$

where

D_e is the effective diffusivity [m^2 y^{-1}],

D_p is the pore diffusivity [m^2 y^{-1}],

θ is the porosity (the volume fraction of the medium that allows transport) [m^3 m^{-3}],

δ is the constrictivity [unitless], and

τ is the tortuosity [unitless].

The δ/τ ratio will always be less than one (by definition). For a generic assessment, it is difficult to choose values for this ratio, and, in this assessment, the value of 1 has been used to give an upper bound for the diffusion rate.

Equation 4-12 is continuous in space. In a compartment model, a continuous direction in space is represented by stacking homogenous compartments next to each other. Therefore diffusion between two adjacent compartment layers can be approximated as:

$$J = -D_e \frac{C_{w,z1} - C_{w,z2}}{\Delta z} \tag{4-13}$$

where

$C_{w,i}$ is the pore water concentration in compartment i, and

Δz is the distance between the mid points of the two compartments.

In the radionuclide model, sediment layers are treated as homogenously mixed compartments. Moreover to describe diffusion as a first-order process with respect to one compartment only, the net diffusion between sediment layers is described by two flows (back and forth) against zero concentration. Thus the net flow of radionuclides is described by two fluxes, *Diff.→*. [Bq $year^{-1}$], where each flux is proportional to the activity concentration in the source compartment, the contact area and the distance between the compartments, and the diffusivity of the compartments:

$$Diff_{i\rightarrow j}=\frac{AC_i^D area_{cross,i,j}}{\frac{z_i/2}{D_{e,i}}+\frac{z_j/2}{D_{e,j}}},\qquad Diff_{j\rightarrow i}=\frac{AC_j^D area_{cross,i,j}}{\frac{z_i/2}{D_{e,i}}+\frac{z_j/2}{D_{e,j}}},\tag{4-14}$$

where

AC_i^D	is the activity concentration in pore water of sediment layer i [Bq m^{-3}],
$area_{cross,i,j}$	is the cross-sectional area of sediment layers perpendicular to direction of diffusion [m^2], corresponding to the area of the aquatic sediments or the terrestrial part of the object.
$D_{e,i}$	is the effective diffusion coefficient in sediment layer i [m^2 y^{-1}], and
Z_i	is the depth of the sediment layers i [m].

Diffusion between the upper aquatic regolith layer and surface water is also modelled in a similar way. However, here we only consider the diffusion to a thin boundary layer of water which is in direct contact with the sediment surface (and radionuclides are assumed to be fully mixed by advection once they reach the water column):

$$Diff_{Water,Up}=\frac{AC_{water}^D area_{obj,aqu}}{\frac{z_{regoUp,aqu}/2}{D_{e,regoUp,aqu}}},\quad Diff_{Up,Water}=\frac{AC_{regoUp,\,aqu}^D area_{obj,aqu}}{\frac{z_{regoUp,aqu}/2}{D_{eff,regoUp,aqu}}},\tag{4-15}$$

where

AC_{water}^D	is the activity concentration of radionuclides dissolved in the water column (see Equation 5-4 in next chapter), and
$area_{obj,aqu}$	is the area of the aquatic sediment in contact with the water column [m^2].

5 Aquatic ecosystem

Today the marine basins in Forsmark are characterised by brackish water, i.e. the salt concentration is between true saltwater and limnic. The brackish water supports an impoverished flora and fauna compared with a fully marine environment and the brackish marine environment is inhabited by a mix of marine and freshwater species adapted to the conditions. The archipelago of Forsmark has an open exposed character, and macrophyte vegetation is dominated by red algae, brown filamentous algae and bladder rack (*Fucus vesiculosus*). The highest biomass of benthic fauna is found in vegetation associated with soft bottoms, were the Baltic mussel (*Macoma baltica*) dominates the biomass. In the inner areas, pike and perch are the most abundant fish species, whereas sprat and herring dominates the more off-shore areas. The near-shore areas and the secluded bays exhibit dense benthic vegetation and very high primary production. In deeper offshore areas, primary production is lower and is dominated by pelagic species.

Lakes in the Forsmark area are small (<0.6 km^2) and shallow (0.6–2 m deep) oligotrophic hardwater lakes, with a high pH and high concentrations of calcium and bicarbonate. Relatively slight water colour and shallow depths of the lakes permits photosynthesis in the entire benthic zones and the bottoms are covered by dense stands of the macroalgae *Chara sp.* Many lakes also have a thick microbial mat consisting of cyanobacteria, diatoms and heterotrophic bacteria. The high biomass of benthic macro and microalgae gives rise to high primary production, whereas the pelagic primary production is relatively low. The fish community is dominated by species resistant to low oxygen conditions. Crayfish have not been found in the Forsmark lakes, presumably due to soft bottoms with low oxygen concentrations.

Several small natural ponds are present in the area. These are shallow (<0.5 m), generally lack fish and house several amphibians protected by Swedish law (e.g. the pool frog, *Rana lessonae*). Streams in the area are small and many dry out during summer. The streams are used by migratory fish, but there are no permanent stream populations of fish. The characteristics of the present and future aquatic ecosystems in Forsmark are briefly described in Section 2.2.1, and detailed descriptions are given in the limnic and marine ecosystem reports (Andersson 2010, Aquilonius 2010).

Groundwater from the SFR repository can reach aquatic ecosystems through direct discharge from the bedrock. Radionuclides can then be transported by water exchange with neighbouring sea basins or with surface water to downstream lakes and wetlands. In Chapter 3 (model structure) compartments and transport mechanisms that are important for the distribution of radionuclides in surface ecosystems are identified. In this chapter, the mathematical description of these compartments (Section 5.1) and fluxes (Section 5.2) for the aquatic ecosystems is presented. The biogeochemical cycle of carbon differs significantly from that of other elements (e.g. carbon is the principal structural component in biomass, it is the main carrier of energy in the ecosystem and it is present in gaseous form at ambient temperature). Consequently, the mathematical description relating to radiocarbon (C-14) frequently differs from that of other radionuclides. Water-bound transport of radionuclides by advection and diffusion is described in Section 4.2. That section also includes a description of radionuclide fluxes between biosphere objects, and between the lake and mire ecosystems within a biosphere objects. A schematic representation of the sub-model of transport in aquatic ecosystems is presented at the end of this chapter (Figure 5-1).

5.1 Mathematical representation of radionuclide inventories

In this section, inventories of radionuclides in the aquatic ecosystem are described in mathematical terms. These include those present in water, suspended particulate matter, primary producer biomass, and regolith layers (as previously identified in Section 3.1). In the biosphere model, radionuclides in the atmosphere are not modelled as a separate dynamically changing inventory (see Chapter 8), instead the surface atmosphere is regarded as a source/sink of radionuclides.

5.1.1 Surface water [Water and PM_{org}]

The radionuclide inventory in surface water of sea basins, lakes and streams is represented by two compartments *Water* and $PM_{org.}$ The inventory *Water* includes species in solution and adsorbed on suspended particulate matter. The time to reach equilibrium between solid and aqueous phases is assumed to be short (time scale in the order of days/months, Periáñez 2005) compared with the model time resolution (year). Radionuclides incorporated into organic particulate carbon are represented by PM_{org}. Note that all radionuclides that are incorporated into biomass by plant uptake (Section 5.2.1) are in organic form until released by litter respiration or organic matter mineralisation (Sections 5.2.2 or 5.2.8). Dissolved low molecular weight organic carbon is typically mineralised within days or weeks (Howard 1991), and all C-14 released to the aqueous phase is therefore assumed to be transformed to inorganic carbon within the yearly time resolution of the model.

The activity concentration AC [Bq m^{-3}] of the inorganic and the organic compartments in the water column are by definition the inventory divided by the water volume:

$$AC_{water} = \frac{Water}{vol_{water}},\quad AC^{PM}_{water,org} = \frac{PM_{org}}{vol_{water}} \tag{5-1}$$

where

Water is the activity of a radionuclide in surface water (dissolved or adsorbed) [Bq],

PM_{org} is the activity of a radionuclide in suspended particulate organic matter [Bq], and

vol_{water} is the water volume in the sea/lake basin or stream of the object [m^3] defined as:

$$vol_{water} = area_{obj,aqu} z_{water}, \tag{5-2}$$

Where

$area_{obj,aqu}$ is the surface area of open water (sea basin/lake/stream) in the biosphere object [m^2],

z_{water} is the average depth of the water body [m].

Radionuclides from deep groundwater are assumed to enter the biosphere in inorganic form, with the exception of radioactive carbon (C-14), which may be in the form of low molecular weight organic compounds (e.g. dissolved methane, acetate and formate). However, as the mineralisation rate of low molecular weight organic compounds (and organic exudates from aquatic primary producers) is fast, the pool of particulate refractory carbon is considered sufficient for modelling the fate of radionuclides in organic form in the water column. The activity concentration of radionuclides in organic form, $AC_{PM,org}$ [Bq kg$_{DW}^{-1}$], can thus be expressed relative to the dry weight (DW) of suspended particulate matter in the water column:

$$AC_{PM,org} = \frac{PM_{org}}{M^{PM}_{water}}, \tag{5-3}$$

where the total mass of suspended particulate matter in the water column M^{PM}_{water} [kg$_{DW}$] is defined as:

$M^{PM}_{water} = vol_{water} conc_{PM}$, and

$conc_{PM}$ is the concentration of suspended matter [kg$_{DW}$ m^{-3}] in water.

Radionuclides in the water column can be partitioned into an aqueous (dissolved) and a solid (adsorbed) phase, where the equilibrium between the two phases is described by a constant, element specific, solid/liquid distribution coefficient (Kd). The activity concentration in the solute phase [Bq m^{-3}] is therefore defined as:

$$AC^{D}_{water} = \frac{Water\, f^{D}_{water}}{vol_{water}}, \tag{5-4}$$

where

f^{D}_{water} is the fraction of the radionuclide partitioned to the solute phase [Bq Bq^{-1}] given by:

$$f^{D}_{water} = \frac{1}{1+K_{d,PM}\, conc_{PM}}, \tag{5-5}$$

where

$K_{d,PM}$ is the distribution coefficient for particulate matter in water [m^3 kg_{DW}^{-1}].

The activity concentration adsorbed on particulate matter in the water column [Bq Kg_{DW}^{-1}] is:

$$AC^{PM}_{water} = \frac{Water^{PM}}{M^{PM}_{water}}, \tag{5-6}$$

where

$Water^{PM}$ is the activity adsorbed on particulate matter in the water column [Bq].

The inventory adsorbed on particulate matter $Water^{PM}$ relates to the total inventory of radionuclide in inorganic form as:

$$Water^{PM} = Water\,(1 - f^{D}_{water}) \tag{5-7}$$

5.1.2 Aquatic primary producers [$PP_{plank,micro,macro}$]

There are three different communities of aquatic primary producers, namely benthic macroalgae (macro), benthic microalgae (micro) which are primarily epiphytic, and phytoplankton (plank) which lives pelagically in the water column. These communities differ vastly in their production properties (biomass, carbon fixation rate, and production of refractory organic matter). Moreover, the composition of primary producers (with respect to these communities) changes when the aquatic system becomes shallower. Therefore separate records of the radionuclide inventories in planktonic, microbenthic and macrobenthic primary producers are kept by introducing separate compartments.

The activity concentration in the communities of primary producers is expressed per unit biomass carbon [Bq kgC^{-1}]. As the primary producer biomass is expressed per unit area and the activity concentration in each community is:

$$AC_{biom,i} = \frac{PP_i}{biom_{pp,i}\, area_{obj,aqu}}, \quad i = \{plank, micro, macro\}, \tag{5-8}$$

where

PP_i is the total activity in community i [Bq],

$biom_{pp,i}$ is the area-specific biomass of community i [kgC m^{-2}],

$area_{obj,aqu}$ is the surface area of open water (sea basin, lake or stream) in the object[m^2], and

i is the index for the plankton, micro- and macro-benthic communities.

5.1.3 Upper sediments [RegoUp, $RegoUp_{org}$]

The radionuclide inventory in the upper sediment in aquatic ecosystems is represented by two compartments, namely $RegoUp_{aqu}$ and $RegoUp_{org,aqu}$. In the Baltic Sea, the aerobic, biologically active, zone of aquatic sediments is approximately 10 cm deep, whereas the corresponding zone in lakes and streams is only 5 cm deep (Håkanson et al. 2004, Andersson 2010). The inventory consists of radionuclides in solution, radionuclides adsorbed on sediment particles and radionuclides incorporated into organic particulate matter.

For *all* regolith layers, the time to reach equilibrium between solid and aqueous phases is assumed to be short (time scale of days to months, Periáñez 2005) compared with the model time resolution (year). Moreover, dissolved low molecular weight organic carbon is typically mineralised within days or weeks (Howard 1991), and organic C-14 in the aqueous phase is therefore assumed to be transformed to inorganic carbon within the time resolution of the model.

The activity concentration in the inorganic part of the upper aquatic sediments [Bq kg_{DW}^{-1}] is a function of the radionuclide inventory, the volume of sediments and the sediment density. In general form, this can be expressed as:

$$AC_{rego\_i,aqu} = \frac{Rego\_i,\ aqu}{vol_{rego\_i,aqu} dens_{rego\_i,aqu}}, \tag{5-9}$$

where

$Rego\_i$	is the activity of a radionuclide in inorganic form in aquatic regolith layer i [Bq],
$dens_{rego\_i,aqu}$	is the bulk density of the aquatic regolith layer i [kg_{DW} m^{-3}],
$vol_{rego\_i,aqu}$	is the volume of the aquatic regolith layer i [m^3] defined as:

$$vol_{rego\_i,aqu} = area_{obj,sed,aqu} z_{rego\_i,aqu} \tag{5-10}$$

where

$area_{obj,sed,aqu}$	is the surface area of the aquatic sediments in the marine basin or lake [m^2] (see below),
$z_{rego\_i,aqu}$	is the depth of the aquatic regolith layer i [m], and
i	indicate the regolith layer of interest. That is, $i{=}Up$ representing the upper biological active part of aquatic sediment (*RegoUp*).

During submerged conditions, groundwater from the repository is expected to be discharged to central parts of the sea basins only (Odén et al. 2014). Moreover accumulation of aquatic sediments is largest in the deepest parts of the sea basins (Brydsten and Strömgren 2013). As one of the objectives of the assessment is to capture long-term accumulation across ecosystem succession (Chapter 1), uncertainties with respect to the areas of groundwater discharge and sediment accumulation were cautiously handled by modelling radionuclide accumulation in aquatic sediments in the central part of the basin only. For simplicity, the size of this area was approximated as the extent of the future lake basin or the final mire area (for objects that did not form a lake) (Figure 2-2). This simplistic approach ensures that all radionuclides accumulating during the submerged period are inherited by the lake-mire ecosystems emerging above the sea level.

During the submerged period the surface area of aquatic sediments is constant. However, after isolation, the area of the aquatic sediments decreases as mire vegetation expands into the lake. In the model this is described as:

$$area_{obj,sed,aqu} = \begin{cases} area_{obj}, & in\ submerged\ period \\ area_{obj} - area_{obj,ter}, & otherwise \end{cases} \tag{5-11}$$

where

$area_{obj}$	is the surface area of the biosphere object immediately after emergence [m^2], and
$area_{obj,ter}$	is the area of the biosphere object that is covered by mire vegetation [m^2].

Dissolved organic carbon is assumed to be transformed into inorganic carbon within the time resolution of the model (see above). Consequently, radionuclides in organic matter are taken to be stored in refractory solid organic matter. The activity concentration in solid organic matter [Bq kg_{DW}^{-1}] is defined as:

$$AC_{rego\_i,org,aqu} = \frac{Rego\_i_{org,aqu}}{vol_{rego\_i,aqu} dens_{rego\_i,aqu}}, \tag{5-12}$$

where

$Rego\_i_{org,aqu}$	is the activity in solid organic matter in aquatic regolith layer i [Bq].

The inventory of a radionuclide in inorganic matter in the pore space of a sediment is partitioned into an aqueous (dissolved) and a solid (adsorbed) phase, where the equilibrium between the two phases is described by a constant element specific solid/liquid distribution coefficient (K_d, Equation 3-1).

As all aquatic sediments are water saturated, the activity concentration in the pore water of a regolith compartment is the inventory in the solute phase [Bq] divided by the pore volume [m^3] of the layer:

$$AC^D_{rego\_i,aqu} = \frac{Rego\_i_{aqu} f^D_{rego\_i,aqu}}{vol_{rego\_i,aqu}\, poro_{rego\_i,aqu}}, \tag{5-13}$$

where

$poro_{rego\_iaqu}$ is the porosity of the aquatic regolith layer i [m^3 m^{-3}], and

$f^D_{rego\_i,aqu}$ is the fraction of the radionuclide in the solute phase for the aquatic regolith layer i [Bq Bq^{-1}].

The fraction of a radionuclide in inorganic form in the solute phase in the aquatic regolith layer i [Bq Bq^{-1}] is a function of the distribution coefficient and the physical properties of the sediment layer (Appendix A):

$$f^D_{rego\_i,aqu} = \frac{1}{1 + \frac{K_{d,rego\_i,aqu}\, dens_{rego\_i,aqu}}{poro_{rego\_i,aqu}}} \tag{5-14}$$

where

$K_{d,rego\_i,aqu}$ is the distribution coefficient in the aquatic regolith layer i [m^3 kg$_{DW}^{-1}$].

5.1.4 Post-glacial clay gyttja [RegoPG, RegoPG$_{org}$]

The radionuclide inventory in post glacial clay-gyttja in aquatic ecosystems is represented by two compartments, namely $RegoPG_{aqu}$ and $RegoPG_{org,aqu}$. These regolith layers represents water saturated, anaerobic deposits located directly below the upper sediments. As with the upper aquatic sediments, the inventory in this regolith layer consist of radionuclides in solution, radionuclides adsorbed on sediment particles and radionuclides incorporated into organic matter.

The activity concentration of a radionuclide in inorganic form, $AC_{regoPG,aqu}$ [Bq kg$_{DW}^{-1}$], the activity concentration of the radionuclide in solid organic matter, $AC_{regoPG,aqu,org}$ [Bq kg$_{DW}^{-1}$], and the activity concentration of dissolved radionuclide in the pore water $AC^D_{regoPG,aqu}$ [Bq m^{-3}], of aquatic clay-gyttja are described by Equations 5-9, 5-12 and 5-13, respectively (*i=regoPG).* Similarly, the fraction of inorganic radionuclides in solute phase, $f^D_{regoPG,aqu}$ [Bq Bq^{-1}], is expressed with Equation 5-14.

5.1.5 Glacial clay [RegoGL]

The radionuclide inventory in the glacial clay in aquatic ecosystems is represented by the compartment *RegoGL*. This regolith layer represents water saturated, anaerobic, minerogenic glacial deposits (clay) and it is typically located between post-glacial clay-gyttja and the till layer (see below). The inventory consists of radionuclides in solution and radionuclides adsorbed on sediment particles. As dissolved organic matter is assumed to be mineralised within the time resolution of the model, no radionuclides in organic matter are assumed to reach this regolith layer.

The activity concentration of a radionuclide in inorganic form, $AC_{regoGL,aqu}$ [Bq kg$_{DW}^{-1}$], and the activity concentration of a dissolved radionuclide in the pore water, $AC^D_{regoGL,aqu}$ [Bq m^{-3}], of aquatic glacial clay are described by Equations 5-9 and 5-13, respectively (*i=regoGL).* Similarly, the fraction of a radionuclide in inorganic form in the solute phase, $f^D_{regoGL,aqu}$ [Bq Bq^{-1}], is given by Equation 5-14.

5.1.6 Till [RegoLow]

The radionuclide inventory in the deepest regolith layer in aquatic ecosystems is represented by the compartment *RegoLow*. This regolith layer represents loose, unsorted minerogenic, water-saturated (anoxic) sediments overlying the bedrock. The inventory consists of radionuclides in solution and radionuclides adsorbed on sediment particles. As dissolved organic matter is assumed to be mineralised within the time resolution of the model, no radionuclides in organic matter are assumed to reach this regolith layer.

The activity concentration of a radionuclide in inorganic form, $AC_{regoLow,aqu}$ [Bq kg_{DW}^{-1}], and the activity concentration of a radionuclide dissolved in the pore water $AC^{D}_{regoLow,aqu}$ [Bq m^{-3}], of aquatic glacial clay are described by Equations 5-9 and 5-13, respectively (*i=regoLow)*. Similarly, the fraction of a radionuclide in inorganic form in the solute phase, $f^{D}_{regoLow,aqu}$ [Bq Bq^{-1}], is given by Equation 5-14.

5.2 Mathematical representation of radionuclide fluxes

In this section, radionuclide fluxes associated with plant uptake, litter production, litter decomposition, sedimentation, resuspension, burial, bioturbation, mineralisation of refractory organic matter and gas exchange with the atmosphere are described in mathematical terms. Radionuclide fluxes associated with advective and diffusive transport are described in Chapter 4. Radionuclide fluxes within the atmosphere above the surface water are described in Chapter 8. Radionuclide fluxes associated with the transition from lake to mire are described in Section 6.2.

5.2.1 Plant Uptake

Plant uptake represents fixation of inorganic radiocarbon and other radionuclides in plant biomass as a result of net primary production. It includes active and passive uptake, as well as external adsorption onto plant tissue (see Section 3.2).

Plant uptake [Bq y^{-1}] is the product of net primary production [kgC y^{-1}] and the activity concentration in synthesised biomass [Bq kgC^{-1}]:

$$Plant_{uptake,i} = AC_{NPP,i} NPP_i area_{obj,aqu}, \qquad i = \{plank, micro, macro\} \tag{5-15}$$

where

$AC_{NPP,i}$ is the activity concentration in newly synthesised biomass [Bq kgC^{-1}],

NPP_i is the area specific net primary production for community *i* [kgC m^{-2} y^{-1}], and

$area_{obj,aqu}$ is the surface area of open water (sea basin, lake or stream) in the object [m^2].

Since isotopic discrimination of C-14 (as compared with stable C-12) is small during photosynthesis, plant allocation and biomass production (Peterson 1980), the C-14 concentration in synthesised biomass can be approximated as the specific activity concentration of C-14 in water [Bq kgC^{-1}] for all aquatic communities:

$$AC^{14C}_{NPP,i} = SA^{14C}_{water}, \qquad i = \{plank, micro, macro\} \tag{5-16}$$

where

SA^{14C}_{water} is the C-14 specific activity in water [Bq kgC^{-1}].

The specific activity concentration of C-14 in water [Bq kgC^{-1}] is expressed as the quotient of the C-14 activity concentration in water over the concentration of dissolved inorganic stable carbon:

$$SA^{14C}_{water} = \frac{AC^{D14C}_{water}}{conc_{DIC,aqu}} \tag{5-17}$$

where

AC^{D14C}_{water} is the activity concentration of dissolved inorganic C-14 in water [Bq m^{-3}], and

$conc_{DIC,aqu}$ is the concentration of dissolved inorganic carbon in water [kgC m^{-3}].

The activity concentration in newly synthesised biomass of other radionuclides is derived from the activity concentration in surface water [Bq m^{-3}], assuming that newly synthesised biomass is in equilibrium with the surrounding water:

$$AC^{RN}_{NPP,i} = AC^{D,RN}_{water} CR_{aqu,pp,i}, \qquad i = \{plank, micro, macro\} \tag{5-18}$$

where

$AC^{D,RN}_{water}$ is the concentration of the dissolved radionuclide (i.e. of the solute phase) [Bq m^{-3}], see Equation 5-4, and

$CR_{aqu,pp,i}$, is the concentration ratio for aquatic primary producer community *i* [m^3 kgC^{-1}].

5.2.2 Litter respiration [C-14] / release [other RN]

Fast (within a year) decomposition of plant litter leads to mineralisation and releases of radionuclides to the water. On the time scale of a year, the primary producer biomass is assumed to be in equilibrium, which means that the net primary production (NPP, kgC y^{-1}) is balanced by an equal loss of biomass through consumption and litter production.

The flux of C-14 associated with the fast decomposition of dead organic matter, $Litter_{resp}^{14C}$ [Bq y^{-1}], can be expressed as the product of the activity concentration in plant biomass, and the net production of labile (non-refractory) carbon. Thus, the total flux of C-14 from litter mineralisation associated with primary producer community *i* is:

$$Litter_{resp,i}^{14C} = AC_{biom,i}^{14C} NPP_i area_{obj,aqu} (1 - f_{refrac,i}) \quad (5\text{-}19)$$

where

$AC_{biom,i}^{14C}$ is the activity concentration of C-14 for community *i* [Bq kgC^{-1}],

NPP_i is the area specific net primary production for community *i* [kgC m^{-2} y^{-1}],

$area_{obj,aqu}$ is the surface area of open water (sea basin, lake or stream) in the object [m^2],

$f_{refrac,i}$ is the fraction of refractory organic matter in litter from community *i* [kgC kgC^{-1}], and

i is the index for the communities plankton (plank), microbenthos (micro) and macrobenthos (macro).

When easily metabolised carbon sources are respired, radionuclides other than C-14 stored in the organic matter are also released to the water column. The release of radionuclides in organic matter will be roughly proportional to the respired carbon. However, radionuclides that are stored in inorganic form in plant tissue will be released at a greater rate than carbon is mineralised, whereas other fractions of radionuclides may be subject to a much slower release.

The release of radionuclides from dead organic matter to the water column $Litter_{release}^{RN}$ [Bq y^{-1}] is a function of the activity concentration in plant biomass, the net production of labile (non-refractory) carbon in biomass and a correction factor for diminishment/enrichment of the radionuclide in the refractory organic matter:

$$Litter_{release,i}^{RN} = AC_{biom,i}^{RN} NPP_i area_{obj,aqu} (1 - f_{refrac,i})\ df_{decomp,aqu} \quad (5\text{-}20)$$

where

$AC_{biom,i}^{RN}$ is the activity concentration in community *i* [Bq kgC^{-1}],

$df_{decomp,aqu}$ is the fraction of the radionuclide that is available for release through mineralisation of labile organic matter [Bq Bq^{-1}], and

i is the index for the communities plankton (plank), microbenthos (micro) and macrobenthos (macro).

5.2.3 Litter production

Litter production $Litter_{prod}$ [Bq y^{-1}] represents the transfer of radionuclides from dead organic matter (from primary producers) to the upper sediments. It can be expressed as the product of the activity concentration in biomass, the net primary production and the fraction of biomass that is not easily decomposed:

$$Litter_{prod,i}^{14C} = AC_{biom,i}^{14C} NPP_i area_{obj,aqu} f_{refrac,i}, \qquad i = \{plank, micro, macro\}, \quad (5\text{-}21)$$

where

$AC_{biom,i}^{14C}$ is the activity concentration of C-14 in the biomass of community *i* [Bq kgC^{-1}],

NPP_i is the area specific net primary production for community *i* [kgC m^{-2} y^{-1}],

$area_{obj,aqu}$ is the surface area of open water (sea basin, lake or stream) in the object [m^2],

$f_{refrac,i}$ is the fraction of refractory organic matter for community *i* [kgC kgC^{-1}].

From mass balance it follows that the flux of radionuclides (other than C-14) in refractory litter, $Litter_{prod,i}^{RN}$ [Bq year^{-1}] equals the total flux in dead organic matter less radionuclides released through decomposition of the labile pool (within a year):

$$\begin{aligned} Litter_{prod,i}^{RN} &= AC_{biom,i}^{RN} NPP_i area_{obj,aqu} - Litter_{release,i}^{RN} \\ &= AC_{biom,i}^{RN} NPP_i area_{obj,aqu} \left(1 - \left(1 - f_{refrac,i}\right) df_{decomp,aqu}\right) \end{aligned} \tag{5-22}$$

where

$AC_{biom,i}^{RN}$	is the activity concentration for community i [Bq kgC^{-1}],
$df_{decomp,aqu}$	is discrimination factor during decomposition [Bq Bq^{-1}],
i	is the index for the communities plankton (plank), microbenthos (micro) and macrobenthos (macro)

5.2.4 Sedimentation

Sedimentation refers to the transport of a radionuclide on the surfaces of suspended particulate matter or stored in suspended particulate organic matter to the upper aquatic sediments [Bq y^{-1}]. The flux of a radionuclide from the water column is a product of the total sedimentation rate [kg$_{DW}$ y^{-1}] and the activity concentration of sedimenting particulate matter.

$$\begin{aligned} Sedimentation &= sed_{rate} area_{obj,aqu} dens_{regoUp,aqu} AC_{water}^{PM} \\ Sedimentation_{org} &= sed_{rate} area_{obj,aqu} dens_{regoUp,aqu} AC_{PM,org} \end{aligned} \tag{5-23}$$

where

sed_{rate}	is the area specific volumetric gross sedimentation rate [m^3 m^{-2} y^{-1}],
$area_{obj,aqu}$	is the surface area of open water (sea basin, lake or stream) in the object [m^2],
$dens_{regoUp,aqu}$	is the density of the upper layer of aquatic regolith [kg$_{DW}$ m^{-3}],
AC_{water}^{PM}	is the concentration of radionuclides in inorganic form adsorbed on suspended particulate matter in the water [Bq kg$_{DW}$$^{-1}$], and
$AC_{PM,org}$	is the concentration of radionuclides stored in particulate organic matter in the water [Bq kg$_{DW}$$^{-1}$].

5.2.5 Resuspension

Resuspension refers to the transport of radionuclide from aquatic sediments to the water column [Bq y^{-1}]. This mass flux affects radionuclides adsorbed to or stored in sediment solids, as well as radionuclides in the pore water. The flux of radionuclides from the upper sediments is a product of the total resuspension rate [kg$_{DW}$ y^{-1}] and the activity concentration of surface sediments:

$$\begin{aligned} Resuspension &= res_{rate} area_{obj,aqu} dens_{regoUp,aqu} AC_{regoUp,aqu} \\ Resuspension_{org} &= res_{rate} area_{obj,aqu} dens_{regoUp,aqu} AC_{regoUp,org,aqu} \end{aligned} \tag{5-24}$$

where

res_{rate}	is the gross area specific volumetric resuspension rate [m^3 m^{-2} y^{-3}],
$area_{obj,aqu}$	is the surface area of open water (sea basin, lake or stream) in the object [m^2],
$dens_{regoUp,aqu}$	is the density of the aquatic upper sediment layer [kg$_{DW}$ m^{-3}],
$AC_{regoUp,aqu}$	is the activity concentration of adsorbed and dissolved radionuclides in inorganic form in the upper aquatic sediments [Bq kg$_{DW}$$^{-1}$],
$AC_{regoUp,org,aqu}$	is the activity concentration in organic matter in the upper aquatic sediments [Bq kg$_{DW}$$^{-1}$].

5.2.6 Burial

If the sedimentation rate of particulate matter is larger than the resuspension rate, there is a net accumulation of sediment, and radionuclide from the upper sediment will be transferred to the oxygen-depleted sediment layer below. Burial [Bq y^{-1}] affects radionuclides adsorbed to or stored in sediment solids, as well as radionuclides in the pore water. The flux of radionuclides from the upper sediment by this process is a product of the net sedimentation rate [$kg_{DW}\ y^{-1}$] and the activity concentration of surface sediments:

$$\begin{aligned} Burial &= net_{sed} area_{obj,aqu} dens_{regoUp,aqu} AC_{regoUp,aqu}, \quad &| \ net_{sed} > 0 \\ Burial_{org} &= net_{sed} area_{obj,aqu} dens_{regoUp,aqu} AC_{regoUp,org,aqu}, \quad &| \ net_{sed} > 0 \end{aligned} \tag{5-25}$$

where

net_{sed}	is the area-specific volumetric net sedimentation rate [$m^3\ m^{-2}\ y^{-1}$],
$area_{obj,aqu}$	is the surface area of open water (sea basin, lake or stream) in the object [m^2],
$dens_{regoUp,aqu}$	is the density of the aquatic upper sediment layer [$kg_{DW}\ m^{-3}$],
$AC_{regoUp,aqu}$	is the activity concentration of adsorbed and dissolved radionuclides in inorganic form in the upper aquatic sediments [$Bq\ kg_{DW}^{-1}$],
$AC_{regoUp,org,aqu}$	is the activity concentration in organic matter in the upper aquatic sediments [$Bq\ kg_{DW}^{-1}$].

The area specific net sedimentation rate is as:

$$net_{sed} = sed_{rate} - res_{rate} \tag{5-26}$$

where

sed_{rate}	is the area specific volumetric gross sedimentation rate [$m^3\ m^{-2}\ y^{-1}$], and
res_{rate}	is the area specific volumetric gross resuspension rate [$m^3\ m^{-2}\ y^{-1}$].

5.2.7 Bioturbation

If resuspension of particulate matter is larger than the sedimentation, there is a net loss of sediments from the upper aquatic sediment layer. Due to bioturbation, the oxic horizon will penetrate into previously anoxic sediments, which then become part of the upper biologically active sediments. Bioturbation refers to the translocation of radionuclides from clay-gyttja to surface sediments, and it can be expressed as the product of the net resuspension rate [$kg_{DW}\ y^{-1}$] and the activity concentration of clay-gyttja:

$$\begin{aligned} Bioturb &= (res_{rate} - sed_{rate}) area_{obj,aqu} dens_{regoPG} AC_{regoPG,aqu} \quad &| \ net_{sed} < 0 \\ Bioturb_{org} &= (res_{rate} - sed_{rate}) area_{obj,aqu} dens_{regoPG} AC_{regoPG,org,aqu}, &| \ net_{sed} < 0 \end{aligned} \tag{5-27}$$

where

sed_{rate}	is the area specific volumetric gross sedimentation rate [$m^3\ m^{-2}\ y^{-1}$],
res_{rate}	is the area specific volumetric gross resuspension rate [$m^3\ m^{-2}\ y^{-1}$].
net_{sed}	is the area specific volumetric net sedimentation rate [$m^3\ m^{-2}\ y^{-1}$],
$area_{obj,aqu}$	is the surface area of open water (sea basin, lake or stream) [m^2],
$dens_{regoUp,aqu}$	is the density of aquatic clay-gyttja [$kg_{DW}\ m^{-3}$],
$AC_{regoUp,aqu}$	is the activity concentration of inorganic radionuclides in clay-gyttja [$Bq\ kg_{DW}^{-1}$],
$AC_{regoPG,org,aqu}$	is the activity concentration in organic matter in clay-gyttj [$Bq\ kg_{DW}^{-1}$].

5.2.8 Mineralisation

Mineralisation refers to the release of radionuclides from slow decomposition of organic matter in aquatic sediments. Through this process, radionuclides stored in organic matter in regolith layers are released to the pore water, and radionuclides in suspended organic matter are released to the sea/lake water. Mineralisation is assumed to follows first-order kinetics for homogenous substrates, and if it is assumed that trace elements are evenly distributed in the substrate, the release of all radionuclides will approximately follow that of carbon mineralisation. Therefore, the release of all radionuclides through mineralisation [$Bq\ y^{-1}$] can be expressed as a fractional release rate of the organic inventories:

$$
\begin{aligned}
&Mineralization_{water} = PM_{org} minRate_{water,PM} \\
&Mineralization_{regoUp,aqu} = RegoUp_{org,aqu} minRate_{regoUp,aqu} \\
&Mineralization_{regoPG,aqu} = RegoPG_{org,aqu} minRate_{regoPG,aqu}
\end{aligned} \tag{5-28}
$$

where

PM_{org} is the activity in suspended particulate organic matter in the water column [Bq],

$RegoUp_{org,aqu}$ is the activity in organic matter in the upper aquatic sediments [Bq],

$RegoPG_{org,aqu}$ is the activity in organic matter in aquatic clay-gyttja [Bq],

$minRate$ is the mineralisation rate of refractory organic matter in the water column, in surface sediments or in clay-gyttja [kgC kgC^{-1} y^{-1}].

5.2.9 Gas exchange with the atmosphere

There will be a gas exchange of inorganic carbon between the water column and the overlying atmosphere. The net rate of water-air gas exchange depends on the concentration gradient between the water-air boundary layer and a combination of turbulence and diffusion properties within the two layers. The area specific net flux of gas from the water into the overlying air layer, J [mol m^{-2} y^{-1}], can be described as (Cole and Caraco 1998):

$$
J = piston_{vel,aqu} \, (Conc_{surface} - Conc_{eqwater}) \tag{5-29}
$$

where

$piston_{vel,aqu}$ is the gas-exchange coefficient (or piston velocity), which is the height of the water that is equilibrated with the atmosphere per unit time for a given gas at a given temperature [m y^{-1}],

$Conc_{surface}$ is the concentration of gas in the surface water [mol m^{-3}], and

$Conc_{eqwater}$ is the concentration of gas in water in equilibrium with the atmosphere [mol m^{-3}].

The equilibrium concentration of gas in surface water ($Conc_{eqwater}$) will depend on the partial pressure of CO_2 in the atmosphere and the solubility of CO_2, which in turn depends on temperature, salinity, and atmospheric pressure (Weiss 1974). However, for a given set of environmental conditions it will be proportional to the concentration of the gas in the air.

The actual concentration of CO_2 gas in the water ($Conc_{surface}$) will depend on the total concentration of dissolved inorganic carbon, and the fraction of the inorganic carbon in the form of CO_2/H_2CO_3, which in turn primarily depends on pH. Thus for a given pH, the concentration of CO_2 in the water will be proportional to the total concentration of inorganic carbon.

As the activity (Bq) is directly proportional to the molar amount (mol) of a radionuclide, the above equation can be applied to the activity concentration of inorganic carbon. Below we list equations for the corresponding gross fluxes of inorganic carbon activity by considering bidirectional fluxes against zero concentrations:

$$
\begin{aligned}
\mathrm{J} &= piston_{vel,aqu}\left(Conc_{surface} - 0\right) - piston_{vel,aqu}\left(Conc_{eqwater} - 0\right) \\
&= Degassing - GasUptake
\end{aligned} \tag{5-30}
$$

5.2.10 Degassing

Degassing refers to the radionuclide flux caused by the gross exchange of solved gas in the water column with the atmosphere (sink). The gross flux from the water to the atmosphere [Bq y^{-1}] is a function of the dissolved C-14 activity in the form of CO_2/H_2CO_3, the piston velocity and the surface area of water in contact with the atmosphere:

$$Degassing_{aqu}^{14C} = AC_{water}^{D14C} f_{H_2CO_3,aqu} piston_{vel,aqu} area_{obj,aqu} \tag{5-31}$$

where

AC_{water}^{D14C}	is the activity concentration of inorganic carbon dissolved in the water column [Bq m^{-3}], see Equation 5-4),
$f_{H_2\,CO_3,aqu}$	is the fraction of dissolved inorganic carbon that is in the form of CO_2/H_2CO_3 [mol mol^{-1} or Bq Bq^{-1}],
$piston_{vel,aqu}$	is the gas exchange coefficient, (i.e. the height of the water that is equilibrated with the atmosphere per unit time) [m y^{-1}], and
$area_{ob\,j,aqu}$	is the surface area of open water (sea basin, lake or stream) in the object [m^2].

5.2.11 Gas uptake

Gas uptake refers to the radionuclide flux caused by the gross exchange of gas between the atmosphere (source) and gas solved in the water column. The gross flux from the atmosphere to the water [Bq y^{-1}] is a function of the equilibrium concentration of C-14 activity in water (see below), piston velocity and the surface area of the water body:

$$GasUptake_{aqu}^{14C} = AC_{equlib,aqu}^{14C} piston_{vel,aqu} area_{obj,aqu} \tag{5-32}$$

where

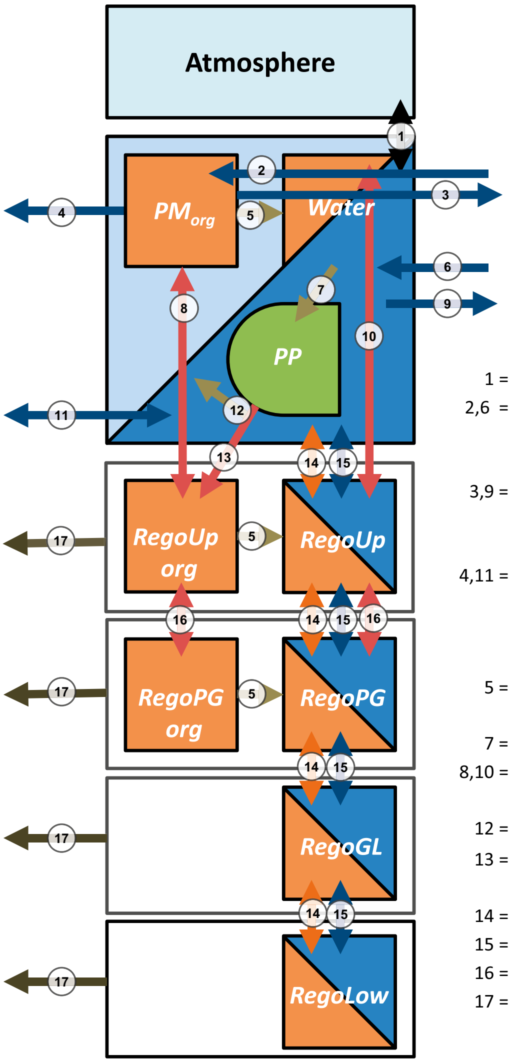
$AC_{equlib,aqu}^{14C}$	is the activity concentration in water expected at equilibrium with the atmosphere above the water [Bq m^{-3}],
$piston_{vel,aqu}$	is the gas exchange coefficient [m y^{-1}], and
$area_{ob\,j,aqu}$	is the surface area of open water (sea basin, lake or stream) in the object [m^2].

The equilibrium concentration in water ($AC_{equlib,aqu}^{14C}$) is a function of the atmospheric activity concentration [Bq m^{-3}] and a solubility constant of CO_2 [mol m^{-3} per mol m^{-3}]:

$$AC_{equlib}^{14C} = AC_{atmos,aqu}^{14C} solubilityCoef_{aqu} \tag{5-33}$$

where

$AC_{atmos,aqu}^{14C}$	is the activity concentration in air above the water [Bq m^{-3}], and
$solubilityCoef_{aqu}$	is a solubility coefficient [mol m^{-3} per mol m^{-3}].



1 = degassing/gas uptake
2,6 = horizontal advection into the biosphere object (import)
3,9 = horizontal advection out of the biosphere object (export)
4,11 = horizontal advective exchange between mire and aquatic ecosystems (within a biosphere object)
5 = mineralisation of refractory organic matter
7 = plant uptake
8,10 = sedimentation/resuspension
12 = litter respiration/release
13 = production of refractory litter
14 = diffusion
15 = vertical advection
16 = burial/bioturbation
17 = mire ingrowth

Figure 5-1. *A graphical representation of the radionuclide transport model used to simulate transport and accumulation in aquatic ecosystems. Boxes represents radionuclide inventories (Section 5.1) and arrows fluxes of radionuclides (Section 5.2). Inventories on the right represents radionuclides in inorganic form, and these are partitioned between adsorbed radionuclides (orange) and radionuclides in solution (blue). Radionuclides in organic form are represented by a solid state only (left column, including primary producers). Note that primary producers (PP) is made up of three compartments, namely radionuclides in plankton and in micro- and macro-phytobenthos. Arrows (and numbers) represents radionuclide fluxes between compartments and fluxes into and out of the system. These fluxes are linked to mass fluxes of gas (black), water (dark blue) and solid matter (dark red), to transitions between inorganic and organic forms of radionuclides (brown), to diffusion in soil pore water (orange), and to ingrowth of wetland vegetation (dark brown). The atmosphere serves as a source and sink of radionuclides. Note that radionuclides may enter aquatic ecosystems by discharge of groundwater into RegoLow (primary object) or by horizontal advective transport into the biosphere object (6,2, all objects).*

6 Mire ecosystem

In the Forsmark region the characteristics of the wetland ecosystems vary systematically with time since the wetland was formed (Hedenström and Sohlenius 2008), which is roughly proportional to the distance from the sea. Thus, the wetland areas associated with newly isolated shallow lakes are dominated by dense reed belts (*Phragmites australis*), which successively expands from the shores into the central parts of the lake. Reeds are replaced by vegetation communities dominated by *Bryales* and *Carex* species, forming the rich fens that are characteristic of Northern Uppland. The pH and contents of nutrients are relatively high in fen peat, which makes them suitable for cultivation after drainage (Berglund 1996). As the depth of the peat layer thickens with time, the influence of calcite-enriched groundwater diminishes. The rich fen vegetation is replaced by poor fen communities dominated by *Sphagnum* species. Once a raised bog has developed, production on the bog plane will be fed by rain and the connections to the groundwater table will be limited (or non-existent). The peat developed during ombrotrophic conditions has low nutritional value and low pH (Osvald 1937). Stages with trees may occur, primarily in the marginal parts of the fens (e.g. Bergström 2001). Occasionally trees may cover most of the mire surface (e.g. von Post and Granlund 1926), but these periods are relatively short and infrequent, and this is not considered a distinct successional stage in the mire ontogeny in Forsmark (Löfgren 2010).

Groundwater from the SFR repository can reach the mire ecosystem through direct discharge from the bedrock. However, the potential for accumulation in vegetation of radionuclides entering from below will level off as the bog plane rises. Therefore the treeless rich fen stage was considered to be most relevant for the safety assessment. From the mire, radionuclides can be transported with surface water to down-stream lakes and wetlands. In Chapter 3 (model structure) compartments and transport mechanisms which are important for the distribution of radionuclides in surface ecosystems are identified. In this chapter, the mathematical description of these compartments (Section 6.1) and fluxes (Section 6.2) in a mire ecosystem is presented. The biogeochemical cycle of carbon differs significantly from that of other elements (e.g. carbon is the principal structural component in peat, it is the main carrier of energy in the ecosystem, and it is present in gaseous form at ambient temperature). Consequently, the mathematical description for radiocarbon (C-14) frequently differs from that for other radionuclides. Water-borne transport of radionuclides by advection and diffusion is described in Section 4.2. That section also includes a description of radionuclide fluxes between biosphere objects, and between the lake and mire ecosystems within a biosphere object. A schematic representation of the sub-model of transport in mire ecosystems is presented at the end of the chapter (Figure 6-2).

6.1 Mathematical representation of radionuclide inventories

6.1.1 Mire Primary Producers [PP]

The compartment mire primary producers [Bq] represents radionuclides stored in plant biomass. The inventory primarily reflects radionuclide accumulation in bryophytes and vascular plants inhabiting treeless fens.

The activity concentration in terrestrial primary producers is expressed as the inventory of a radionuclide in primary producers per unit biomass carbon [Bq kgC^{-1}], (i.e. carbon specific activity):

$$AC_{PP,ter} = \frac{PP}{biom_{pp,ter}\,area_{obj,ter}}, \tag{6-1}$$

where

PP is the activity in mire vegetation [Bq],

$biom_{pp,ter}$ is the area-specific biomass of terrestrial primary producers [kgC m^{-2}], and

$area_{obj,ter}$ is the surface area of the mire [m^2].

6.1.2 Peat [RegoUp, $\mathbf{RegoUp_{org}}$, RegoPeat, $\mathbf{RegoPeat_{org}}$]

The radionuclide inventory in the upper, partly oxygenated and biologically active peat layer is represented by two compartments: $RegoUp_{ter}$ and $RegoUp_{ter,org}$. Similarly, radionuclides in deep anoxic peat are represented by *RegoPeat* and $RegoPeat_{org}$. The inventories in both regolith layers consist of radionuclides in solution, adsorbed on soil particles and incorporated into organic particulate matter.

For *all* regolith layers, the time to reach equilibrium between solid and aqueous phases is assumed to be short (time scale of days to months, Periáñez 2005) compared with the model time resolution (year). Moreover, dissolved low molecular weight organic carbon is typically mineralised within days or weeks (Howard 1991). Consequently, organic C-14 in the aqueous phase is assumed to be transformed to inorganic carbon within the time resolution of the model.

The activity concentration of the inorganic part of the organic mire sediments [Bq kg_{DW}^{-1}] is a function of the radionuclide inventory, the volume of sediments and the peat density. In general form this can be expressed as:

$$AC_{rego\_i,ter} = \frac{Rego\_i_{ter}}{vol_{rego\_i,ter}\, dens_{rego\_i,ter}}, \tag{6-2}$$

where

$Rego\_i$	is the activity of a radionuclide in inorganic form in the mire regolith layer *i* [Bq],
$dens_{rego\_i,ter}$	is the bulk density of the mire regolith layer *i* [kg_{DW} m^{-3}],
$vol_{rego\_i,ter}$	is the volume of the mire regolith layer *i* [m^3] defined as:

$$vol_{rego\_i,ter} = area_{obj,ter} z_{rego\_i,ter}, \tag{6-3}$$

where

$area_{obj,ter}$	is the surface area of covered by mire vegetation [m^2],
$z_{rego\_i,ter}$	is the depth of peat layer *i* [m], and
i	indicate the peat layer of interest. That is, *i = Up* and *Peat*, represents the upper biological active peat (*RegoUp*) and deep anoxic peat (*RegoPeat*), respectively.

The amount of the stable carbon stored in the anoxic deep peat layer at time t is calculated by solving the differential equation, with initial conditions of zero peat carbon at the time of emergence:

$$\frac{dRegoPeat^{SOC}}{dt} = Burial^{C} area_{obj,ter} - RegoPeat^{SOC} minRate_{regoPeat} \tag{6-4}$$

where

$Burial^{C}$	is the yearly input of soil organic carbon to the anoxic peat layer from the partly oxygenated peat layer per unit area [kgC m^{-2} y^{-1}], (see below),
$RegoPeat^{soc}$	is the amount of soil organic carbon in the peat layer [kgC], and
$minRate_{regoPeat}$	is the mineralisation rate of refractory organic carbon in the anoxic environment of the peat [kgC kgC^{-1} y^{-1}].

The thickness of the layer is derived from the amount of soil organic carbon, the area of the mire and properties of the peat:

$$z_{regoPeat} = \frac{RegoPeat^{SOC}}{area_{obj,ter}\, dens_{regoPeat} f_{C,peat}} \tag{6-5}$$

where

$f_{c,peat}$	is the fraction of carbon in peat [kgC kg_{DW}^{-1}].

Dissolved organic carbon is assumed to be transformed into inorganic carbon within the time resolution of the model (see above). Thus, all radionuclides in organic matter are stored in refractory solid organic matter. The activity concentration in solid organic matter [Bq kg_{DW}^{-1}] is defined as:

$$AC_{rego\_i,org,ter} = \frac{Rego\_i_{org,ter}}{vol_{rego\_i,ter}\, dens_{rego\_i,ter}}, \tag{6-6}$$

where

$Rego\_i_{org,ter}$ is the activity in soil organic carbon in regolith layer i [Bq].

The inventory of inorganic radionuclides in soil pores space is partitioned into an aqueous (dissolved) and a solid (adsorbed) phase, where the equilibrium between the two phases is described by a constant, element-specific solid/liquid distribution coefficient (K_d, Equation 3-1).

As all mire regolith layers are water saturated, the activity concentration in the pore water of a regolith compartment is the inventory in the solute phase [Bq] divided by the pore volume [m^3] of the layer:

$$AC^{D}_{rego\_i,ter} = \frac{Rego\_i_{ter}\, f^{D}_{rego\_i,ter}}{vol_{rego\_i,ter}\, poro_{rego\_i,ter}} \tag{6-7}$$

where

$poro_{rego\_i,ter}$ is the porosity of the mire regolith layer i [m^3 m^{-3}], and

$f^{D}_{rego\_i,ter}$ is the fraction of radionuclides in solute phase for mire regolith layer i [Bq Bq^{-1}].

The fraction of a radionuclide that is in inorganic form in the solute phase in the mire regolith layer i [Bq Bq^{-1}] is a function of the distribution coefficient and the physical properties of the regolith layer (Appendix A):

$$f^{D}_{rego\_i,ter} = \frac{1}{1 + \frac{K_{d,rego\_i,ter}\, dens_{rego\_i,ter}}{poro_{rego\_i,ter}}} \tag{6-8}$$

where

$K_{d,rego\_i,ter}$ is the distribution coefficient in the mire regolith layer i [m^3 kg_{DW}^{-1}]

6.1.3 Post-glacial deposits [RegoPG, $RegoPG_{org}$]

The radionuclide inventory in post glacial clay-gyttja in mire ecosystems is represented by two compartments, namely $RegoPG_{ter}$ and $RegoPG_{org,ter}$. This regolith layer represents water saturated, anaerobic, sediments located directly below the deep peat layer. As with the peat layers, the inventory in this regolith layer consist of radionuclides in solution, radionuclides adsorbed on soil particles and radionuclides incorporated into organic matter.

The activity concentration of a radionuclide in inorganic form, $AC_{regoPG,ter}$ [Bq kg_{DW}^{-1}], the activity concentration of a radionuclide in solid organic matter, $AC_{regoPG,org,ter}$ [Bq kg_{DW}^{-1}], and the activity concentration of a radionuclide dissolved in the pore water $AC^{D}_{regoPG,ter}$ [Bq m^{-3}] of post-glacial clay-gyttja in mire ecosystems are given by Equations 6-2, 6-6 and 6-7, respectively (*i=regoPG)*. Similarly, the fraction of a radionuclide in inorganic form in the solute phase, $f^{D}_{regoPG,ter}$ [Bq Bq^{-1}], is expressed using Equation 6-8.

6.1.4 Glacial deposits [RegoGL]

The radionuclide inventory in the glacial clay in mire ecosystems is represented by the compartment *RegoGL*. This regolith layer represents water-saturated, anaerobic, minerogenic glacial deposits and it is typically located between post-glacial clay gyttja and the till layer (see below). The inventory consists of radionuclides in solution and radionuclides adsorbed on soil particles. As dissolved organic matter is assumed to be mineralised within the time resolution of the model, no radionuclides in organic matter are assumed to reach this regolith layer.

The activity concentration of a radionuclide in inorganic form, $AC_{regoGL,ter}$ [Bq kg_{DW}^{-1}], and of a dissolved radionuclide in the pore water, $AC^{D}_{regoLow,ter}$ [Bq m^{-3}], of glacial deposits in mire ecosystems are described by Equations 6-2 and 6-7, respectively (*i=regoGL).* Similarly, the fraction of a radionuclide in inorganic form in the solute phase, $f^{D}_{regoLow,ter}$ [Bq Bq^{-1}], is expressed with Equation 6-8.

6.1.5 Till [RegoLow]

The radionuclide inventory in the deepest regolith layer in a mire ecosystem is represented by the compartment *RegoLow*. This regolith layer represents loose, unsorted minerogenic, wate- saturated (anoxic) sediments overlying the bedrock (primarily till). The inventory consists of radionuclides in solution and radionuclides adsorbed on soil particles. As dissolved organic matter is assumed to be mineralised within the time resolution of the model, no radionuclides in organic matter are assumed to reach this regolith layer.

The activity concentration of a radionuclide in inorganic form, $AC_{regoLow,ter}$ [Bq kg_{DW}^{-1}], and of a dissolved radionuclide in the pore water $AC^{D}_{regoLow,ter}$ [Bq m^{-3}] of till in mire ecosystem are described by Equations 6-2 and 6-7, respectively (*i=regoLow).* Similarly, the fraction of a radionuclide in inorganic form in the solute phase, $f^{D}_{regoLow,ter}$ [Bq Bq^{-1}], is expressed with Equation 6-8.

6.2 Mathematical representation of radionuclide fluxes

In this section, radionuclide fluxes associated with plant uptake, litter production, litter decomposition, burial, mineralisation of refractory organic matter, gas exchange with the atmosphere and the ingrowth of mire vegetation are described in mathematical terms. Radionuclide fluxes associated with advective and diffusive transport are described in Chapter 4. Radionuclide fluxes within the atmosphere above the mire ecosystem are described in Chapter 8.

6.2.1 Plant uptake

Plant uptake represents fixation of inorganic radiocarbon and other radionuclides in plant biomass as a result of net primary production. It includes fixation of atmospheric C-14 and root uptake of all radionuclides (see Section 3.2).

Plant uptake of atmospheric C-14 [Bq y^{-1}] is the product of net primary production [kgC y^{-1}] and the activity concentration in newly synthesised biomass resulting from fixation of atmosphere carbon [Bq kgC^{-1}], scaled by the fraction for atmospheric carbon assimilation:

$$Plant^{14C}_{Uptake,ter} = AC^{14C}_{NPP,atm} NPP_{ter} area_{obj,ter} \left(1 - f_{rootUptake}\right) \quad (6\text{-}9)$$

where

$AC^{14C}_{NPP,atm}$	is the activity concentration of C-14 in newly synthesised biomass resulting from fixation of atmospheric carbon [Bq kgC^{-1}].
NPP_{ter}	is the area-specific net primary production in a mire ecosystem [kgC m^{-2} y^{-1}],
$area_{obj,ter}$	is the surface area of the mire [m^2], and
$f_{rootUptake}$	is the fraction of carbon assimilated via root uptake [kgC kgC^{-1}].

Since isotopic discrimination of C-14 (relative to stable C-12) is small during photosynthesis, plant allocation and biomass production (Peterson 1980), the C-14 concentration in synthesised biomass can be approximated with the specific activity concentration of C-14 in the canopy atmosphere, $SA^{14C}_{atmos,ter}$ [Bq kgC^{-1}]:

$$AC^{14C}_{NPP,atm} = SA^{14C}_{atmos,ter} \quad (6\text{-}10)$$

Similarly, root uptake of C-14 [Bq y^{-1}] is the product of net primary production [kgC y^{-1}] and the activity concentration in synthesised biomass resulting from fixation of pore water carbon [Bq kgC^{-1}], scaled by the fraction for root carbon assimilation:

$$Plant^{14C}_{uptake,ter} = AC^{14C}_{NPP,root} NPP_{ter} area_{obj,ter} f_{rootUptake} \tag{6-11}$$

where

$AC^{14C}_{NPP,root}$ is the activity concentration in newly synthesised root biomass resulting from root uptake of carbon [Bq kgC^{-1}],

Assuming that root uptake is confined to partly oxygenated peat layer and that the discrimination between root uptake of C-14 and stable carbon (C-12) is marginal, the C-14 activity concentration in newly synthesised biomass resulting from root uptake can be approximated by the specific activity of C-14 in DIC in the surface peat layer $SA^{D14C}_{regoUp,ter}$. This specific activity is by definition the quotient of the activity concentration of inorganic carbon in the solute phase [Bq m^{-3}] and the concentration of dissolved stable inorganic carbon [kgC m^{-3}]:

$$SA^{D14C}_{regoUp,ter} = \frac{AC^{D14C}_{regoUp,ter}}{conc_{DIC,regoUp,ter}} \tag{6-12}$$

where

$SA^{D14C}_{regoUp,ter}$ is the activity concentration of dissolved inorganic carbon in surface peat [Bq m^{-3}],

$conc_{DIC,regoUp,ter}$ is the the concentration of dissolved inorganic carbon in surface peat [kgC m^{-3}].

For other radionuclides, root uptake [Bq y^{-1}] from surface peat can also been calculated from net primary production [kgC y^{-1}] and the activity concentration in synthesised biomass [Bq kgC^{-1}]:

$$Plant^{RN}_{uptake,mire} = AC^{RN}_{NPP,root} NPP_{ter} area_{obj,ter} \tag{6-13a}$$

where

$AC^{RN}_{NPP,root}$ is the activity concentration in synthesised biomass [Bq kgC^{-1}].

Assuming that newly synthesised biomass is in equilibrium with the radionuclide concentration in the biologically active surface peat, the activity concentration in newly synthesised biomass can be calculated from the empirical equilibrium concentration ratio (Tröjbom et al. 2013):

$$AC^{RN}_{NPP,root} = AC^{RN}_{regoUp,ter,tot} CR_{ter,pp} \tag{6-14}$$

where

$CR_{ter,pp}$ is the empirical equilibrium concentration ratio [kg_{DW} kgC^{-1}], and

$AC^{RN}_{regoUp,ter,tot}$ is the total activity concentration in surface peat [Bq kg_{DW}^{-1}] (including the radionuclide in organic and inorganic forms):

$$AC^{RN}_{regoUp,ter,tot} = AC^{RN}_{regoUp,ter} + AC^{RN}_{regoUp,org,tot} \tag{6-15}$$

As plants primarily access elements in inorganic form (Kabata-Pendias 2011), root uptake is modelled exclusively from the inorganic pool. However, as empirical plant-soil relationships are traditionally defined by concentration ratios between the plant and the total soil concentration, including elements in organic form, Equation 6-13 may yield unrealistic or even impossible rates of plant uptake. This is particularly a problem for parameter combinations that yield a high rate of biological uptake (high CR and NPP) and a large accumulation in organic matter in surface peat (i.e. a high litter production and a low rate of mineralisation and/or burial, see below). Therefore the equation for plant uptake was modified to include an upper limit on the uptake, $Max\_Plant_{\text{uptake}}$:

$$Plant^{RN}_{uptake,mire} = \min(AC^{RN}_{NPP,root} NPP_{ter} area_{obj,ter}, MaxPlant_{uptake}) \tag{6-13b}$$

The plant tissue concentration of essential elements (nutrients) such as chlorine is often regulated (Van den Hoof and Thiry 2012). Consequently, the chloride concentration varies significantly between plant species, but has little to do with the concentrations in boreal soils (Edwards et al. 1981). Thus, the plant nutrient concentration can be used to derive an upper limit for net uptake of chlorine, and the Cl-36 concentration in newly synthesised biomass can be approximated from the specific Cl-36 activity concentration of dissolved inorganic chloride in the surface peat layer:

$$MaxPlant_{uptake}^{36Cl} = conc_{Cl,plant}\, NPP_{ter} area_{obj,ter}\, SA_{regoUp,ter}^{D36Cl} \quad (6\text{-}16)$$

where

$conc_{Cl,plant}$ is the concentration of stable chloride in newly synthesised biomass [kgCl kgC^{-1}], and the specific activity $SA_{regoUp,ter}^{D36Cl}$ [Bq $kgCl^{-1}$] is defined as:

$$SA_{regoUp,ter}^{D36Cl} = \frac{AC_{regoUp,ter}^{D,36Cl}}{conc_{Cl,porewat}} \quad (6\text{-}17)$$

where

$SA_{regoUp,ter}^{D36Cl}$ is the concentration of dissolved inorganic Cl-36 in surface peat [Bq m^{-3}], and

$conc_{Cl,porewat}$ is the pore water concentration of stable chloride in surface peat [kg_{Cl} m^{-3}],

This approach cannot be extended to non-regulated uptake of trace elements. Therefore, for other radionuclides than Cl-36 (and C-14), it was simply assumed that plants cannot take up radionuclides at a rate that would deplete 99% or more of the inorganic soil inventory in a year:

$$MaxPlant_{uptake}^{RN} = RegoUp_{ter}^{RN} MaxRate_{uptake} \quad (6\text{-}18)$$

where

$MaxRate_{uptake}$ is a rate constant the depletes 99% of the inorganic radionuclide inventory in a year. It takes the numeric value of 4.6 [y^{-1}].

6.2.2 Litter respiration [C-14] / release [other RN]

Fast (within a year) decomposition of plant litter leads to mineralisation and releases of radionuclides to the water in the saturated surface peat. On the time scale of a year the primary producer biomass is assumed to be in equilibrium, which means that the net primary production (NPP, kgC y^{-1}) is balanced by an equal loss of biomass through consumption and litter production.

The flux of C-14 associated with the fast decomposition of dead organic matter, $Litter_{resp,ter}^{14C}$ [Bq y^{-1}], can be expressed as the product of the activity concentration in plant biomass, and the net production of labile (non-refractory) carbon.

$$Litter_{resp,ter}^{14C} = AC_{PP,ter}^{14C} NPP_{ter} area_{obj,ter} (1 - f_{refrac,ter}) \quad (6\text{-}19)$$

where

$AC_{PP,ter}^{14C}$ is the activity concentration of C-14 in mire vegetation [Bq kgC^{-1}],

NPP_{ter} is the area-specific net primary production of mire vegetation [kgC m^{-2} y^{-1}],

$area_{obj,ter}$ is the surface area of the mire and

$f_{refrac,ter}$ is the fraction of plant biomass that is refractory and does not mineralise within a year [kgC kgC^{-1}].

When easily metabolised carbon sources are respired, radionuclides stored in the organic matter are released to the water saturated surface peat. The release of radionuclides in organic matter will be roughly proportional to the respired carbon. However, radionuclides that are stored in inorganic form in plant tissue will be released at a greater rate than the carbon that is mineralised, whereas other fractions of radionuclides may be subject to a much slower release.

The release of a radionuclide other than C-14 from dead organic matter to the water column $Litter_{release,ter}^{RN}$ [Bq y^{-1}] is a function of the activity concentration in plant biomass, the net production of labile (non-refractory) carbon in biomass and a correction factor for diminishment/enrichment of the radionuclide in the refractory organic matter:

$$Litter_{release,ter}^{RN} = AC_{PP,ter}^{RN} NPP_{ter} area_{obj,ter} (1 - f_{refrac,ter}) df_{decomp,ter} \quad (6\text{-}20)$$

where

$AC_{PP,ter}^{RN}$ is the activity concentration in the mire vegetation [Bq kgC^{-1}], and

$df_{decomp,ter}$ is the fraction of the radionuclide that is available for release through mineralisation of labile organic matter [Bq Bq^{-1}].

6.2.3 Litter production

Litter production $Litter_{prod,ter}^{14C}$ [Bq y^{-1}] represents the transfer of a radionuclide from dead organic matter (from primary producers) to surface peat in mire ecosystems. It can be expressed as the product of the activity concentration in biomass, the net primary production and the fraction of biomass that is not easily decomposed:

$$Litter_{prod,ter}^{14C} = AC_{PP,ter}^{14C} NPP_{ter} area_{obj,ter} f_{refrac,ter} \tag{6-21}$$

where

$AC_{PP,ter}^{14C}$ is the activity concentration of C-14 in mire vegetation [Bq kgC^{-1}],

NPP_{ter} is the area specific net primary production of mire vegetataion [kgC m^{-2} y^{1}],

$area_{obj,ter}$ is the surface area of the mire [m^2],

$f_{refrac,ter}$ is the fraction of refractory organic matter of mire vegetataion [kgC kgC^{-1}].

From mass balance it follows that the flux of radionuclides (other than C-14) in refractory litter equals the total flux in dead organic matter less radionuclides released through decomposition of the labile pool (within a year):

$$\begin{aligned} Litter_{prod,ter}^{RN} &= AC_{PP,ter}^{RN} NPP_{ter} area_{obj,ter} - Litter_{release,ter}^{RN} \\ &= AC_{PP,ter}^{RN} NPP_{ter} area_{obj,ter} \left(1 - \left(1 - f_{refrac,ter}\right) df_{decomp,ter}\right) \end{aligned} \tag{6-22}$$

where

$AC_{PP,ter}^{14C}$ is the activity concentration in mire vegetation [Bq kgC^{-1}], and

$df_{decomp,ter}$ is the discrimination factor during decomposition [Bq Bq^{-1}].

6.2.4 Burial

If the input of refractory organic carbon to the upper peat layer [kgC y^{-1}] is larger than the rate of mineralisation from this layer, organic matter will accumulate. However, as the depth of the upper peat layer ($z_{regoUp,ter}$) is limited by penetration of oxygen from the atmosphere, the amount of organic carbon in this layer is at an approximate steady-state, and the surplus of organic matter will be transferred to deeper peat horizons. From mass balance, it follows that the rate of carbon burial per unit area [$Burial^{C}$, kgC m^{-2} y^{-1}] equals the input of refractory carbon from litter less the loss of carbon from mineralisation in surface peat (see below):

$$Burial^{C} = NPP_{ter} f_{refrac,ter} - RegoUp_{ter}^{SOC} minRate_{regoUp,ter} \tag{6-23}$$

where

NPP_{ter} is the area specific net primary production of the mire vegetation [kgC m^{-2} y^{-1}],

$f_{refrac,ter}$ is the fraction of refractory organic matter [kgC kgC^{-1}],

$minRate_{regoUp,ter}$ is the mineralisation rate of refractory organic surface peat [kgC kgC^{-1} y^{-1}], and

$RegoUp_{ter}^{SOC}$ is the area-specific amount of soil organic carbon in the surface peat [kg C m^{-2}]:

$$RegoUp_{ter}^{SOC} = z_{regoUp,ter} dens_{regoUp,ter} f_{C,peat} \tag{6-24}$$

where

$z_{regoUp,ter}$ is the thickness of surface peat [m],

$dens_{regoUp,ter}$ is the density of surface peat layer [kg$_{DW}$ m^{-3}], and

$f_{c,peat}$ is the fraction of carbon in surface peat [kgC kg$_{DW}$$^{-1}$].

The flux of other radionuclides from surface peat ($RegoUp_{org,ter}$) to underlying anoxic peat ($RegoPeat_{org}$) follows the flow of soil organic matter. Consequently, *Burial* [Bq y^{-1}] can be expressed as the product of the net accumulation rate of peat [kgC y^{-1}] and the carbon specific activity concentration in the upper peat layer [Bq kgC^{-1}]:

$$\begin{gathered} Burial_{org} = AC^{C}_{regoUp,org,ter} Burial^{C} area_{obj,ter} \\ Burial = AC^{C}_{regoUp,ter} Burial^{C} area_{obj,ter} \end{gathered} \tag{6-25}$$

where

$area_{obj,ter}$ is the surface area of the mire [m^2], and

$AC^{C}_{regoUp,org,ter}$ is the carbon specific activity concentration of surface peat [Bq kgC^{-1}]:

$$\begin{gathered} AC^{C}_{regoUp,org,ter} = \frac{RegoUp_{org,ter}}{RegoUp^{SOC}_{ter} area_{obj,ter}} \\ AC^{C}_{regoUp,ter} = \frac{RegoUp_{ter}}{RegoUp^{SOC}_{ter} area_{obj,ter}} \end{gathered} \tag{6-26}$$

where

$RegoUp_{org,ter}$ is the activity in soil organic carbon in surface peat [Bq],

$RegoUp_{ter}$ is the activity in inorganic form in surface peat [Bq],

$RegoUp^{soc}_{ter}$ is the area-specific amount of soil organic carbon in surface peat [kgC m^{-2}]:

6.2.5 Mineralisation

Mineralisation refers to the release of radionuclides from slow decomposition of refractory organic matter. Through this process, radionuclides stored in organic matter in regolith layers are released to the pore water. Mineralisation is assumed to follow first-order kinetics for homogenous substrates, and if it is assumed that trace elements are evenly distributed in the substrate, the release of all radionuclides will approximately follow that of carbon mineralisation. Thus, the release of each radionuclide through mineralisation [Bq y^{-1}] can be expressed as a fractional release rate of the organic inventories:

$$\begin{gathered} Mineralisation_{regoUp,ter} = RegoUp_{org,ter} minRate_{regoUp,\mathrm{ter}} \\ Mineralisation_{regoPeat,ter} = RegoPeat_{org,ter} minRate_{regoPeat,ter} \\ Mineralisation_{regoPG,ter} = RegoPG_{org,ter} minRate_{regoPG,ter} \end{gathered} \tag{6-27}$$

where

$RegoUp_{org,ter}$ is the activity in organic matter in surface peat [Bq],

$RegoPeat_{org,ter}$ is the activity in organic matter in deep peat [Bq],

$RegoPG_{org,ter}$ is the activity in organic matter in mire clay-gyttja [Bq],

$minRate_i$ is the mineralisation rate of refractory organic matter in surface peat, deep peat or mire clay-gyttja [kgC kgC^{-1} y^{-1}].

6.2.6 Degassing [C-14]

There will be a gas exchange of inorganic carbon between the pore-water in the upper peat layer and the overlying atmosphere (see Gas exchange with the atmosphere in Chapter 5). The gross fluxes of inorganic carbon activity to and from the atmosphere can be described by partitioning the net flux into two fluxes in opposite directions against zero concentrations.

Terrestrial degassing refers to the radionuclide flux caused by the gross exchange of solved gas in the pore water of surface peat with the atmosphere (sink). The gross flux from the water to the atmosphere [Bq y^{-1}] is a function of the dissolved C-14 activity in the form of CO_2/H_2CO_3, the piston velocity and the surface area the mire in contact with the atmosphere:

$$Degassing^{14C}_{ter} = AC^{D14C}_{regoUp,ter} f_{H_2CO_3,ter} piston_{vel,ter} area_{obj,ter} \tag{6-28}$$

where

$AC^{D14C}_{regoUp,ter}$	is the activity concentration of inorganic carbon dissolved in the pore water of surface peat [Bq m^{-3}],
$f_{H_2CO_3,ter}$	is the fraction of dissolved inorganic carbon that is in the form of CO_2/H_2CO_3 in pore water of surface peat [mol mol^{-1} or Bq Bq^{-1}],
$piston_{vel,ter}$	is the gas exchange coefficient for peat in contact with the atmosphere (i.e. the height of the upper peat layer that is equilibrated with the atmosphere per unit time) [m y^{-1}], and
$area_{obj,ter}$	is the area of the mire [m^2].

6.2.7 Gas uptake

Gas uptake refers to the radionuclide flux caused by the gross exchange of gas between the atmosphere (source) and gas solved in the pore water of saturated surface peat. The gross flux from the atmosphere to the surface peat [Bq y^{-1}] is a function of the equilibrium concentration of C-14 activity in pore water (see below), piston velocity and the surface area of the mire:

$$GasUptake^{14C}_{ter} = AC^{14C}_{equlib,ter} piston_{vel,ter} area_{obj,ter} \tag{6-29}$$

where

$AC^{14C}_{equlib,ter}$	is the activity concentration in surface water expected at equilibrium with the atmosphere above the mire [Bq m^{-3}],
$piston_{vel,ter}$	is the gas exchange coefficient for the mire [m y^{-1}], and
$area_{obj,ter}$	is the area of the mire [m^2].

The equilibrium concentration in pore water (AC^{14C}_{equlib}) can be expressed as a function of the atmospheric concentration [Bq m^{-3}] and solubility constant [Bq m^{-3} per Bq m^{-3}]:

$$AC^{14C}_{equlib} = AC^{14C}_{atmos,ter,CA} solubilityCoef_{ter} \tag{6-30}$$

where

$AC^{14C}_{atmos,ter,CA}$	is the activity concentration in air (canopy layer) above the mire [Bq m^{-3}], and
$solubilityCoef_{ter}$	is a solubility coefficient for the upper peat layer [mol m^{-3} per mol m^{-3}].

6.2.8 Transition from lake to mire (terrestrialisation)

Wetland vegetation will colonise sea and lake shores and bottoms. The establishment of emergent and floating-leaved vegetation in closed bays and lakes will generate litter that accumulates in soft bottoms. Due to oxygen-depleted conditions decomposition is slow. Thus, peat starts to build up and ultimately fills the lake basin. The systematic changes associated with the transition from lake to wetland are captured by a coordinated change in the parameters describing the area and water depth of the lake (water column) and the area and depth of the parts of the lake covered by wetland vegetation (mire) (see Figure 6-1 and SKB 2014a).

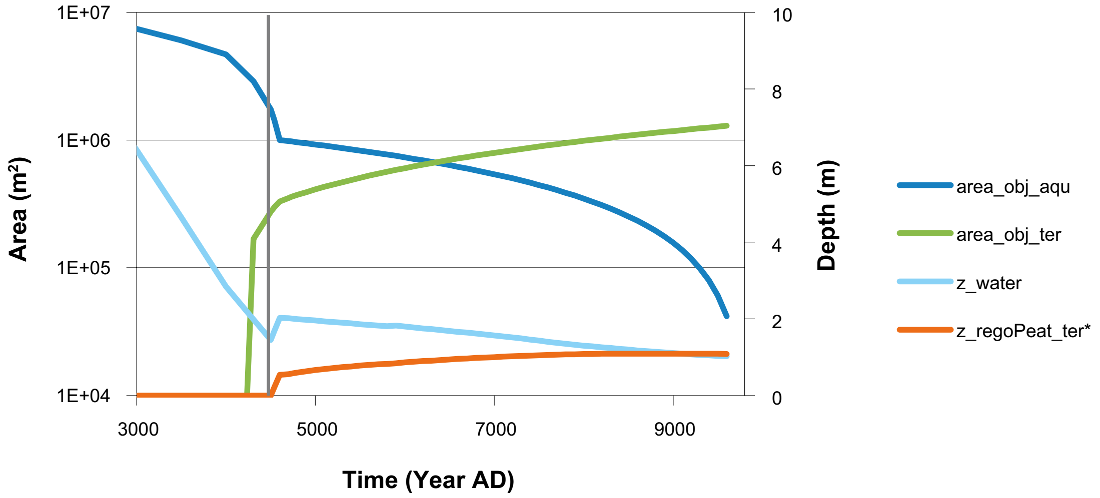


Figure 6-1. *Development of biosphere object 116 from a sea basin to lake/mire ecosystem. Grey vertical line indicates time of isolation (~ 4500 AD). Object areas are displayed on a logarithmic scale. The aquatic area (dark blue,* $area_{obj,aqu}$*) of the object represents open water (sea or lake/stream), whereas the terrestrial area (green,* $area_{obj,ter}$*) represents areas covered by wetland vegetation. At steady-state the part of the lake that is covered by wetland vegetation will be filled by peat with a thickness corresponding to the depth of the covered lake basin (z_regoPeat_ter*). At 9800 AD mire vegetation has colonised the entire lake basin and the aquatic ecosystem has been reduced to a stream.*

6.2.9 Ingrowth of mire vegetation

In the model, the ingrowth rate of emerged wetland vegetation [m^2 y^{-1}] (and the development of soft bottoms) is used to capture the areal transition from an aquatic to a mire ecosystem. Assuming that the radionuclide concentration is homogenous in the upper aquatic sediments, the amount of radionuclide that is covered by mire vegetation each year will be proportional to the area-specific activity concentration in the sediments [Bq m^2] and the rate of horizontal peat growth into the lake [m^2 y^{-1}]. Thus, the transfer of radionuclides [Bq y^{-1}] from the two uppermost aquatic sediments can be expressed as the product of the area-specific activity concentration in the upper aquatic sediments [Bq m^{-2}] and the rate of horizontal mire vegetation expansion into the lake [m^2 y^{-1}]:

$$Ter_{ingrowth,PG} = \frac{RegoUp_{aqu} + RegoPG_{aqu}}{area_{obj,sed,aqu}} Ter_{growth},$$

$$Ter_{ingrowth,PG,org} = \frac{RegoUp_{org,aqu} + RegoPG_{org,aqu}}{area_{obj,sed,aqu}} Ter_{growth}, \qquad (6\text{-}31)$$

where

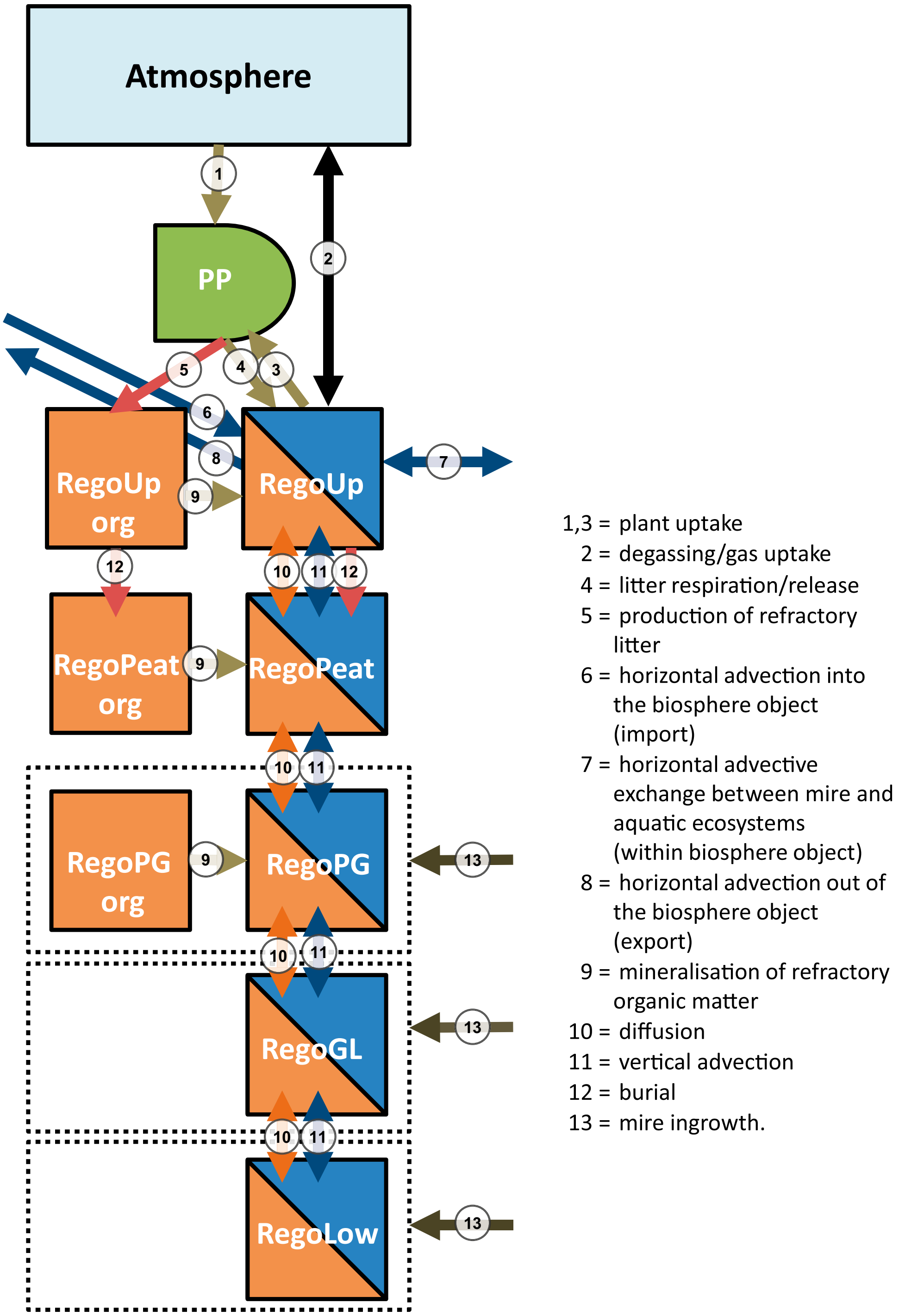
$RegoUp_{aqu}$ $RegoPG_{aqu}$	is the inorganic activity in aquatic regolith compartments [Bq],
$RegoUp_{org,aqu}$ $RegoPG_{org,aqu}$	is the corresponding activity in organic matter [Bq],
$area_{obj,sed,aqu}$	is the surface area of the aquatic sediments [m^2] (Equation 5-11),
$area_{obj,ter}$	is the surface area of the mire [m^2], and
Ter_{growth}	is the horizontal rate of mire vegetation ingrowth [m^2 y^{-1}]:

$$Ter_{growth} = \frac{\partial area_{obj,ter}}{\partial t}$$

The transfer of radionuclides from minerogenic aquatic sediments (glacial clay and till) to the corresponding mire regolith layer, associated with the expansion of wetland vegetation, can be described by similar expressions:

$$Ter_{ingrowth,GL} = \frac{RegoGL}{area_{obj,sed,aqu}} Ter_{growth},$$

$$Ter_{ingrowth,regoLow} = \frac{RegoLow}{area_{obj,sed,aqu}} Ter_{growth}, \qquad (6\text{-}32)$$



1,3 = plant uptake
2 = degassing/gas uptake
4 = litter respiration/release
5 = production of refractory litter
6 = horizontal advection into the biosphere object (import)
7 = horizontal advective exchange between mire and aquatic ecosystems (within biosphere object)
8 = horizontal advection out of the biosphere object (export)
9 = mineralisation of refractory organic matter
10 = diffusion
11 = vertical advection
12 = burial
13 = mire ingrowth.

Figure 6-2. *A graphical representation of the radionuclide transport model used to simulate transport and accumulation in mire ecosystems. Boxes represent radionuclide inventories (Section 6.1) and arrows fluxes of radionuclides (Section 6.2). Inventories on the right represents radionuclides in inorganic form, and these are partitioned between adsorbed radionuclides (orange) and radionuclides in solution (blue). Radionuclides in organic form are represented by a solid state only (left column, including primary producers). Arrows (and numbers) represent radionuclide fluxes between compartments and fluxes into and out of the system. These fluxes are linked to mass fluxes of gas (black), water (dark blue) and solid matter (dark red), to transitions between inorganic and organic forms of radionuclides (brown), to diffusion in soil pore water (orange), and to ingrowth of wetland vegetation (dark brown). The atmosphere serves as a source and sink of radionuclides. Note that radionuclides may enter mire ecosystems by discharge of groundwater into RegoLow (primary object) or by horizontal advective transport into the biosphere object (6, all objects).*

7 Agricultural ecosystem

Agricultural land is the most intensively managed land type in the landscape and a major provider of food for humans either directly as crop production or as production of fodder for livestock. Arable land includes intensively cultivated land and pastures or meadows. The arable land is regularly ploughed and harvested, whereas grasslands used for livestock grazing or hay-making are less managed (Löfgren 2010). Today, the largest arable land unit in the Forsmark area is found on clayey till. Smaller areas of mostly abandoned arable land are also found on soils on sedimentary deposits in the area. These are young and of the Cambisol type (Hedenström and Sohlenius 2008).

Based on knowledge of present and past agricultural practices, it is possible to identify several exposure pathways relating to radionuclides in the future Forsmark landscape (see Chapter 3). These exposure pathways can then be combined into sets representing exposed populations or agricultural land-use variants. For example, discharge of deep groundwater typically occurs in the topographically low-lying areas associated with lake-mire complexes (Bosson et al. 2010). These areas may be used to collect wetland grasses and sedges (i.e. hay) for winter fodder (Section 7.1). When the discharge area has emerged sufficiently above the sea level (to make salt water intrusions unlikely), draining and cultivation of a lake-mire system is considered a relevant exposure pathway (Section 7.2). Large-scale irrigation of cereals or fodder producing land is rare in central Sweden. However potatoes and vegetables are frequently irrigated, and irrigation of a kitchen garden is considered as a separate exposure pathway. On this scale, fertilisation with seaweed, or ash from combustion of biofuel, are also considered plausible exposure pathways (Section 7.3).

In Chapter 3 compartments and transport mechanisms for radionuclides in agricultural ecosystems were identified. In this chapter, the mathematical description of these compartments and fluxes is presented for three separate cultivation systems.

The mathematical models describe dynamic transport and accumulation of radionuclides (by differential equations) in the uppermost biologically active layer of cultivated soil only. Equilibrium conditions are assumed for plant uptake and litter fall release, and the radionuclide distribution between plants and soil is described by an algebraic expression (Appendix C). Moreover, it is assumed that radionuclides that are exported from the system through harvest are recycled back to the soil through fertilisation (cautiously ignoring losses that may occur during ingestion, decomposition and handling of manure). The solutions to the differential equations describing radionuclide accumulation in cultivated soils are relatively simple. Algebraic expressions can therefore be used to evaluate the consequences of exposure through cultivation of soil at each time point in the assessment period, without having to run dynamic simulations.

The biogeochemical cycle of carbon differs significantly from that of other elements (e.g. carbon is a structural component in soil organic matter and it is present in gaseous form at ambient temperature). Consequently the mathematical description relating to radiocarbon (C-14) occasionally differs from that for other radionuclides. A schematic representation of the sub-model of transport in agricultural ecosystems is presented in Figure 7-1.

7.1 Infield-outland agriculture

The agrarian infield–outland system is characterized by intensively fertilised arable fields and enclosed meadows (infield) on the one hand, and livestock grazing nearby pastures and forests (outlands) on the other hand. The key principle behind this system, which characterized agricultural practices in Scandinavia from the Iron Age to the 18th century, was to fertilise the arable land with nutrients from meadows and pastures, by the use of animal manure as organic fertiliser (Welinder et al. 1998). Thus, the agricultural infield-outland system offers a plausible conceptual model for exposure pathways from wetland vegetation to domestic animals and agricultural soil (Saetre et al. 2013).

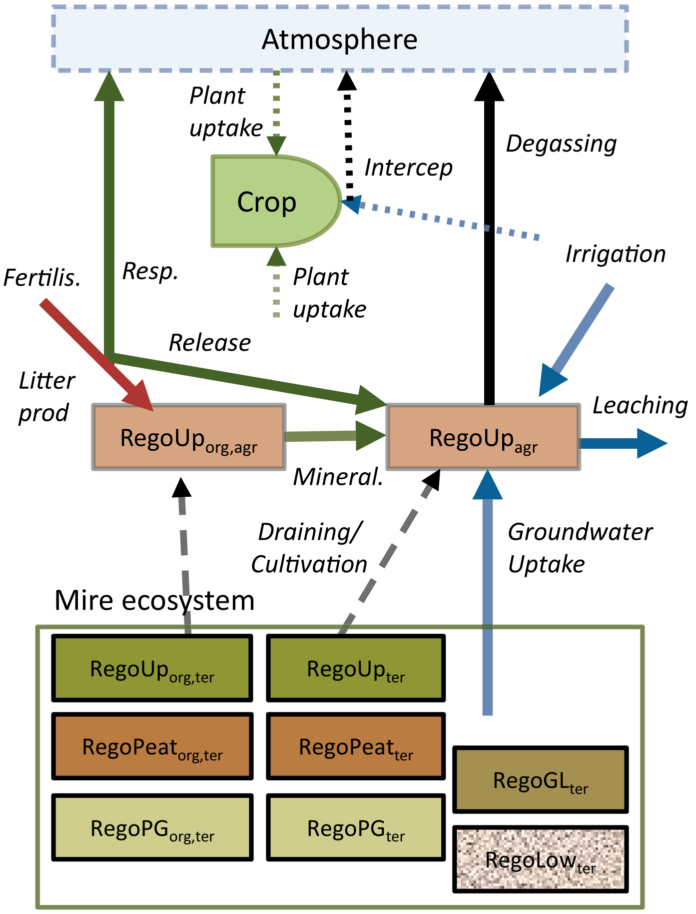


Figure 7-1. *Conceptual Model of transport and accumulation in cultivated soil. Three different agricultural systems are represented 1) fertilisation with hay/manure (infield-outland agriculture), 2) draining and cultivation of a lake-mire system (industrial age agriculture), and 3) cultivation of an irrigated and fertilezed kitchen garden. Boxes and arrows represent radionnuclide inventories and radionuclide fluxes, respectively. Dashed arrows represent radionuclide transfers that are not represented as dynamic fluxes in the soil system. Note that the mire ecosystem (green square) provides initial and boundary conditions for cultivation after drianage (see text for details).*

Below we describe the simple compartment model (Figure 7-2) that is used to calculate the upper boundaries for activity concentrations in agricultural soil, and the export of inorganic C-14 to the overlying crop canopy atmosphere, given an infield-outland agricultural system. Although the source of radionuclides for this land-use variant is mire vegetation in the biosphere object, no explicit location of the infield land is assigned. Instead it is simply assumed that cultivation occurs on suitable soil (Grolander 2013) at a sufficient elevation in the landscape to avoid periodic salt water intrusion.

7.1.1 Agricultural soil [RegoUp, $RegoUp_{org}$]

The radionuclide inventory in the upper layer of agricultural soil (or top soil) is represented by the compartments *RegoUp* and *$RegoUp_{org}$*. The top regolith layer represents the part of the soil that is influenced by plowing and bioturbation, and where crops primarily take up nutrients and trace elements. This layer is assumed to be well drained and have a high soil biological activity. For the radionuclides associated with organic matter the solid phase is considered only, whereas the inorganic compartment includes activity in pore water and activity adsorbed to the solid phase.

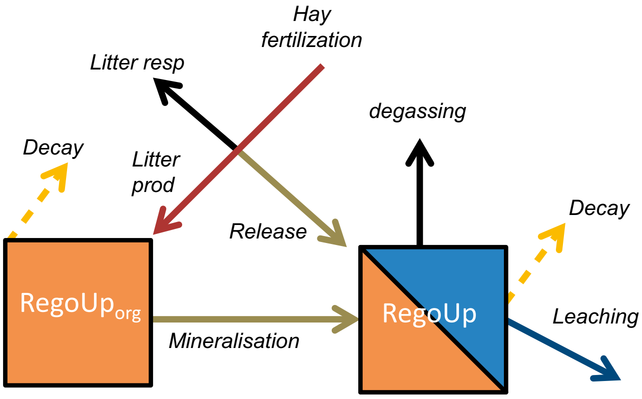


Figure 7-2. *A graphical representation of the model used to simulate transport and accumulation of radionuclides in the infield-outland agricultural ecosystems. Boxes represents radionuclide inventories and arrows fluxes of radionuclides. The right box represents radionuclides in inorganic form, and these are partitioned between adsorbed radionuclides (orange) and radionuclides in solution (blue). Radionuclides in organic form are represented by a solid state only (left box). Arrows represent radionuclide fluxes between compartments and fluxes into and out of the system. These fluxes are linked to mass fluxes of gas (black), water (dark blue) and solid matter (dark red), to transitions between inorganic and organic forms of radionuclides (brown), to radioactive decay (yellow). The atmosphere serves as a sink of radionuclides. Note that litter respiration and degassing is taken into account for carbon C-14 only.*

The radionuclide activity concentration of the agricultural soil [Bq kg_{DW}^{-1}] is derived from the radionuclide inventory in the soil compartment [Bq] and the mass of the soil layer [kg_{DW}]:

$$AC_{regoUp,IO}^{RN} = \frac{RegoUp_{IO}^{RN}\ (1 - f_{cereal})}{area_{IO} z_{regoUp,IO}\ dens_{regoUp,IO}},$$

$$AC_{regoUp,IO}^{14C} = \frac{RegoUp_{IO}^{14C}}{area_{IO} z_{regoUp,IO}\ dens_{regoUp,IO}}, \tag{7-1}$$

$$AC_{regoUp,org,IO} = \frac{RegoUp_{org,IO}}{area_{IO} z_{regoUp,IO}\ dens_{regoUp,IO}}$$

where

$RegoUp_{IO}^{RN}$ is the activity of radionuclide RN in inorganic form in cultivated soil [Bq],

$RegoUp_{IO}^{14C}$ is the total activity of inorganic C-14 in cultivated soil [Bq],

$RegoUp_{org,IO}$ is the activity of a radionuclide (including C-14) stored in soil organic matter [Bq],

$area_{IO}$ is the surface area of the cultivated field [m^2],

$z_{regoUp,IO}$ is the thickness of biologically active layer [m],

$dens_{regoUp,10}$ is the soil density [kg_{DW} m^{-3}], and

f_{cereal} is the fraction of inorganic radionuclides partitioned to cereal [Bq Bq^{-1}]:

$$f_{cereal} = \frac{biom_{cereal,IO}}{biom_{cereal,IO} + \frac{z_{regoUp,IO} dens_{regoUp,IO}}{CR_{agri,cereal}}} \tag{7-2}$$

where

$biom_{cereal,10}$ is the area-specific cereal crop biomass in carbon units [kgC m^{-2}], and

$CR_{agri,cereal}$ is the ratio between concentration in crop biomass and in soil at equilibrium [kg_{DW} kgC^{-1}].

Radionuclide pools in primary producers are expected to be small, and the turnover time of plant material short, as compared to the regolith inventories, and consequently plant uptake and biomass turnover was not represented by a dynamic flux to a separate compartment in the agricultural ecosystem models (see Section 3.1.2 Terrestrial ecosystems). However, a few non-metals (including plant nutrients) and rare elements are known to bio-accumulate, and therefore the fraction of plant available radionuclides partitioned to cereal can be used to accounted for plant immobilisation (f_{cereal}, Appendix C). Accounting for plant immobilisation also ensures that mass balance is maintained when crop concentrations are calculated with CR from soil concentrations (see Chapter 9).

The activity concentration of inorganic carbon and other radionuclides in the solute phase of agricultural soil, $AC^{D\,RN}_{regoUp,\mathrm{IO}}$ [Bq m^{-3}] is by definition the inventory in the solute phase [Bq] divided by the water-filled pore volume [m^3]:

$$
\begin{aligned}
AC^{D\,RN}_{regoUp,IO} &= \frac{RegoUp^{RN}_{IO}(1-f_{cereal})f^{D}_{regoUp}}{area_{cult}z_{regoUp,IO}poro_{regoUp,IO}S_{w,regoUp,IO}} \\
&= \frac{f^{D}_{regoUp,IO}dens_{regoUp,IO}}{poro_{regoUp,IO}S_{w,regoUp,IO}}AC^{RN}_{regoUp,IO}, \\
AC^{D\,14C}_{regoUp} &= \frac{RegoUp^{14C}_{IO}f^{D}_{regoUp,IO}}{area_{IO}z_{regoUp,IO}poro_{regoUp,IO}S_{w,regoUp,IO}} \\
&= \frac{f^{D}_{regoUp,IO}dens_{regoUp,IO}}{poro_{regoUp,IO}S_{w,regoUp,IO}}AC^{14C}_{regoUp,IO},
\end{aligned}
\qquad (7\text{-}3)
$$

where

$dens_{regoUp,IO}$	is the soil density [kg$_{\mathrm{DW}}$ m^{-3}],
$poro_{regoUp,IO}$	is the soil porosity [m^3 m^{-3}],
$S_{w,regoUp,IO}$	is the degree of saturation (i.e. the water filled fraction of the pore space) [m^3 m^{-3}], and
$f^{D}_{regoUp,ter}$	is the fraction of a radionuclide in solute phase in cultivated soil [Bq Bq^{-1}].

The fraction of a radionuclide in the solute phase in unsaturated cultivated soil [Bq Bq^{-1}] is:

$$
f^{D}_{regoUp,IO} = \frac{1}{1+\dfrac{Kd_{d,regoUp,IO}dens_{regoUp,IO}}{poro_{regoUp,IO}S_{w,regoUp,IO}}} \qquad (7\text{-}4)
$$

where

$K_{d,regoUp,IO}$	is the distribution coefficient in cultivated soil [m^3 kg$_{\mathrm{DW}}^{-1}$].

7.1.2 Fertilisation

Fertilisation is the input of radionuclides from the mire to infield soil that results from using wetland hay as animal fodder. In an infield-outland system approximately half of the fodder demand would typically be covered by grazing pastures and forest, whereas hay from meadows would cover the demand for winter fodder (Widgren 1979). The assumption that the mire in the biosphere object is the primary source for winter fodder yields an upper boundary for the input of radionuclides from mire hay to the cultivated infield soil.

Ignoring the loss of radionuclides through ingestion and handling of manure, the area-specific influx of radionuclides [Bq m^{-2} y^{-1}] is proportional to the activity concentration in mire vegetation [Bq kgC^{-1}], and the demand for organic fertiliser for a unit area of arable land [kgC m^{-2} y^{-1}]. If the mire has an area that is insufficient to cover the demand for winter fodder, then the influx has to be adjusted with the proportion of hay originating from the mire in the biosphere object. This can be expressed as:

$$
Fertilization_{hay} = AC_{hay}\,demand_{hay}\,\alpha_{hay}, \qquad (7\text{-}5)
$$

where

AC_{hay}	is the activity concentration in hay from primary producers growing in the area with the highest concentration of radionuclides [Bq kgC^{-1}].
$demand_{hay}$	is the area-specific demand for winter fodder for organic fertilisation of arable land [kgC m^{-2} y^{-1}], and
α_{hay}	is the fraction of hay originating from the mire (see below), [m^2 m^{-2}].

If the mire in the biosphere object has an area that is sufficient to cover the demand for winter fodder for the most exposed group, then all hay is assumed to originate from the biosphere object. However if the productivity of the mire is insufficient, then it is assumed that all hay is harvested and that the remaining demand for winter fodder is covered by hay not affected by radionuclides released from the repository. This potential dilution can be expresses by the following factor alpha:

$$\alpha_{hay} = \begin{cases} 1, & area_{obj,ter} \geq N_{group} area_{support} \\ \dfrac{area_{obj,ter}}{N_{group,IO} area_{support}}, & area_{obj,ter} < N_{group} area_{support} \end{cases} \qquad (7\text{-}6)$$

where

$area_{obj,ter}$	is the area of the wetland [m^2],
$N_{group,IO}$	is the number of individuals in the most exposed group of infield-outland farmers, and
$area_{support}$	is the wetland area required to support winter fodder for the livestock associated with one individual of the most exposed group [m^2].

The specific radionuclide activity in harvested wetland sedges and grass is approximated by the specific activity concentration in terrestrial primary producers (PP, Bq kgC^{-1}) (see Equation 6-1):

$$AC_{hay} = AC_{PP,ter}, \qquad (7\text{-}7)$$

7.1.3 Litter respiration (C-14) / release (other RN)

These transfers, *litter respiration* and *release*, refer to the release of radionuclides from organic fertilisers to the atmosphere (C-14) and soil (other radionuclides) as a consequence of fast decomposition (within the order of a year) of organic matter in animal manure.

Most of the wetland hay biomass that is ingested by livestock is metabolized by grazers and decomposers (respiration), but a part of the hay biomass is made up of refractory carbon which decomposes much more slowly and therefore will contribute to the build-up of soil organic matter (see litter production below). The flow of C-14 to the atmosphere associated with the fast decomposition of wetland hay [Bq y^{-1}] can be expressed as the product of the activity concentration in wetland grass and sedges [Bq kgC^{-1}], and the input of labile (non-refractory) carbon in organic fertiliser [kgC y^{-1}].

$$Litter_{resp}^{14C} = Fertilization_{hay}^{14C}\, area_{IO}\, N_{group,IO}(1 - f_{refrac,ter}) \qquad (7\text{-}8)$$

where

$Fertilization_{hay}^{14C}$	is the area specific input of C-14 to infield soil, originating from wetland hay used as animal fodder (Bq m^{-2} y^{-1}),
$area_{IO}$	is the surface area of the cultivated field supporting one individual a group of infield-outland farmers [m^2],
$N_{group,IO}$	is the number of individuals in the most exposed group of infield-outland farmers, and
$f_{refrac,ter}$	is the fraction of refractory organic matter [kgC kgC^{-1}].

When easily metabolized carbon sources are respired radionuclides other than C-14 stored in the organic matter are released to the soil. The release of radionuclides will be roughly proportional to the respired carbon. Thus, the release of radionuclides from hay to agricultural soil [Bq y^{-1}] can be expressed as the product of the input of radionuclides in organic fertilisers, the fraction of labile (non-refractory) carbon of [kgC y^{-1}] and a correction factor for enrichment or depletion of the radionuclide in the refractory organic matter:

$$Litter_{release}^{RN} = Fertilization_{hay}^{RN} area_{IO}\, N_{group,IO}(1 - f_{refrac,ter}) df_{decomp,ter} \qquad (7\text{-}9)$$

where

$df_{decomp,ter}$	is the fraction of the radionuclides that are available for release through mineralisation of labile organic matter [Bq Bq^{-1}], and the other parameters are as described in the previous equation.

7.1.4 Litter production

Litter production represents the flux of radionuclides stored in refractory organic matter in fertilisers to soil organic matter in the cultivated soil. The input rate of C-14 to soil by litter production equals the total flux of C-14 activity in the organic fertiliser [Bq y^{-1}] scaled by the fraction of refractory organic matter:

$$Litter_{Prod}^{14C} = Fertilization_{hay}^{14C}\ area_{IO}\ N_{group,IO} f_{refrac,ter} \tag{7-10}$$

where

$Fertilization_{hay}^{14C}$	is the input of C-14 to infield soil, originating from wetland hay used as animal fodder (see above),
$area_{IO}$	is the surface area of the cultivated field supporting one individual [m^2],
$N_{group,IO}$	is the number of individuals in the most exposed group, and
$f_{refrac,ter}$	is the fraction of refractory organic matter [kgC kgC^{-1}].

The flux of other radionuclides in refractory organic matter equals the total flux in organic fertilisers less radionuclides released through decomposition of the labile pool (within a year). This flux can be expressed as a function of the input of radionuclides in organic fertilisers [Bq y^{-1}], the fraction of labile (non-refractory) carbon and a correction factor for enrichment/depletion of the radionuclide in the refractory organic matter:

$$\begin{aligned} Litter_{prod}^{RN} &= Fertiliz_{hay}^{RN} area_{IO}\ N_{group} - Litter_{release}^{RN} \\ &= Fertiliz_{hay}^{RN} area_{IO}\ N_{group}\left(1 - \left(1 - f_{refrac,ter}\right) df_{decomp,ter}\right) \end{aligned} \tag{7-11}$$

where

$df_{decomp,ter}$	is the fraction of a radionuclide that is available for release through mineralisation of labile organic matter [Bq Bq^{-1}].

7.1.5 Leaching

Leaching refers to the flux of a radionuclide from cultivated soil associated with percolation of meteoric water. The flux is proportional to concentration of the radionuclide in the pore water [Bq m^{-3}] and the downward flux of water [m^3 y^{-1}]:

$$Leaching = AC_{regoUp}^{D}\ percolation_{agri}\ area_{IO} N_{group,IO}, \tag{7-12}$$

where

AC_{regoUp}^{D}	is the activity concentration of inorganic carbon or other radionuclide in the solute phase of agricultural soil [Bq m^{-3}], see Equation 7-1,
$percolation_{agri}$	is the area-specific downward flux of water from the cultivated soil [m^3 m^{-2} y^{-1}],
$area_{IO}$	is the surface area of the cultivated field supporting one individual [m^2], and
$N_{group,IO}$	is the number of individuals in the most exposed group.

Replacing the concentration AC_{regoUp}^{D} by Equation 7-3, in the above equation, leaching can be expressed as a function of the inorganic inventory *RegoUp*:

$$\begin{aligned} Leaching^{RN} &= RegoUp_{IO}^{RN} \frac{(1 - f_{cereal}) f_{regoUp}^{D}\ percolation_{agri}}{z_{regoUp,IO}\, poro_{regoUp,IO} S_{w,regoUp,IO}} = RegoUp_{IO}^{RN}\ k_{leach}^{RN}, \\ Leaching^{14C} &= RegoUp_{IO}^{14C} \frac{f_{regoUp}^{D}\ percolation_{agri}}{z_{regoUp,IO}\, poro_{regoUp,IO} S_{w,regoUp,IO}} = RegoUp_{IO}^{14C}\ k_{leach}^{14C}, \end{aligned} \tag{7-13}$$

where

$RegoUp_{IO}$	is the activity in inorganic form in cultivated soil [Bq], with respect to carbon and other radionuclides,
f_{cereal}	is the fraction of radionuclide inventory partitioned in cereal [Bq Bq^{-1}],
$z_{regoUp,IO}$	is the thickness of biologically active layer [m],
$poro_{regoUp,IO}$	is the soil porosity of agricultural soil [m^3 m^{-3}],
$S_{w,regoUp,IO}$	is the degree of saturation [m^3 m^{-3}], and
$f_{regoUp,IO}^{D}$	is the fraction of the radionuclide in the solute phase in agricultural soil [Bq Bq^{-1}].

7.1.6 Mineralisation

Mineralisation refers to the release of radionuclides from slow decomposition of refractory organic matter. Through this process, radionuclides stored in soil organic matter are released to the pore water. Mineralisation is assumed to follow first-order kinetics for homogenous substrates, and if it is assumed that trace elements are evenly distributed in the substrate, the release of all radionuclides will approximately follow that of carbon mineralisation. Thus, the release of a radionuclide through mineralisation [Bq y^{-1}] can be expressed as a fractional release rate of the organic inventory:

$$Mineralisation_{regoUp,org} = RegoUp_{org,IO}\ minRate_{IO} \tag{7-14}$$

where

$RegoUp_{org,IO}$ is the total activity in soil organic matter [Bq], and

$minRate_{IO}$ is the mineralisation rate associated with the organic matter in cultivated soil [kgC kgC^{-1} y^{-1}].

7.1.7 Degassing

Degassing refers to the flux of inorganic C-14 from soil pores to the atmosphere. Empirical studies of inorganic carbon transport in unsaturated soil suggest that this flux is primarily driven by the diffusion of gas (e.g. Sheppard et al. 1994). As the exchange of CO_2 between soil water and soil gas is very fast (Thorne 2005), degassing of CO_2 from the soil to the canopy atmosphere can be assumed to be limited by the diffusion of gas through soil pores (rather than by a phase transition). Degassing [Bq y^{-1}] is described as a function of the inventory of the inorganic C-14 in soil, the gas diffusivity of CO_2 in soil and the depth of the cultivated soil (see Appendix D):

$$Degassing^{14C} = 2\frac{RegoUp^{14C} f_{gas}\, D_{CO_2,soil,IO}}{{z_{regoUp,IO}}^2\ poro_{regoUp,IO}(1 - S_{w,regoUp,IO})} = RegoUp_{IO}^{14C} k_{degass}^{14C} \tag{7-15}$$

where

$RegoUp_{IO}^{14C}$ is the activity of inorganic C-14 in the agricultural soil [Bq],

$D_{CO_2,soil,IO}$ is the diffusivity of CO_2 in soil, based on the porosity and water content of the soil [m^2 y^{-1}],

$poro_{regoUp,IO}$ is the soil porosity [m^3 m^{-3}],

$S_{w,regoUp,IO}$ is the fraction of pore space filled with water [m^3 m^{-3}],

$z_{regoUp,IO}$ is the thickness of biologically active layer [m], and

f_{gas} is the fraction of non-solid inorganic carbon available for degassing (Appendix B):

$$f_{gas} = \frac{1 - S_{w,regoUp,IO}}{1 - S_{w,regoUp,IO}(1 - solubilityCoef_{ter})} \tag{7-16}$$

where

$solubilityCoef_{ter}$ is the solubility coefficient for CO_2 in pore water [mol m^{-3} per mol m^{-3}].

7.1.8 Activity concentration at steady state

The activity concentration in agricultural soil fertilised with manure will increase with time, asymptotically reaching a steady-state concentration, when the input of activity equals the loss through radioactive decay, leaching and degassing. At steady state, the activity balance equations for any radionuclide, including C-14, are:

$$\begin{aligned} Litter_{prod} &= Decay_{org} + Mineralization_{RegoUp,org} \\ Mineralization_{RegoUp,org} &= Decay \quad + Leaching \quad + Degassing^{14C} \end{aligned} \tag{7-17}$$

where

$Decay_c = \lambda^{RN} RegoUp_c^{RN}$ is the loss by radioactive decay [Bq y^{-1}] of radionuclide *RN* in compartment *c* with λ^{RN} being the nuclide's decay constant.

Note that degassing affects the steady-state of C-14 only. Rearranging the equations (and leaving out subscripts), the steady state inventories for organic C-14 can be written:

$$RegoUp_{org,IO}^{14C\,*} = \frac{Litter_{prod}^{14C}}{(\lambda^{14C} + minRate)} \tag{7-18}$$

This inventory sets the upper boundary for C-14 accumulation in agricultural soils resulting from fertilisation. The corresponding steady-state activity concentration is:

$$AC_{regoUp,org,IO}^{14C^*} = \frac{Fertilization_{hay}^{14C}\, f_{refrac,ter}}{(\lambda^{14C} + minRate) z_{regoUp} dens_{regoUp}} \tag{7-19}$$

Similarly, the steady state inventory of inorganic C-14 can be written:

$$RegoUp_{IO}^{14C^*} = \frac{Litter_{prod}^{14C}\, minRate}{(\lambda^{14C} + minRate)\left(\lambda^{14C} + k_{degass}^{14C} + k_{leach}^{14C}\right)} \tag{7-20}$$

And the corresponding steady state activity concentration [Bq kg_{DW}^{-1}] is:

$$AC_{regoUp,IO}^{14C^*} = \frac{Fertilization_{hay}^{14C}\, f_{refrac,ter}\, minRate}{(\lambda^{14C} + minRate)\left(\lambda^{14C} + k_{degass}^{14C} + k_{leach}^{14C}\right) z_{regoUp} dens_{regoUp}} \tag{7-21}$$

For radionuclides other than C-14 it can be deduced from the first equation that the steady state activity, which sets the upper boundary for accumulation of a radionuclide from fertilisation, can be written:

$$RegoUp_{org,IO}^{RN\,*} = \frac{Litter_{prod}^{RN}}{(\lambda^{RN} + minRate)} \tag{7-22}$$

The corresponding steady state activity concentration [Bq kg_{DW}^{-1}] is:

$$AC_{regoUp,org,IO}^{RN\,*} = \frac{Fertilization_{hay}^{RN}\left(1 - \left(1 - f_{refrac,ter}\right) df_{decomp,ter}\right)}{(\lambda^{RN} + minRate) z_{regoUp} dens_{regoUp}} \tag{7-23}$$

Similarly, the steady state inventory for the inorganic form of a radionuclide can be written:

$$RegoUp_{IO}^{RN^*} = \frac{\dfrac{Litter_{prod}^{RN}}{\lambda^{RN} + minRate} minRate + Litter_{release}^{RN}}{\lambda^{RN} + k_{leach}^{RN}} \tag{7-24}$$

The corresponding steady state activity concentration [Bq kg_{DW}^{-1}] is:

$$AC_{regoUp,IO}^{RN}{}^{*} = \frac{Fertilization_{hay}^{RN}\left(\dfrac{minRate + \left(\left(1 - f_{refrac,ter}\right) df_{decomp,ter}\right) \lambda^{RN}}{\lambda^{RN} + minRate}\right)}{\left(\lambda^{RN} + k_{leach}^{RN}\right) z_{regoUp}\, dens_{regoUp}} \tag{7-25}$$

7.2 Drainage and cultivation of a lake-mire system

In Sweden before the end of the 19th century, mires were mostly used as fields mown for hay. During the first half of the 20th century, numerous wetlands were drained, and lakes were lowered, due to a high demand for food production in combination with increasing technical possibilities for draining mires (Morell 1998). At this time, the majority of the farming population was found on small-scale farms that were self-sufficient with respect to food production, offering a plausible conceptual model for the exposed group associated with cultivation of peat and lacustrine sediments (Saetre et al. 2013).

To meet the requirements of crop cultivation, the groundwater level of wetlands and lake shores must be lowered, and this is labour-intensive. After draining, the accumulation of organic matter stops and the organic sediment layers start to subside due to oxidation, shrinkage and compaction (Lindborg 2010). Subsidence is a fast process and drained land can consequently only be used for cultivation for a limited period of time. Moreover, early cultivation in coastal environments would be associated with the risk of salt water intrusion associated with high water levels.

Sohlenius et al. (2013) studied the development and historical cultivation of a large numbers of mires in northern Uppland, i e the surroundings of Forsmark. The majority of the cultivated peatlands (96%) were in the fen stage when claimed for agricultural purposes, which is in agreement with earlier results from Uppland (Hjertstedt 1946). The time for the transition between the fen and the bog stage varies with local conditions, and fen-like areas may prevail even in the bog stage (SKB 2014e). Therefore, while it is recognized that cultivation of a wetland area after a very long accumulation time (several thousands of years) of organic matter in peat bogs is not probable in the Forsmark area, no upper age limit was used to constrain drainage and cultivation of wetland areas. This is considered a cautious model simplification.

Below, we describe the simple compartment model (Figure 7-2) that is used to calculate the activity concentrations in agricultural soil resulting from draining a mire or lowering a lake, and the export of inorganic C-14 to the overlying crop canopy atmosphere. The model accounts for processes associated with soil subsidence (compaction and oxidation) and cultivation is considered feasible only after the wetland area has emerged sufficiently above the sea level to make periodic saltwater intrusion unlikely (see threshold agriculture Chapter 9 and Grolander 2013).

7.2.1 Agricultural soil (RegoUp, RegoUp$_{org}$)

The radionuclide inventory in the upper layer of agricultural soil (or top soil) is represented by the compartments *RegoUp* and *RegoUp*$_{org}$. The top regolith layer represents the part of the soil which is influenced by plowing and bioturbation, and where crops primarily take up nutrients and trace elements. The soil is assumed to primarily originate from organic sediments (peat and postglacial sediments of aquatic origin) that have been drained through ditching. Moreover, the underlying deposits are assumed to be in contact with deep groundwater partly arising from the repository. For the radionuclides associated with organic matter the solid phase is considered, whereas the inorganic compartment includes activity in pore water and activity adsorbed to the solid phase.

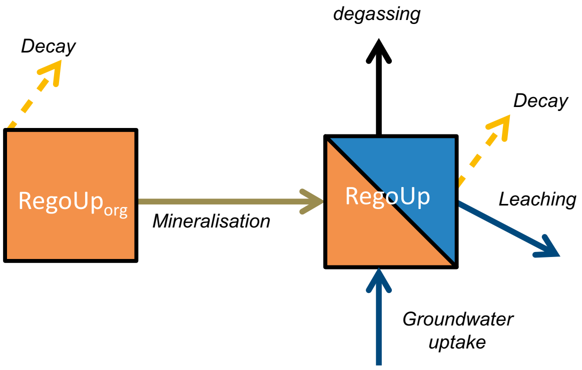


Figure 7-3. *A graphical representation of the model used to simulate transport and accumulation of radionuclides in the drained mire agricultural ecosystem. Boxes represents radionuclide inventories and arrows represent fluxes of radionuclides. The right box represents radionuclides in inorganic form, and these are partitioned between adsorbed radionuclides (orange) and radionuclides in solution (blue). Radionuclides in organic form are represented by a solid state only (left box). Arrows represent radionuclide fluxes between compartments and fluxes into and out of the system. These fluxes are linked to mass fluxes of gas (black), and water (dark blue), to transitions between inorganic and organic forms of radionuclides (brown), and to radioactive decay (yellow). The atmosphere serves as a sink for radionuclides. Note that degassing is taken into account for carbon (C-14) only.*

Successful cultivation requires that the groundwater table has to be lowered to a depth of approximately 0.5 m below the surface after soil compaction (Grolander 2013). Both peat and postglacial deposits will be compacted when drained and cultivated (Grolander 2013). Thus, the initial inventory of radionuclides in agricultural soil ($RegoUp$ and $RegoUp_{org}$, Bq) is the sum of radionuclides stored in organic matter, adsorbed to soil particles or in pore water in peat, post-glacial and glacial sediments in the horizon of the wetland that is drained for cultivation:

$$
\begin{aligned}
&RegoUp_{org,DM,t_0} = \left(RegoUp_{org,ter} + RegoPeat_{org} w_1 + RegoPG_{org,ter} w_2\right) f_{area} \\
&RegoUp_{DM,t_0} = (RegoUp_{ter} + RegoPeat\, w_1 + RegoPG_{ter} w_2 + RegoGL_{ter} w_3) f_{area},
\end{aligned}
\qquad (7\text{-}26)
$$

where

t_0	refers to the start of simulated time for which the relation holds,
$RegoUp_{ter}$ and	is the radionuclide inventory in the surface peat layer [Bq],
$RegoUp_{org,ter}$	adsorbed to soil particles, in soil pore water or stored in organic matter,
$RegoPeat$ and	is the radionuclide inventory in the deep peat layer [Bq],
$RegoPeat_{org}$	adsorbed to soil particles, in soil pore water or stored in organic matter,
$RegoPG_{ter}$ and	is the radionuclide inventory in post glacial clay-gyttja [Bq], adsorbed
$RegoPG_{org,ter}$	to soil particles, in soil pore water or stored in organic matter, respectively,
$RegoGL_{ter}$	is the radionuclide inventory in soil pore water or adsorbed to soil particles in glacial clay [Bq],
f_{area}	is the fraction of the mire area that is cultivated [m^2 m^{-2}], and
w_i	is the fraction of the regolith layer that contributes to the cultivated soil layer above the groundwater table, and is defined as:

$$
w_1 = \begin{cases} \dfrac{z_{drain,agri}}{z_{regoPeat}\, compact_{peat}}, & if\ z_{regoPeat} compact_{peat} > z_{drain,agri} \\ 1, & otherwise \end{cases}
$$

$$
w_2 = \begin{cases} 0, & if\ z_{regoPeat} compact_{peat} \geq z_{drain,agri} \\ 1, & if\ z_{drain,agri} - z_{regoPeat}\, compact_{peat} \geq z_{regoPG,ter} compact_{gyttja} \\ \dfrac{z_{drain,agri} - z_{regoPeat}\, compact_{peat}}{z_{regoPG,ter} compact_{gyttja}}, & otherwise \end{cases} \qquad (7\text{-}27)
$$

$$
w_3 = \begin{cases} 0, & if\ z_{regoPeat} compact_{peat} + z_{regoPG,ter} compact_{gyttja} \geq z_{drain,agri} \\ 1, & if\ z_{drain,agri} - (z_{regoPeat} compact_{peat} + z_{regoPG,ter} compact_{gyttja}) \geq z_{regoGL,ter} \\ \dfrac{z_{drain,agri} - compact_{peat} z_{regoPeat} - compact_{gyttja} z_{regoPG,ter}}{z_{regoGL,ter}}, & otherwise \end{cases}
$$

where

$compact_i$	is subsidence of drained and cultivated peat and clay-gyttja, due to soil compaction and oxidation of organic matter [m m^{-1}]
$z_{rego\_j,ter}$	is the original thickness of mire regolith layers [m].
$z_{drain,agri}$	is the depth of the drained and cultivated soil layer after subsidence [m].

Note that if the peat layer is sufficiently thick (after soil compaction), then deeper sediments will not contribute to the initial inventory in the drained agricultural soil. Similarly, glacial deposits will only contribute to agricultural soil when the depth of the compacted organic soil layers does not exceed the required soil depth.

The fraction of radionuclides in mire regolith layers that ends up in cultivated soil is proportional to the fractional area of the drained part of the mire:

$$
f_{area} = \begin{cases} \dfrac{support\_area_{DM}\, N_{group}}{area_{obj,ter}}, & if\ area_{obj,ter} > support\_area_{DM}\, N_{group} \\ 1, & otherwise \end{cases} \qquad (7\text{-}28)
$$

where

$area_{obj,ter}$ is the surface area of the wetland/mire [m^2],

$support\_area_{DM}$ is the surface area of arable land supporting one individual [m^2], and

$N_{group,DM}$ is the number of individuals in the most exposed group.

The total radionuclide activity concentration of the agricultural soil [Bq kg$_{DW}^{-1}$] is by definition the radionuclide inventory in the soil compartment [Bq] divided by the mass of the soil layer [kg$_{DW}$]:

$$AC_{regoUp,org,DM} = \frac{RegoUp_{org,DM}}{area_{cult} z_{drain,agri} dens_{regoUp,DM}},$$

$$AC_{regoUp,DM}^{14C} = \frac{RegoUp_{DM}^{14C}}{area_{cult} z_{drain,agri} dens_{regoUp,DM}}, \tag{7-29}$$

$$AC_{regoUp,\mathrm{DM},i}^{RN} = \frac{RegoUp_{DM,i}^{RN}(1-f_i)}{area_{cult} z_{drain,agri} dens_{regoUp,DM}}, \qquad i\ = \{fodder, tuber, cereal\},$$

where

$RegoUp_{DM,i}^{RN}$ is the initial activity of other radionuclides in the soil plant system, and is used synonymously with $RegoUp_{DM}^{RN}$ [Bq],

$area_{cult}$ is the surface area of cultivated land (i.e. $support\_area_{DM}$ times N_{group}) [m^2],

$z_{drain,agri}$ is the depth of the unsaturated agricultural soil layer (after compaction) [m],

$dens_{regoUp,DM}$ is the density of cultivated organic soil (see Equation 7-34 below) [kg$_{DW}$ m^{-3}], and

f_i is the fraction of the soil radionuclide inventory that is immobilised in crop [Bq Bq^{-1}]:

$$f_i = \frac{biom_{DM,i},}{biom_{DM,i} + \frac{z_{drain,agri} dens_{regoUp,DM}}{CR_{agri,i}}}, \tag{7-30}$$

where

i is the index for fodder, potatoes and cereal,

$biom_{DMi}$ is the area-specific crop biomass in carbon units [kgC m^{-2}], and

$CR_{agri,i}$ is the ratio between concentration in crop biomass and soil at equilibrium [kg$_{DW}$ kgC^{-1}].

The activity concentration of inorganic carbon and other radionuclides in the solute phase of agricultural soil is by definition the inventory in the solute phase [Bq] divided by the water filled pore volume [m^3]:

$$AC_{regoUp,DM}^{D,14C} = \frac{RegoUp_{DM}^{14C} RegoUp^{14C} f_{regoUp,DM}^{D}}{area_{cult} z_{drain,agri} poro_{regoUp,DM} S_{w,regoUp,DM}},$$

$$AC_{regoUp,\mathrm{DM},i}^{D,RN} = \frac{RegoUp_{DM,i}^{RN}(1-f_i) f_{regoUp,DM}^{D}}{area_{cult} z_{drain,agri} poro_{regoUp,DM} S_{w,regoUp,DM}}, \qquad i\ = \{fodder, potatoes, cereal\}, \tag{7-31}$$

where

$poro_{regoUp,DM}$ is the porosity of cultivated organic soil (Equation 7-33) [m^3 m^{-3}],

$S_{w,regoUp,DM}$ is the degree of saturation, and

$f_{regoUp,DM}^{D}$ is the fraction of a radionuclide in the solute phase [Bq Bq^{-1}].

The fraction of a radionuclide in the solute phase in cultivated organic soil [Bq Bq^{-1}] is:

$$f_{regoUp,DM}^{D} = \frac{1}{1 + \frac{K_{d,regoUp,DM} dens_{regoUp,DM}}{poro_{regoUp,DM} S_{w,regoUp,DM}}} \tag{7-32}$$

where

$K_{d,regoUp,DM}$ is the distribution coefficient in cultivated organic soil [m^3 kg$_{DW}^{-1}$].

Soil properties (e.g. density and porosity) of organic agricultural soils vary with the composition of the parent material. For example the porosity is higher and the density is lower in agricultural soil originating from pure peat than in soil originating from clay gyttja. Considering two types of parent material (peat or clay-gyttja), the soil porosity and the soil density of the drained and cultivated mire can be approximated as a weighted average of properties resulting from either of the two types of parent material:

$$poro_{regoUp,DM} = w_4 poro_{regoUp,peat} + (1 - w_4) poro_{regoUp,clay}, \qquad (7\text{-}33)$$

$$dens_{regoUp,DM} = w_4 dens_{regoUp,peat} + (1 - w_4) dens_{regoUp,clay} \qquad (7\text{-}34)$$

where

$$w_4 = \begin{cases} 1, & if\ z_{regoPeat} compact_{peat} \geq z_{drain,agri} \\ \dfrac{z_{regoPeat} compact_{peat}}{z_{drain,agri}}, & otherwise \end{cases} \qquad (7\text{-}35)$$

7.2.2 Groundwater uptake

The unsaturated biological active layer of agricultural soil is in contact with the saturated soil below. Ditching or lake lowering is assumed to have a limited effect on the groundwater in these soil layers, and the water chemistry is assumed to be relatively undisturbed in the saturated zone (i.e. similar to that in the undisturbed mire ecosystem).

Groundwater uptake refers to the upward transport of radionuclides [Bq y^{-1}] from the saturated to the unsaturated soil layer. This transfer is proportional to the upward flux of water [$m^3\ y^{-1}$] and the concentration of a radionuclide in the solute phase of the deep groundwater water [Bq m^{-3}]:

$$GW_{uptake}^{14C} = Flux_{water,satSoil,agri}\, area_{DM} N_{group,DM} AC_{Soil,Saturated}^{D,14C},$$

$$GW_{uptake,i}^{RN} = Flux_{water,satSoil,agri}\, area_{DM} N_{group,DM} f_{area,i} AC_{Soil,Saturated}^{D,RN}, \qquad (7\text{-}36)$$

$$AC_{Soil,Saturated}^{D} = \begin{cases} AC_{regoGL,ter}^{D}, & w_3 < 1 \\ AC_{regoLow,ter}^{D}, & w_3 = 1' \end{cases}$$

where

$Flux_{wat,satSoil,agri}$	is the water flux from the saturated to the unsaturated zone [$m^3\ m^{-2}\ y^{-1}$],
$area_{DM}$	is the surface area of the cultivated wetland per individual [m^2],
$N_{group,DM}$	is the number of individuals in the most exposed group [–],
$f_{area,i}$	is the fraction of area used for fodder, potato and cereal production [–],
$AC^{D}_{soil,Satur}$	is the activity concentration in the solute phase in the water-saturated layer below the trenching depth [Bq m^{-3}],
$AC^{D}_{regoGL,ter}$	is the activity concentration in the pore water of the terrestrial glacial regolith layer [Bq m^3] (Equation 6-7),
$AC^{D}_{regoLow,ter}$	is the activity concentration in the pore water of the terrestrial lower regolith layer [Bq m^3] (Equation 6-7),
w_3	is the fraction of the glacial regolith incorporated in the cultivated soil (Equation 7-27).
i	is the index for the crops: fodder, potato and cereal.

7.2.3 Leaching

Leaching refers to the flux of a radionuclide from cultivated soil associated with percolation of meteoric water. The flux [Bq y^{-1}] is proportional to concentration of the radionuclide in the pore water and the downward flux of water (percolation), and can be calculated with Equation 7-12 (Infield-outland agriculture), using parameters for the drained and cultivated mire and the three associated crops (fodder, potato and cereal).

Inserting Equations (7-30) into 7-12, leaching can be expressed as a function of the inorganic radionuclide inventory, the fraction of the inventory partitioned to the solute phase, soil physical properties, and the area specific percolation for each of the three crops:

$$Leaching^{14C}_{regoUp,DM,i} = \frac{RegoUp^{14C}_{DM,i} f^{D}_{regoUp,DM,i} percolation_{agri}}{z_{drain,agri}\, poro_{regoUp,DM} S_{w,regoUp,DM}} = RegoUp^{14C}_{DM,i} k^{14C}_{leach,i},$$

$$Leaching^{RN}_{regoUp,DM,i} = \frac{RegoUp^{RN}_{DM,i}(1-f_i) f^{D}_{regoUp,DM} percolation_{agri}}{z_{drain,agri}\, poro_{regoUp,DM} S_{w,regoUp,DM}} = RegoUp^{RN}_{DM,i} k^{RN}_{leach,i}, \tag{7-37}$$

where

i	is the index standing for fodder, potato or cereal,
$percolation_{agri}$	is the area specific downward flux of water (runoff) from cultivated soil [$m^3\ m^{-2}\ y^{-1}$],
$RegoUp_{DM,i}$	is the activity of radionuclides in inorganic form in cultivated soil [Bq],
$z_{drain,agri}$	is the trenching depth required for drainage (after compaction) [m],
$poro_{regoUp,DM}$	is the porosity of cultivated organic soil [$m^3\ m^{-3}$] (see Equation 7-33),
$S_{w,regoUp,DM}$	is the degree of saturation,
$f^{D}_{regoUp,DM,i}$	is the fraction of the radionuclide in solute phase [$Bq\ Bq^{-1}$] (see Equation 7-32),
f_i	is the fraction of radionuclide inventory partitioned to the crop [$Bq\ Bq^{-1}$] (see Equation 7-30).

7.2.4 Mineralisation

Mineralisation refers to the release of radionuclides from slow decomposition of refractory organic matter in peat and clay-gyttja. Through this process, radionuclides stored in soil organic matter are released to the soil pore water. The release of radionuclides [$Bq\ y^{-1}$] can be expressed as a fractional release rate of the organic inventory:

$$Mineralisation_{regoUp,org} = RegoUp_{org,DM}\ minRate_{DM} \tag{7-38}$$

where

$RegoUp_{org,DM}$	is the total radionuclide activity in soil organic matter,
$minRate_{DM}$	is the mineralisation rate associated with peat and post-glacial deposits when drained and cultivated [$kgC\ kgC^{-1}\ y^{-1}$].

7.2.5 Degassing

Degassing refers to the flux of inorganic C-14 from soil pores to the atmosphere. Degassing [$Bq\ y^{-1}$] is described as a function of the inventory of the inorganic C-14 in soil, the gas diffusivity of CO_2 in soil and the depth of the cultivated soil:

$$Degassing^{14C} = 2\frac{RegoUp^{14C}_{DM} f_{gas}\, D_{CO_2,soil,DM}}{{z_{drain,agri}}^2\, poro_{regoUp,DM}(1-S_{w,regoUp,DM})} = RegoUp^{14C}_{DM}\, k^{14C}_{degass} \tag{7-39}$$

where

$RegoUp^{14C}_{DM}$	is the activity of inorganic C-14 in the agricultural soil [Bq],
$D_{CO_2,soil,DM}$	is the diffusivity of CO_2 in soil,
$poro_{regoUp,DM}$	is the soil porosity (Equation 7-33)[$m^3\ m^{-3}$],
$S_{w,regoUp,DM}$	is the fraction of pore space filled with water [$m^3\ m^{-3}$],
$z_{regoUp,DM}$	is the trenching depth required for drainage (after compaction) [m], and
f_{gas}	is the fraction of non-solid inorganic carbon available for degassing (Equation 7-16).

7.2.6 Average activity concentration over the first 50 years after draining

Sustainable use of organic soils, which originates from drained wetlands, is expected to be productive for agriculture for a limited time only. This is partly because peat subsides when groundwater is lowered, and the thickness of the organic layer may be reduced by as much as 60% by compaction and oxidation within 100 years (Berglund 2008). In many areas, the deposits underlying the peat may not be suitable for cultivation and it may also be difficult to maintain drainage when the peat has subsided (Lindborg 2010). However, from the perspective of the safety assessment, 50 to 100 years is a relevant assessment period for the life-time risk.

In mathematical terms, the dynamic behaviour of the radionuclide inventory in agricultural soil after drainage can be described by the following system of ordinary differential equations given:

$$\frac{dRegoUp_{org,DM}}{dt} = -Decay_{regoUp,org} - Miner_{regoUp,org},$$

$$\frac{dRegoUp_{DM}^{14C}}{dt} = Miner_{regoUp,org}^{14C} + GW_{uptake}^{14C} - Decay_{regoUp}^{14C} - Leaching_{regoUp}^{14C} - Degassing_{regoUp}^{14C}, \qquad (7\text{-}40)$$

$$\frac{dRegoUp_{DM,i}^{RN}}{dt} = Miner_{regoUp,org}^{RN} + GW_{uptake,i}^{RN} - Decay_{regoUp,i}^{RN} - Leaching_{regoUp,i}^{RN},$$

where

i	is the index for the crops: fodder, potato and cereal,
$RegoUp_{org,DM}$	is the inventory of a radionuclide, including C-14, stored in organic matter in cultivated soil [Bq],
$RegoUp_{DM}^{14C}$	is the inventory of inorganic C-14 in cultivated soil [Bq],
$RegoUp_{DM,i}^{RN}$	is the inventory of a radionuclide in inorganic form in cultivated soil of crop *i* [Bq],

and the following notation for radionuclide fluxes applies:

$Decay_{regoUp,org} = \lambda\, RegoUp_{org,DM}$	*Decay* denotes losses of radionuclides
$Decay_{regoUp} = \lambda\, RegoUp_{DM}$	[Bq y^{-1}] due to radioactive decay.
$Miner_{regoUp,org} = minRate\ RegoUp_{org,DM}$	*Miner* denotes flux [Bq y^{-1}] from an organic inventory to an inorganic inventory due to mineralisation.
GW_{uptake}	*GW* denotes the import of radionuclides in inorganic form with deep groundwater [Bq y^{-1}], assumed to be constant.
$Leaching_{regoUp,i} = k_{i,leach}\, RegoUp_{DM,i}$	*Leaching* denotes losses of radionuclides [Bq y^{-1}] due to percolation of groundwater
$Degassing_{regoUp}^{14C} = k_{degass}^{14C} RegoUp^{14C}$	*Degassing* denotes the loss of inorganic C-14 [Bq y^{-1}] due to degassing.

Taking the initial conditions into account (Equation 7-26), the solution of the system of ordinary differential Equations (7-40) for the radionuclide inventories $RegoUp_{org,DM}$, $RegoUp_{DM}^{14C}$ and $RegoUp_{DM,i}^{RN}$ [Bq] is as follows:

$$\begin{aligned}
&RegoUp_{org,DM}(t) = RegoUp_{org,\mathrm{DM},t_0} e^{-(\lambda+minRate)t}\\
&RegoUp_{DM}^{14C}(t) = RegoUp_{DM,t_0}^{14C} e^{-\left(\lambda^{14C}+k_{leach}^{14C}+k_{degass}^{14C}\right)t}\\
&\qquad + GW_{uptake}^{14C} \frac{1-e^{-\left(\lambda^{14C}+k_{leach}^{14C}+k_{degass}^{14C}\right)t}}{\lambda^{14C}+k_{leach}^{14C}+k_{degass}^{14C}}\\
&\qquad + RegoUp_{org,DM,t_0}^{14C} minRate \frac{e^{-\left(\lambda^{14C}+minRate\right)t}-e^{-\left(\lambda^{14C}+k_{leach}^{14C}+k_{degass}^{14C}\right)t}}{k_{leach}^{14C}+k_{degass}^{14C}-minRate}\\
&RegoUp_{DM,i}^{RN}(t) = RegoUp_{DM,t_0}^{RN} e^{-\left(\lambda^{RN}+k_{i,leach}^{RN}\right)t}\\
&\qquad + GW_{uptake}^{RN} \frac{1-e^{-\left(\lambda^{RN}+k_{i,leach}^{RN}\right)t}}{\lambda^{RN}+k_{i,leach}^{RN}}\\
&\qquad + RegoUp_{org,DM,t_0}^{RN} minRate \frac{e^{-\left(\lambda^{RN}+minRate\right)t}-e^{-\left(\lambda^{RN}+k_{i,leach}^{RN}\right)t}}{k_{i,leach}^{RN}-minRate}\\
&\qquad i = \{fodder, tuber, cereal\}
\end{aligned} \tag{7-41}$$

When wetland peat is drained and cultivated, radionuclides which have accumulated over a long period will be released, and the contribution from the initial inventory is expected to far exceed the slow accumulation caused by continuing groundwater uptake. As the inherited soil inventory of organic and inorganic radionuclides is expected to decrease with time (Equation 7-41), so is the concentration and the exposure of the most exposed group. As risk is assessed over an adult life-time (50 years) the average annual dose from consuming food or spending time on working on arable land will be proportional to the average soil concentration during the first 50 years after drainage. The average soil activity in the organic component of the agricultural soils over this period can be described by:

$$RegoUp_{org,aver,DM} = \frac{\int_{t_0}^{t_{50}} RegoUp_{org}(t)dt}{t_{50}-t_0} = RegoUp_{org,t_0} f_{regoUp,org,aver} \tag{7-42}$$

where

$$f_{regoUp,org,aver} = \frac{1-e^{-(\lambda+minRate)(t_{50}-t_0)}}{(\lambda+minRate)(t_{50}-t_0)}$$

Similarly, average activities for the inorganic component of agricultural soils for the first 50 years are derived by:

$$\begin{aligned}
&RegoUp_{aver,DM}^{14C} = \frac{\int_{t_0}^{t_{50}} RegoUp_{DM}^{14C}(t)dt}{t_{50}-t_0}\\
&RegoUp_{i,aver,DM}^{RN} = \frac{\int_{t_0}^{t_{50}} RegoUp_{DM,i}^{RN}(t)dt}{t_{50}-t_0}, \qquad i = \{fodder, potatoes, cereal\}
\end{aligned} \tag{7-43}$$

By ignoring the differences in indices for C-14 and other radionuclides, the analytical solution to Equation (7-43) is given by:

$$\begin{aligned} RegoUp_{aver,DM} &= \frac{\int_{t_0}^{t_{50}} RegoUp_{DM}(t)dt}{t_{50} - t_0} \\ &= RegoUp_{DM,t_0} f_{regoUp,aver} + \frac{GW_{uptake}}{\lambda + k_{degass} + k_{leach}} \left(1 - f_{regoUp,aver}\right) \\ &+ \frac{RegoUp_{org,\mathrm{DM},t_0} minRate}{k_{degass} + k_{leach} - minRate} \left(f_{regoUp,org,aver} - f_{regoUp,aver}\right) \end{aligned} \qquad (7\text{-}44)$$

where

$$f_{regoUp,aver} = \frac{1 - e^{-(\lambda + k_{degass} + k_{leach})(t_{50} - t_0)}}{(\lambda + k_{degass} + k_{leach})(t_{50} - t_0)}$$

Finally activity concentration for the initial period of cultivation is calculated by dividing the average soil activity by the soil mass for the period of interest:

$$AC_{regoUp,aver,DM}^{14C} = \frac{RegoUp_{aver,DM}^{14C}}{area_{DM} N_{group,DM} z_{drain,agri} dens_{regoUp,DM}},$$

$$AC_{regoUp,i,aver,DM}^{RN} = \frac{RegoUp_{i,aver,DM}^{RN}}{area_{DM} f_{area,i} N_{group,DM} z_{drain,agri} dens_{regoUp,DM}}, \qquad i = \{fodder, tuber, cereal\}, \quad (7\text{-}45)$$

$$AC_{regoUp,org,aver,DM} = \frac{RegoUp_{org,aver,DM}}{area_{DM} N_{group,DM} z_{drain,agri} dens_{regoUp,DM}},$$

where

$RegoUp_{i,aver,DM}^{RN}$	is the average inventory for crop i, which implies that all right hand terms of Equation 7-44 is scaled by the fraction of area used for each crop respectively ($f_{area,i}$),
$area_{DM}$	is the surface area of the cultivated wetland per individual [m^2],
$N_{group,DM}$	is the number of individuals in the most exposed group [–],
$f_{area,i}$	is the fraction of area used for fodder, potato and cereal production [–],
$dens_{regoUp}$	is the density of cultivated organic soil [kg$_{\mathrm{DW}}$ m^{-3}], and
$z_{drain,agri}$	is the trenching depth required for drainage (after compaction) [m].

Since the average inventories (Equations 7-43–7-45 and equations referred to within) and the soil volume are both proportional to the cultivated surface area, the average activity concentrations in the cultivated soil is not dependent of the surface area of the cultivated land, as long as the drained area is large enough to support the most exposed group. However, the area of a crop field affects the concentration of C-14 in the canopy atmosphere (see Chapter 8).

7.3 Garden plot cultivation

There are several possible pathways of exposure that are likely to affect a small group of future permanent residents, who are not necessarily self-sustained with respect to food production. These exposure pathways include irrigation, biomass fuel burning and soil improvement through fertilisation with seaweed or ash.

Large scale irrigation of cereals is unlikely in the Forsmark region given the present and predicted future climate, but present-day horticultural practice involves irrigation of potatoes and vegetables (SKB 2014e). Thus, for the safety assessment, a kitchen garden appears to be an appropriate basis for assessment of exposure from irrigation with well or surface water.

Moreover, biomass and peat are natural resources that may be used to produce heat and energy. However, the area of the land possibly affected by radionuclide release is limited, and in a lifetime (50-year) perspective, the biomass resources would be sufficient to heat a few households only. Likewise, the ash from heat production on this scale would be sufficient to fertilise less than a hectare of land (SKB 2014e). Therefore, a household appears to be reasonable unit to assess exposure from inhalation of combustion gases in the safety assessment, and a kitchen garden an appropriate unit for assessment of exposure from the use of ash as fertiliser.

Marine algae (i.e. seaweeds) have been a valuable source for fertilisers and mulch in Baltic coastal communities. At the late succession stage of a sea basin, when the water exchange is limited and activity concentrations are expected to peak, the production of algal biomass is limited, and only a fraction of this resource can be expected to be washed ashore locally (SKB 2014e). Consequently, it seems reasonable to assume that exposure from radionuclides in seaweeds will occur at a local household-scale, where this resource is used as fertiliser and mulch for horticultural production.

Below we describe a simple compartment model used to calculate concentrations in cultivated soil, and the export of inorganic C-14 to the overlying canopy atmosphere. The model of a garden plot is structurally similar to that of the infield-outland agricultural system (Section 7.1), and for the assessment of algal fertilisation the models are identical. However in the garden plot model radionuclides in irrigation water and ashes are represented as additional source terms to the inorganic radionuclide inventory (Figure 7-4). Moreover, the processes of radionuclide retention in water on leaf structures, and subsequent volatilization (of $^{14}CO_2$), have been added to the garden plot model.

Given the small scale of the land resource used and the limited lifetime of a dug or drilled well, a period of 50 years of continual use (i.e. a human adult life-time) has been considered a reasonable scale for the assessment of radionuclide accumulation in this land-use variant. The source of radionuclides originates from the use of natural resources in the biosphere object. However, the location of the garden plot is not assigned explicitly. Instead it is simply assumed that cultivation occur on suitable soil located somewhere outside the biosphere object.

7.3.1 Cultivated soil [RegoUp, $RegoUp_{org}$]

The radionuclide inventory in the upper layer of cultivated soil (or top soil) is represented by the compartments *RegoUp* and *$RegoUp_{org}$*. The top regolith layer represents the part of the soil which is influenced by digging and bioturbation, and where crops primarily take up nutrients and trace elements. This layer is assumed to be well drained and have a high soil biological activity. For the radionuclides associated with organic matter the solid phase is considered only, whereas the inorganic compartment includes activity in pore water and activity adsorbed to the solid phase.

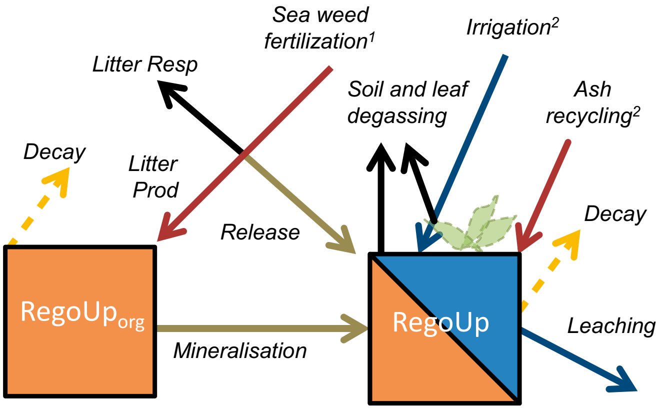


***Figure 7-4.** A graphical representation of the model used to simulate transport and accumulation of radionuclides associated with cultivation of a garden plot. Boxes represents radionuclide inventories and arrows fluxes of radionuclides. The right box represents radionuclides in inorganic form, and these are partitioned between adsorbed radionuclides (orange) and radionuclides in solution (blue). Radionuclides in organic form are represented by a solid state only (left box). Arrows represents radionuclide fluxes between compartments and fluxes into and out of the system. These fluxes are linked to mass fluxes of gas (black), water (dark blue) and solid matter (dark red), to transitions between inorganic and organic forms of radionuclides (brown), to radioactive decay (yellow). The atmosphere serves as a sink of radionuclides. Note that fertilisation with algal biomass (1) is considered during the submerged period, whereas ash-recycling and irrigation (2) are considered only when the biosphere object has emerged out of the sea. Litter respiration and soil and leaf degassing are taken into account for carbon (C-14) only.*

The radionuclide activity concentration in cultivated soil, $AC_{regoUp,GP}$ and $AC_{regoUp,org,GP}$ [Bq kg_{DW}^{-1}], and the activity concentration in the solute phase, $AC^{D}_{regoUp,GP}$ [Bq m^{-3}], are calculated with Equations 7-1 and 7-3, respectively, using parameters for the garden plot and the two associated crops (potatoes and vegetables).

7.3.2 Fertilisation

Fertilisation results in an entry of radionuclides in seaweed from the sea ecosystem, or of radionuclides in deep peat or biomass ash from the mire ecosystem to the cultivated soil. Radionuclides in algal biomass are initially in organic form, but through fast decomposition a major part will contribute to the inorganic soil inventory or the atmosphere (C-14) (see fate of organic fertilisers in Infield-outland agriculture). Radionuclides left in ash after combustion are in inorganic form, and will contribute only to the inorganic soil inventory.

Fertilisation with seaweed is considered a relevant exposure pathway when the biosphere object is submerged (time < threshold_isolation), whereas fertilisation with ash from biomass combustion can occur during the land period only, either directly after the biosphere objected has been isolated (plant biomass, time > threshold isolation) or when peat has accumulated and can be harvested (time > threshold_stop) (Grolander 2013).

Fertilisation with sea macrophytes

To get an upper boundary on the input of radionuclides from seaweed, it is assumed that aquatic macrophytes from the biosphere object are used as the primary source of fertilisation of the garden plot during the submerged period. Ignoring loss of radionuclides associated with handling, the area-specific influx of radionuclides, $Fertilisation_{algae}$ [Bq y^{-1} m^{-2}], is proportional to the activity concentration in the fertiliser and the demand for fertiliser on a unit area of arable land.

$$Fertilisation_{algae} = AC_{biom,macro}\, demand_{AlgFertil}\, \alpha_{AlgFertil} \qquad (7\text{-}46)$$

where

$AC_{biom,macro}$ is the activity concentration in algal fertilisers (Equation 5-8) [Bq kgC^{-1}].

$demand_{AlgFertil}$ is the fertiliser demand per square metre of garden plot [kg_C m^{-2} y^{-1}],

$\alpha_{AlgFertil}$ is the fraction of fertiliser originating from the biosphere object (see below) [kgC y^{-1} per kgC y^{-1}].

If the production of macrophyte biomass in the biosphere object is insufficient to cover the demand for fertilisation, it is assumed that the organic fertiliser will be diluted with seaweed from areas outside the biosphere object. This may result in a potential dilution at late stages of the submerged stage according to the dilution factor alpha:

$$\alpha_{AlgFertil} = \frac{supply_{algFertil}}{demand_{AlgFertil}\, area_{GP}\, N_{group,GP}} \qquad (7\text{-}47)$$

where

$N_{group,GP}$ is the number of individuals in the garden plot household,

$area_{GP}$ is the size of the garden plot per individual in the household [m^2], and

$supply_{algFertil}$ is the supply of algal fertiliser (see below) [kgC y^{-1}]:

$$supply_{algFertil} = NPP_{macro}\, area_{obj,aqu} \qquad (7\text{-}48)$$

where

NPP_{macro} is the area specific net primary production of aquatic macrophytes [kgC m^{-2} y^{-2}],

$area_{obj,aqu}$ is the surface area of the aquatic part of the biosphere object [m^2].

Through fast (within-year) decomposition, radionuclides in organic fertilisers are released to the atmosphere (*Repiration*, C-14) or become available in inorganic form (*Release*), whereas radionuclides in refractory organic matter are added to the organic compartment of cultivated soil (*Litter production*). The partitioning of C-14 to the atmosphere is proportional to the fraction of easily decomposable matter, whereas the flux of other radionuclides to the inorganic soil compartment is further modified by the elements availability for release through initial mineralisation (see Section 7.1.3 for details).

Recycling of ashes from house-hold heating

In this land-use variant, we consider that a household is heated using firewood or peat collected in the biosphere objects. The amount of ash produced from heating a household is relatively small; namely 50 and 170 kg_{DW} ash per year for combustion of fire wood and peat respectively (SKB 2014e). Thus, it is cautiously assumed that all bottom ash is recycled to the household garden plot as a fertiliser supplement. The area specific input of radionuclides is then [Bq $y^{-1}m^{-2}$]:

$$Fertilization_{ash,type} = \frac{AC_{fuel,type}(1 - f_{combust})fuel_{cons,type}}{area_{GP}\ N_{group,GP}} \qquad (7\text{-}49)$$
$$type = \{peat, wood\}$$

where

AC_{fuel}	is the activity concentration in peat and wood used as biomass fuel [Bq kg_{DW}^{-1}],
$f_{combust}$	is fraction of the fuel inventory that ends up in fly ash and gas after combustion of wood or peat [kg_{DW} kg_{DW}^{-1}].
$fuel_{cons}$	is the consumption of biomass fuel needed to heat a household (~20,000 kWh) by combustion of peat or firewood [kg_{DW} y^{-1}],
$area_{GP}$	is the area of the garden plot, expressed per individual [m^{-2}], and
$N_{group,GP}$	is the number of individuals in the garden plot household,

The activity concentrations in the fuel [Bq kg_{DW}^{-1}] can be approximated by those in natural resources (vegetation or peat) from the mire ecosystem, scaled to dry weight:

$$AC_{fuel,wood} = AC_{PP,ter} f_{C,pp.ter} \alpha_{fuel,wood}$$
$$AC_{fuel,peat} = AC_{regoPeat,tot} \alpha_{fuel,peat} \qquad (7\text{-}50)$$

where

$AC_{PP,ter}$	is the activity concentration in mire vegetation[Bq kgC^{-1}]
$f_{C,pp.ter}$	is the amount of carbon in unit dry weight of mire vegetation [kgC kg_{DW}^{-1}]
α_{fuel}	is a potential dilution factor due to limitations of fuel resources in the biosphere object (see below) [unitless], and
$AC_{regoPeat,tot}$	is the total activity concentration in peat, including contributions from the radionuclide in organic and inorganic forms [Bq kg_{DW}^{-1}]:

$$AC_{regoPeat,tot} = AC_{regoPeat,ter} + AC_{regoPeat,org,ter} \qquad (7\text{-}51)$$

where

$AC_{regoPeat,ter}$	is the radionuclide concentration in inorganic form [Bq kgC^{-1}], and
$AC_{regoPeat,org,ter}$	is the concentration in organic form in deep mire peat [Bq kgC^{-1}].

If the yearly production of biomass, or the accumulated peat, in the biosphere object is insufficient to cover the demand of a household for biomass fuel, then it is assumed that the fuel will be diluted with biomass from areas outside the biosphere object. This will result in a potential dilution at early stages of the land period of a biosphere object:

$$\alpha_{ash,wood} = \begin{cases} 1, & area_{obj,ter} \geq N_{group} area_{support} \\ \dfrac{area_{obj,ter}}{N_{group} area_{support,wood}}, & area_{obj,ter} < N_{group} area_{support,wood} \end{cases}$$
$$\alpha_{ash,peat} = \begin{cases} 1, & supply_{peat} \geq N_{group}\ demand_{peat} \\ \dfrac{supply_{peat}}{N_{group}\ demand_{peat}}, & supply_{peat} < N_{group}\ demand_{peat} \end{cases} \qquad (7\text{-}52)$$

where

$area_{obj,ter}$	is the wetland area of the biosphere object [m^2],
$area_{support,wood}$	is the area supporting the demand of a household for firewood for heating (~20,000 kWh y^{-1}), expressed per individual [m^2],
$demand_{peat}$	is the mass of peat needed to heat a household over a 50-year period (~20,000 kWh y^{-1}), expressed per individual [kg_{DW}], and
$Supply_{peat}$	is the mass of peat that has accumulated in the biosphere object [kg_{DW}]:

$$Supply_{peat} = area_{obj,ter}\, z_{regoPeat} dens_{regoPeat} \qquad (7\text{-}53)$$

where

$z_{regoPeat}$	is the average depth of accumulated peat [m], and
$dens_{regoPeat}$	is the density of accumulated peat [$kg_{DW}\ m^{-3}$].

7.3.3 Irrigation

Irrigation refers to the flux of radionuclides in the irrigation water infiltrating into cultivated soil. Some of the radionuclides in the irrigation water will be intercepted by vegetation (see leaf retention below). However, the fraction of radionuclide that can be removed by sorption (and other processes) after interception is small (~ 7% with typical parameter values used in this assessment[5]), and intercepted radionuclides will be washed down to the ground with time. For simplicity the loss of intercepted radionuclides are not explicitly included in the activity balance model. Instead the intercepted radionuclides contribute both to the activity concentration on intercepted leaves and to the input of radionuclides to the cultivated soil.

The influx of radionuclides to the cultivated soil [Bq y^{-1}] with irrigation water equals the product of the annual volume of irrigation water [$m^3\ y^{-1}$] and the radionuclide concentration of the irrigation water [Bq m^{-3}]:

$$Irrigation_{crop} = amount_{irrig} area_{GP} N_{group} f_{area,crop}\, AC_{water,irrig}, \\ crop = \{vegetables, tuber\} \qquad (7\text{-}54)$$

where

$amount_{irrig}$	is the total area-specific amount of water used each year [$m^3\ m^{-2}\ y^{-1}$],
$area_{GP}$	is the surface area of the garden plot per individual [m^2],
N_{group}	is the number of individuals in a household,
$f_{area,crop}$	is the fraction of the garden plot used to cultivate vegetables or potatoes, [$m^2\ m^{-2}$] and,
$AC_{water,irrig}$	is the activity concentration of irrigation water [Bq m^{-3}].

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[5] Potential fraction of leaf uptake from intercepted water = $LeafStoreCapacity_{veg}\ LAI_{veg}$ / ($amount_{irrig}/N_{irrig}$) = 3×10^4 [m] × 5 [m/m] / (0.09/4) [m] = 6.7%. Parameter names explained in text.

7.3.4 Leaf retention

Leaf retention refers to the temporary retention of radionuclides in intercepted irrigation water on the surface of crop leaves. The resulting activity concentration is added to the internal activity concentration of vegetables to calculate the dose from ingestion of vegetables (see Chapter 9).

For each irrigation event, the amount of a radionuclide that is intercepted depends on the leaf area and the thickness of the water film retained on the leaf. After irrigation, precipitation, wind and plant growth causes the concentration on plant leaves to decrease over time, and the net outcome of these processes can be described by a characteristic weathering first-order rate constant (y^{-1}) (IAEA 2010). Consequently, the activity of a radionuclide on the surface of leaves [Bq m^{-2}] at time *t* after an irrigation event can be expressed as a function of the activity concentration in irrigation water, the thickness of the retained water layer from which the radionuclide can be removed by sorption and other processes to the leaf surface, the leaf area index, the rate of weathering and the time (after irrigation):

$$LeafRetention^{RN}(t) = AC_{water,irrig} LeafStoreCapacity_{veg} LAI_{veg} e^{-washoffCoef\, t}, \quad (7\text{-}55)$$

where

$AC_{water,irrig}$	is the activity concentration in irrigation water [Bq m^{-3}],
$LeafStoreCapacity_{veg}$	is the leaf storage capacity for leafy vegetables [m^3 m^{-2}],
LAI_{veg}	is the leaf area index, the area of leaves per unit surface area of soil for leafy vegetables [m^2 m^{-2}],
$washoffCoef$	is a rate constant describing the decrease of activity on leafy vegetables with time due to wind, rain and growth [y^{-1}], and
t	is the time since the last irrigation event [y].

Since leafy vegetables are harvested repeatedly during the whole growing period, and since the weathering half life is short (in the order of weeks) compared with the period between irrigation events (in the order of months), the mean concentration between two irrigation episodes is a reasonable approximation of the contribution of leaf retention to vegetable activity concentrations. The average activity concentration (Bq kgC^{-1}) can be calculated by integrating the above equation over the time between two irrigation events and dividing by the net above-ground primary production and the time interval:

$$AC_{LeafRet} = \frac{\int_0^{time_{IP}} LeafRetention(t)}{time_{IP}(NPP_{ag,veg}/N_{irrig})}, \quad (7\text{-}56)$$

where

$NPP_{ag,veg}$	is the yearly above-ground production of vegetables [kgC m^{-2} y^{-1}],
N_{irrig}	is the number of irrigation events during the vegetation period [y^{-1}], and
$time_{IP}$	is the time period between two irrigation events [y], which is approximately the vegetation period divided by the number of irrigation events:

$$time_{IP} = time_{vegPeriod}/N_{irrig} \quad (7\text{-}57)$$

where

$time_{vegPeriod}$ is the length of the vegetation period [y].

The average activity concentration on the leaves of vegetables is:

$$AC_{LeafRet} = \frac{AC_{water,irrig}\, LeafStoreCapacity_{veg}\, LAI_{veg}\, N_{irrig}\left(1 - e^{-washoffCoef\, time_{IP}}\right)}{time_{IP}\, NPP_{ag,veg}\, washoffCoef} \quad (7\text{-}58)$$

Note that leaf retention is considered a source of human exposure for leafy vegetables only. That is, for root crops (potatoes) the concentration in edible parts is assumed to be sufficiently represented by the equilibrium concentration with soil.

7.3.5 Leaf degassing (C-14)

Leaf degassing refers to the release of C-14 in intercepted irrigation water to the canopy atmosphere. This flux is used as a source term for calculations of radionuclide activity concentrations in the crop canopy atmosphere and the air above the crops (see Chapter 8).

Given the thin layer of water that is stored on crop leaves under irrigation (less than 0.3 mm, Barfield et al. 1973), the degassing process of DIC in intercepted water is expected to be very fast. For example, at pH 6.5 and low wind speeds (i.e. piston velocity ~2 cm/h), the dissolved inorganic carbon in a 0.3 mm layer of water is expected to have a turnover time of minutes. For the purpose of the assessment it can therefore be assumed that all C-14 intercepted by crops will equilibrate with the surrounding atmosphere before the crop is harvested.

As the specific activity of C-14 is expected to be many orders of magnitude smaller in the canopy atmosphere as compared to the concentration of C-14 in irrigation water, the back flux of C-14 from the atmosphere to the intercepted water will be very small. Thus it is reasonable to use the upper limit for leaf degassing as an approximation of the flux to the atmosphere. That is, to approximate the net release of C-14 to the atmosphere [Bq y^{-1}] with the activity in the intercepted irrigation water:

$$LeafDegassing^{14C} = Irrigation\, f_{LeafIntercepted} \tag{7-59}$$

where

$Irrigation$ is the yearly input of C-14 with irrigation water [Bq y^{-1}], and

$f_{LeafIntercepted}$ is the fraction of the irrigated water intercepted by the leaves [unitless]:

$$f_{LeafIntercepted} = \frac{LeafStoreCapacity_{crop}\ LAI_{crop}}{amount_{irrig}/N_{irrig}} \tag{7-60}$$

where

$LeafStoreCapacity_{crop}$ is the leaf storage capacity for crop [$m^3\ m^{-2}$],

$LAIcrop$ is the leaf area index, the area of leaves per unit surface area, for leafy vegetables [$m^2\ m^{-2}$],

$amount_{irrig}$ is the total area specific amount of water used each year [$m^3\ m^{-2}\ y^{-1}$],

N_{irrig} is the number of irrigation events during the vegetation period [y^{-1}].

Combining the two equations above, we get the following equation for degassing of C-14 for each crop:

$$\begin{aligned} &LeafDegassing^{14C}_{crop} \\ &= N_{irrig} area_{GP} N_{group} f_{area,crop}\ AC_{water,irrig} LeafStoreCapacity_{crop}\ LAI_{crop} \\ &crop = \{tuber, vegetables\} \end{aligned} \tag{7-61}$$

where

$AC_{water,irrig}$ is the activity concentration in irrigation water [Bq m^{-3}],

$area_{GP}$ is the area of the garden plot per individual [m^2],

N_{group} is the number of individuals in a household,

$f_{area,crop}$ is the fraction of the garden plot area used to cultivate vegetables or potatoes [$m^2\ m^{-2}$].

7.3.6 Leaching

Leaching refers to the flux of radionuclides from cultivated soil associated with percolation of meteoric water. The flux [Bq y^{-1}] is proportional to concentration of a radionuclide in the pore water and the downward flux of water (percolation), and can be calculated with Equation 7-12 (Infield-outland agriculture), using parameters for the garden plot and the two associated crops (potatoes and vegetables).

Equation 7-12 can be re-arranged, and leaching expressed as a function of the radionuclide inventory, the fraction of the inventory partitioned to the solute phase, soil physical properties, and the area specific runoff:

$$\begin{aligned} Leaching_{regoUp,GP,crop}^{RN} &= \frac{RegoUp_{GP,crop}^{RN}(1-f_{crop})f_{regoUp,GP}^{D}percolation_{agri}}{z_{regoUp,GP}\; poro_{regoUp,GP}S_{w,regoUp,GP}} \\ &= RegoUp_{GP,crop}^{RN}\; k_{leach,crop}^{RN}, \\ Leaching_{regoUp,crop}^{14C} &= \frac{RegoUp_{GP,crop}^{14C}\; f_{regoUp,GP}^{D}percolation_{agri}}{z_{regoUp,GP}\; poro_{regoUp,GP}S_{w,regoUp,GP}} \\ &= RegoUp_{GP,crop}^{14C}\; k_{leach,crop}^{14C} \end{aligned} \tag{7-62}$$

$crop = \{tuber, vegetables\}$

where

$percolation_{agri}$	is the area-specific downward flux of water [m^3 m^{-2} y^{-1}],
$RegoUp_{GP}$	is the inorganic activity in cultivated soil [Bq],
$z_{regoUp,GP}$	is the thickness of the biologically active layer [m],
$poro_{regoUp,GP}$	is the porosity of cultivated soils [m^3 m^{-3}],
$S_{w,regoUp,GP}$	is the degree of saturation of cultivated soils [m^3 m^{-3}],
$f_{regoUp,GP}^{D}$	is the fraction of radionuclides in solute phase in cultivated soil [Bq Bq^{-1}],
$percolation_{agri}$	is the area-specific downward flux of water in cultivated soil [m^3 m^{-2} y^{-1}], and
k_{leach}	is the proportional rate constant for leaching from soil associated with vegetables or potatoes.

7.3.7 Mineralisation

Mineralisation refers to the release of radionuclides from slow decomposition of refractory organic matter. Through this process, radionuclides originally stored in seaweeds are released to the soil pore water (Section 7.2). The release of a radionuclide [Bq y^{-1}] can be expressed as a fractional release rate of the organic inventory:

$$Mineralisation_{regoUp,org} = RegoUp_{org,GP}\; minRate_{GP} \tag{7-63}$$

where

$RegoUp_{org,GP}$	is the total radionuclide activity in soil organic matter,
$minRate_{GP}$	is the mineralisation rate associated with peat and post glacial deposits when drained and cultivated [kgC kgC^{-1} y^{-1}].

7.3.8 Degassing (Soil)

Degassing refers to the flux of inorganic C-14 from soil pores to the atmosphere.). Degassing [Bq y^{-1}] is described as a function of the inventory of the inorganic C-14 in soil, the gas diffusivity of CO_2 in soil and the depth of the cultivated soil (Equation 7-15):

$$Degassing^{14C} = 2\frac{f_{regoUp,GP}^{D}\, f_{gas\,,GP} D_{CO_2,soil,GP} RegoUp_{GP}^{14C}}{{z_{regoUp,GP}}^{2}\; poro_{regoUp,GP}(1 - S_{w,regoUp,GP})} = RegoUp_{GP}^{14C} k_{degass}^{14C} \qquad (7\text{-}64)$$

where

$RegoUp_{GP}^{14C}$	is the activity of inorganic C-14 in the cultivated soil [Bq],
$D_{CO_2,soil,GP}$	is the diffusivity of CO_2 in cultivated soil [m^2 y^{-1}],
$poro_{regoUp,GP}$	is the porosity of cultivated soil [m^3 m^{-3}],
$S_{w,regoUp,GP}$	is the degree of saturation in cultivated soil [m^3 m^{-3}],
$z_{regoUp,GP}$	is the thickness of biologically active layer [m],
$f_{regoUp,GP}^{D}$	is the fraction of C-14 in solute (non-solid) phase soil [Bq Bq^{-1}],
$f_{gas,GP}$	is the fraction of non-solid inorganic carbon available for degassing (Appendix B):

$$f_{gas,GP} = \frac{1 - S_{w,regoUp,GP}}{1 - S_{w,regoUp,GP,}(1 - solubilityCoef_{ter})} \qquad (7\text{-}65)$$

where

$solubilityCoef_{ter}$ is the solubility coefficient for CO_2 in pore water [mol m^{-3} per mol m^{-3}].

7.3.9 Release to atmosphere from combustion

This flux refers to the release of radionuclides in combustion gas and fly ash from burning firewood or peat. This flux is used as source term for calculating exposure through outdoor inhalation (see exposure and dose calculations in Chapter 9).

The release of radionuclides through combustion of biomass fuel [Bq y^{-1}] is proportional to the activity concentration in the fuel, the amount of fuel burnt, and the fraction of radionuclides released in gas and fly ash:

$$\begin{aligned} Release_{combust,wood} &= fuel_{cons,wood}\, f_{combust}\, AC_{PP,ter} f_{C,pp.ter}\, \alpha_{fuel,wood} \\ Release_{combust,peat} &= fuel_{cons,peat}\, f_{combust}\, AC_{regoPeat,tot} \alpha_{fuel,peat} \end{aligned} \qquad (7\text{-}66)$$

where

$fuel_{cons,type}$	is the consumption rate of biomass fuel needed to heat a household (~20,000 kWh) by combustion of peat or firewood [kg_{DW} y^{-1}],
$AC_{PP,ter}$	is the activity concentration in mire vegetation[Bq kgC^{-1}]
$f_{C,pp.ter}$	is the mass of carbon per unit dry weight of mire vegetation [kgC ${kg_{DW}}^{-1}$]
$AC_{regoPeat,tot}$	is the total activity concentration in peat, including radionuclides in organic and inorganic form [Bq ${kg_{DW}}^{-1}$],
$f_{combust}$	is the fraction of the fuel inventory that ends up in fly ash and gas after combustion of wood or peat [kg_{DW} ${kg_{DW}}^{-1}$], and
$\alpha_{fuel,type}$	is a potential dilution factor due to limitations of fuel resources in the biosphere object (see Section 7.3.2) with ash type being wood or peat [unit less].

7.3.10 Average activity concentration from irrigation and fertilisation

A kitchen garden is considered to continually be used for one generation or 50 years, (which equals the time used to evaluate risk in the assessment). During this period, the radionuclide inventory in the soil will increase, and the dynamics can be described by the following ordinary differential equation (ignoring GP subscripts for simplicity):

$$\begin{aligned}\frac{dRegoUp^{14C}_{org,GP}}{dt} &= Input^{14C}_{Fertil,org} - Decay^{14C}_{regoUp,GP} - Miner^{14C}_{regoUp,GP} \\ &= Input^{14C}_{Fertil,org} - RegoUp^{14C}_{org,GP}\left(\lambda^{14C} + minRate_{regoUp,GP}\right)\end{aligned} \tag{7-67}$$

$$Input^{14C}_{Fertil,org} = \begin{cases} Fertil^{14C}_{algae}\ area_{GP}\ N_{group}\ f_{refrac,ter} & \text{during submerged period} \\ 0 & \text{during land period}\end{cases}$$

$$\begin{aligned}\frac{dRegoUp^{14C}_{GP}}{dt} &= Irrig^{14C} + Miner^{14C}_{regoUp,GP} - Decay^{14C}_{regoUp,GP} - Leach^{14C}_{regoUp,GP} \\ &\quad - Degass^{14C}_{regoUp,GP} \\ &= Irrig^{14C} + Miner^{14C}_{regoUp} - RegoUp^{14C}\left(\lambda^{14C} + k^{14C}_{leach} + k^{14C}_{degass}\right)\end{aligned} \tag{7-68}$$

$$\begin{aligned}\frac{dRegoUp^{RN}_{org,GP}}{dt} &= Input^{RN}_{Fertil,org} - Decay^{RN}_{regoUp,GP} - Miner^{RN}_{regoUp,GP} \\ &= Input^{RN}_{Fertil,org} - RegoUp^{RN}_{org,GP}\left(\lambda^{RN} + minRate_{regoUp,GP}\right)\end{aligned} \tag{7-69}$$

$$Input^{RN}_{Fertil,org} =$$
$$= \begin{cases} Fertil^{RN}_{algae} area_{GP}\ N_{group}\left(1 - \left(1 - f_{refrac,ter}\right)df_{decomp,ter}\right) & \text{during submerged period} \\ 0 & \text{during land period}\end{cases}$$

$$\begin{aligned}\frac{dRegoUp^{RN}_{GP,crop}}{dt} &= f_{area,crop}\left(\left(Input^{RN}_{Fertil} + Irrig^{RN} + Miner^{RN}_{regoUp,GP}\right)\right. \\ &\quad \left. - \left(Decay^{14C}_{regoUp,GP} + Leach^{14C}_{regoUp,GP,crop}\right)\right) \\ &= f_{area,crop}\left(\left(Input^{RN}_{Fertil} + Irrig^{RN} + Miner^{RN}_{regoUp,GP}\right)\right. \\ &\quad \left. - RegoUp^{14C}_{GP}\left(\lambda^{14C} + k^{14C}_{leach,crop}\right)\right)\end{aligned} \tag{7-70}$$

$$crop = \{vegetables, tuber\}$$

Solving the differential equations for the radionuclide inventories, and division by the soil mass, yields the following expressions for activity concentrations in cultivated soil. Note that for the activity concentration of radionuclides other than C-14 in inorganic form we approximate the input of radionuclides from mineralisation as a constant flux.

$$AC^{14C}_{RegoUp,org,GP}(t) = \left(Input^{14C}_{Fertil,org}\right)\frac{1 - e^{-\left(\lambda^{14C} + minRate\right)t}}{\left(\lambda^{14C} + minRate\right)V\rho} \tag{7-71}$$

$$AC^{14C}_{RegoUp,GP}(t) = \left(Irrig^{14C} + Miner^{14C}_{regoUp,GP}\right)\ \frac{1 - e^{-\left(\lambda^{14C} + k^{14C}_{leach} + k^{14C}_{degass}\right)t}}{\left(\lambda^{14C} + k^{14C}_{leach} + k^{14C}_{degass}\right)V\rho} \tag{7-72}$$

$$AC^{RN}_{RegoUp,org,GP}(t) = \left(Input^{RN}_{Fertil,org}\right)\frac{1 - e^{-\left(\lambda^{\mathrm{RN}} + minRate\right)t}}{\left(\lambda^{14C} + minRate\right)V\rho} \tag{7-73}$$

$$AC^{RN}_{RegoUp,crop,GP}(t) = \left(Input^{RN}_{Fertil} + Irrig^{RN} + Miner^{RN}_{regoUp,GP}\right)\ \frac{1 - e^{-\left(\lambda^{\mathrm{RN}} + k^{\mathrm{RN}}_{leach,crop}\right)t}}{\left(\lambda^{\mathrm{RN}} + k^{\mathrm{RN}}_{leach,crop}\right)V\rho} \tag{7-74}$$

$$V\rho = z_{regoUp}\ area_{cult}\ N_{group} dens_{regoUp} \tag{7-75}$$

To get the average soil activity for the interval T between the time of cultivation start (t_0) and the end of the assessment period (t_{end}), we need to integrate the above functions and divide by the length of the time interval for cultivating the garden-plot. Thus the average activity concentration in cultivated soil, with respect to the organic compartment, is:

$$\begin{aligned}\overline{AC}_{RegoUp,org,GP} &= \frac{1}{(t_{\mathrm{end}} - t_0)} \int_{t_0}^{t_{\mathrm{end}}} AC_{RegoUp,org}(t)dt = \{\mathrm{t}_0 = 0\} \\ &= \frac{Input_{Fertil,org}/\left(area_{GP}\ N_{group}\right)}{(\lambda + minRate) z_{regoUp} dens_{regoUp}} f_{regoUp,org,dynam}\end{aligned} \quad (7\text{-}76)$$

where

$f_{regoUp,org,dynam}$ describes the deviation of the inventory from steady-state conditions:

$$f_{regoUp,org,dynam} = 1 - \frac{1 - \mathrm{e}^{-(\lambda + minRate)t_{\mathrm{end}}}}{(\lambda + minRate)t_{\mathrm{end}}} \quad (7\text{-}77)$$

Similarly, by setting $k_{degass}= 0$ for other radionuclides than C-14, and $f_{crop} = 0$ for C-14, the average activity concentration in cultivated soil, with respect to the inorganic compartment, is:

$$\begin{aligned}\overline{AC}_{RegoUp,GP} &= \frac{1}{(t_{50} - t_0)} \int_{t_0}^{t_{50}} AC_{RegoUp}(t)dt = \{\mathrm{t}_0 = 0\} \\ &= \frac{\sum Input/\left(area_{cult}\ N_{group}\right)}{(\lambda + k_{degass} + k_{leach}) z_{regoUp} dens_{regoUp}} f_{regoUp,dynam}\end{aligned} \quad (7\text{-}78)$$

Where the expression $1\text{-}f_{regoUp,dynam}$ describes the deviation of the inventory from steady-state conditions [unitless], and k_{degass} and k_{leach} describe the proportional transfer constant associated with degassing and leaching [y^{-1}], and $\sum$ Input represents the sum of radionuclide fluxes into the inorganic regolith compartment [Bq y^{-1}]:

$$f_{regoUp,dynam} = 1 - \frac{1 - e^{-(\lambda + k_{degass} + k_{leach})t_{\mathrm{end}}}}{\left(\lambda + k_{degass} + k_{leach}\right)t_{\mathrm{end}}} \quad (7\text{-}79)$$

$$k_{degass}^{14C} = \frac{2 f_{regoUp}^{D}\ f_{gas}\ D_{CO_2,soil}}{{z_{regoUp}}^2\ poro_{regoUp}\left(1 - S_{w,regoUp}\right)} \quad (7\text{-}80)$$

$$k_{leach,crop} = \frac{(1 - f_{crop}) f_{regoUp}^{D} percolation_{cult}}{z_{regoUp}\ poro_{regoUp} S_{w,regoUp}} \quad (7\text{-}81)$$

$$\sum input = \ Input_{Fertil} + Irrigigation + \ Mineralis_{RegoUp} \quad (7\text{-}82)$$

Where $\mathrm{Input}_{\mathrm{Fertil}} = 0$ for C-14 and the contribution from mineralisation is approximated as the average rate over the assessment period:

$$\begin{aligned}Mineralis_{regoUp} &= \ \overline{AC}_{RegoUp,org} V\rho\ minRate \\ &= \frac{Input_{Fertil,org} minRate}{\lambda + minRate} f_{regoUp,org,dynam}\end{aligned} \quad (7\text{-}83)$$

Note that when the radionuclide input terms are expressed on a per unit area the average activity concentrations is independent on the surface area of the garden plot.

8 Surface atmosphere

This chapter provides a short description of the sub-models used to calculate the activity concentration (and the specific activity) of $^{14}CO_2$ in the near-surface atmosphere. That is, in the atmospheric layers where gas is exchanged with soil and water, where plant uptake occurs and where humans may be exposed to radionuclides in gaseous form due to inhalation. Two models are presented: i) for C-14 releases to the atmosphere from vegetated land driven by soil and plant leaf degassing (Section 8.1) and ii) for C-14 releases to the atmosphere above a water body, such as a lake, driven by water degassing (Section 8.2).

A more complete description of the relatively simple models, and a comparison with more detailed, process-oriented models, can be found in Avila and Kovalets (2014). That report also includes an analysis of the impact of model simplifications on endpoint predictions.

8.1 Surface atmosphere above vegetated land

The conceptual model of the near-surface atmosphere above vegetated land includes three atmospheric layers (Figure 8-1). C-14 is released to the layer closest to the surface (see below), and it is assumed that C-14 will be fully mixed with stable carbon (C-12) in all the atmospheric layers. Both isotopes are transported vertically by turbulence and laterally with the advective flux of air, and steady-state conditions are assumed to occur on the time scale relevant to the safety assessment (i.e. over a growing season or a year). The target endpoints of the simulations with the model are the C-14 activity concentrations in primary producers, used in the calculations of doses from food ingestion and in atmospheric air, used in the calculation of doses from inhalation.

The first atmospheric layer, the canopy layer (CA), extends from the soil surface to the top of the canopy of the vegetation. This is the layer where gas exchange between air and soil occurs, and where primary producers fix atmospheric carbon by photosynthesis. The second layer extends from the top of the canopy of the vegetation to 2.5 m above the ground. As the canopy of all vegetation types included in the assessment is below 1 m (Grolander 2013), it is in this second layer that humans get exposed to gaseous C-14 by inhalation. The third layer is required for representation of recycling of C-14 with turbulent transport. The assumed thickness of this layer has a minor impact on the predictions for the targeted endpoints (Avila and Kovalets 2014).

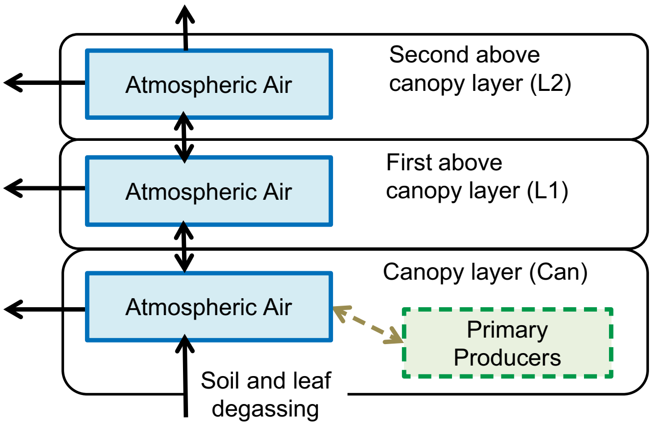

Figure 8-1. *A graphical representation of the model used to calculate the outcome of C-14 transport in the surface atmosphere above vegetated land. Boxes represent radionuclide inventories and arrows fluxes of the radionuclide. Primary producers are assumed to be in steady state exchange with the atmospheric air of the canopy layer through photosynthesis and respiration (brown dashed arrow). The vertical and horizontal arrows represent C-14 fluxes driven by vertical turbulent and lateral advective air fluxes (black arrows). Releases of C-14 to the canopy layer occurs by degassing from the soil and by degassing of irrigation water intercepted by leaves (garden-plot only, Section 7.3).*

8.1.1 C-14 specific activity in the canopy layer

At steady state, the specific C-14 activity in canopy air can be obtained by dividing the C-14 flux from this compartment by the flux of stable carbon from the same compartment. Moreover, if radioactive decay of C-14 in the atmosphere is neglected, the C-14 flux from the canopy atmosphere will equal the net flux of C-14 into this compartment. Consequently, the C-14 specific activity in the canopy atmosphere, $\mathrm{SA}^{14C}_{atmos,can}$ [$\mathrm{BqkgC^{-1}}$], can be expressed as the ratio between the flux of C-14 into the canopy atmosphere and the flux of stable carbon out of the canopy atmosphere:

$$SA^{14C}_{atmos,can,i} = \frac{ReleaseRate_i}{Flux_{PP,i} + Flux_{turb,i} + Flux_{adv,i}} \tag{8-1}$$

where

ReleaseRate	is the area normalized release rate of C-14 to the canopy layer [$\mathrm{Bq\ m^{-2}\ y^{-1}}$],
$Flux_{PP}$	is the area normalized net flux of stable carbon from the canopy air to the primary producers [$\mathrm{kgC\ m^{-2}\ y^{-1}}$],
$Flux_{adv}$	is the area normalized net flux of stable carbon from the canopy air by advective transport [$\mathrm{kgC\ m^{-2}\ y^{-1}}$],
$Flux_{turb}$	is the area normalized net flux of stable carbon from the canopy air by turbulent transport [$\mathrm{kgC\ m^{-2}\ y^{-1}}$], and
i	is the index standing for mire (ter) or crop within the cultivation system ($\mathrm{cereal_{IO}}$, $\mathrm{cereal_{DM}}$, $\mathrm{tuber_{DM}}$, $\mathrm{tuber_{GP}}$, $\mathrm{vegetables_{GP}}$, $\mathrm{fodder_{DM}}$).

8.1.2 Release of C-14 to the canopy layer

In this context, the release rate of C-14 into the canopy layer [$\mathrm{Bq\ m^{-2}\ y^{-1}}$] consists of fluxes from *soil degassing* (all terrestrial ecosystems, see degassing in Chapters 6 and 7), from fast decomposition of organic fertilisers (*litter respiration*, Chapter 7) and degassing of irrigation water intercepted by plant leafs (*leaf degassing*, Garden plot, Section 7.3.5):

$$\begin{aligned}
ReleaseRate_{ter} &= \frac{Degassing^{14C}_{ter}}{area_{obj,ter}} \\
ReleaseRate_{IO,} &= \frac{Degassing^{14C}_{IO} + Litter^{14C}_{resp,IO}}{area_{IO} \cdot N_{group,IO}} \\
ReleaseRate_{DM} &= \frac{Degassing^{14C}_{DM}}{area_{DM} \cdot N_{group,DM}} \\
ReleaseRate_{GP} &= \frac{Degassing^{14C}_{GP} + Litter^{14C}_{resp,GP} + LeafDegassing^{14C}}{area_{GP} \cdot N_{group,GP}}
\end{aligned} \tag{8-2}$$

where

Degassing	is the release rate of C-14 to the canopy layer from soil [$\mathrm{Bq\ y^{-1}}$],
$Litter_{resp}$	is the release rate of C-14 from decomposition of labile organic matter [$\mathrm{Bq\ y^{-1}}$],
$area_{obj,ter}$	is the surface area of the wetland [$\mathrm{m^2}$],
$area_{IO/DM/GP}$	is the surface area of cultivation per individual [$\mathrm{m^2}$], and
N_{group}	is the number of individuals in the most exposed group [unitless].

8.1.3 Flux to primary producers

The net flux of C-14 from the canopy atmosphere to primary producers was described as *plant uptake* in the mire ecosystem (Section 6.2.1). Expressed per unit area, the corresponding uptake of stable carbon, Flux_{PP} [kgC m^{-2} y^{-1}], can be described with one equation for all terrestrial ecosystems:

$$Flux_{PP,i} = NPP_i\,(1 - f_{rootUptake}) \tag{8-3}$$

where

NPP_i is the area-specific net primary production in ecosystem i [kgC m^{-2} y^{-1}],

$f_{rootUptake}$ is the fraction of carbon assimilated via root uptake [kgC kgC^{-1}], and

i is the index standing for mire or crop within cultivation system (cereal_{IO}, cereal_{DM}, tuber_{DM}, tuber_{GP} , $\mathrm{vegetables}_{GP}$, fodder_{DM}).

8.1.4 Turbulent flux

The area normalized net turbulent flux of carbon [kgC m^{-2} y^{-1}] from the canopy air is the product of the area specific turbulent upward flux of air and the carbon concentration in the atmospheric air, corrected for the downward turbulent back-flux of carbon. Ignoring the indexes for vegetation type, this flux is:

$$Flux_{turb} = Vel_{exch,Ca,L1} \cdot Conc_{C,atmos}\,(1 - RF_{L1ToCA}) \tag{8-4}$$

where

$Conc_{C,atmos}$ is the carbon concentration in atmospheric air [kgC m^{-3}],

$Vel_{exch,Ca,L1}$ is the velocity of air exchange between the canopy and first above canopy layers by turbulent transport [m y^{-1}], and

RF_{L1ToCA} is the carbon recycling factor from the first above-canopy layer to the canopy layer [unitless].

The carbon that enters the first above-canopy layer, from the canopy layer, can be transported laterally with the advective flux of air, vertically to the second above-canopy layer or recycled back to the canopy layer by turbulent transport. The fraction that is transported by a specific pathway will depend on the relative magnitude of the air exchange velocity of this pathway, as compared with other competing pathways. The recycled fraction (RF) is calculated as the ratio of the velocity of air exchange between the canopy and first above-canopy layers and the total velocity of air exchange in the first above canopy layer (Equation 8-5). This ratio corresponds to the probability that a molecule of stable carbon (and also C-14) released from the canopy layer to the first above-canopy layer, will be recycled back to the canopy layer.

$$RF_{L1ToCA} = \frac{Vel_{exch,CA,L1}}{Vel_{exch,L1,L2}\left(1 - RF_{L2ToL1}\right) + Vel_{exch,CA,L1} + Vel_{Adv,L1}} \tag{8-5}$$

where

$Vel_{exch,L1,L2}$ is the velocity of air exchange between the first and second above-canopy layers by turbulent transport [m y^{-1}],

$Vel_{Adv,L1}$ is the air exchange velocity of the first canopy layer by advective transport [m y^{-1}] (see below), and

RF_{L2ToL1} is the carbon recycling factor from the second above-canopy layer to the first above-canopy layer [unitless].

The Recycling Factor from the second to the first above-canopy layer is calculated similarly (see Equation 8-6). However, in this case, the recycling from the uppermost layer to the second above-canopy layer has been neglected for simplicity. This is justified by the fact that concentrations in air of stable carbon (and C-14) originated from the canopy layer substantially decreases with the height above canopy. As shown in Avila and Kovalets (2014), this simplification does not lead to a significant underestimation of the Recycling Factor.

$$RF_{L2ToL1} = \frac{Vel_{exch,L1,L2}}{Vel_{exch,L2,Upp} + Vel_{exch,L1,L2} + Vel_{Adv,L2}} \tag{8-6}$$

where

$Vel_{exch,L2,Upp}$ is the velocity of air exchange between the second above canopy layer and upper layers of the atmosphere by turbulent transport [m y^{-1}], and

$Vel_{Adv,L2}$ is the air exchange velocity of the second canopy layer by advective transport [m y^{-1}].

The upward turbulent exchange velocities, Vel_{exch} [m y^{-1}], are calculated as the inverse of the average resistances of two adjacent layers, which follows from the definition of resistance (see below). For simplicity, the resistance of the uppermost layer (above the second above-canopy layer) has been neglected. This simplification is justified by the fact that the resistance decreases substantially with height and as shown in Avila and Kovalets (2014), this simplification does not lead to significant overestimation of the exchange rates. The equations used for calculating the exchange velocities are:

$$Vel_{exch,CA,L1} = \frac{1}{0.5(res_{CA} + res_{L1})} \tag{8-7}$$

$$Vel_{exch,L1,L2} = \frac{1}{0.5(res_{L1} + res_{L2})} \tag{8-8}$$

$$Vel_{exch,L2,Upp} = \frac{1}{0.5 \cdot res_{L2}} \tag{8-9}$$

where

res_{CA} is the resistance of the canopy layer to turbulent transport [y m^{-1}],

res_{L1} is the resistance of the first above-canopy layer to turbulent transport [y m^{-1}], and

res_{L2} is the resistance of the second above-canopy layer to turbulent transport [y m^{-1}].

The resistance to turbulent transport, *res* [y m^{-1}], of an atmospheric layer is defined as the ratio between the thickness and the eddy diffusion coefficient of the layer (Wilson 1989, Wilson and Sawford 1996, Finnigan 2000, Baldocchi et al. 1983, Shuttleworth and Wallace 1985, Shuttleworth and Gurney 1990):

$$res_{CA} = \frac{z_{CA}}{D_{CA}} \qquad res_{L1} = \frac{z_{L1}}{D_{L1}} \qquad res_{L2} = \frac{z_{L2}}{D_{L2}} \tag{8-10}$$

where

z is the thickness of an atmospheric layer (canopy, first and second above the canopy) [m], and

D is the eddy diffusion coefficient of corresponding layer [m^2 y^{-1}],

For the above canopy layers (L1 and L2), we assume that the eddy diffusion coefficient at a given height above the ground (*h*) is proportional to the height above the displacement plane ($height_{displ}$), i.e. to ($h - height_{displ}$). Under this assumption the eddy diffusion coefficients, *D* [m^2 y^{-1}], for these layers become (Avila and Kovalets 2014):

$$D_{L1} = \frac{Karman_{const} \cdot Vel_{frict} \cdot z_{L1}}{ln\left(\frac{height_{L1} - height_{displ}}{height_{CA} - height_{displ}}\right)}$$

$$D_{L2} = \frac{Karman_{const} \cdot Vel_{frict} \cdot z_{L2}}{ln\left(\frac{height_{L2} - height_{displ}}{height_{L1} - height_{displ}}\right)} \tag{8-11}$$

$$height_{displ} = 0.75 \cdot height_{CA}$$

where

$Karman_{const}$ is the von Karman constant [unitless],

Vel_{frict} is the friction velocity [m y^{-1}],

$height_{CA/L1/L2}$ is the height above the ground of the upper boundary of the canopy, the L1 and the L2 layers, respectively [m], and

$height_{displ}$ is the height of the displacement plane assumed to be 75% of the canopy height [m]

The displacement plane is the height above the surface where the wind speed is taken to fall to zero. This is assumed to lie within the canopy. The friction velocity (also called shear velocity), is nothing more than a convenient way of defining a velocity scale close to a surface, by which the shear stress may be re-written in units of velocity. The friction velocity is calculated from the wind speed at reference height, assuming that the wind speed increases logarithmically with height:

$$Vel_{frict} = \frac{Karman_{const} \cdot Vel_{wind,heigh,ref,ter}}{ln\left(\frac{height_{ref,ter} - height_{displ}}{z_0}\right)} \quad (8\text{-}12)$$

$$z_0 = 0.1 \cdot height_{CA}$$

where

$Vel_{wind,heigh,ref,ter}$ is the wind speed at the reference height $height_{ref,ter}$ [m y^{-1}],

$height_{ref,ter}$ is the reference height [m], and

z_0 is the roughness length, assumed to be 10% of the canopy height [m].

More rigorously, Equation 8-11 should only be used for above-canopy layers in neutral atmospheric stability conditions. For other stability conditions and short time scales, it is necessary to account for the impact from the canopy on the turbulence in the layers that are immediate above the canopy (Tagesson 2012). However, for the time scales that are relevant for this safety assessment (season or year) corrections for other stability conditions have negligible effects on the calculated turbulent fluxes and activity concentrations in the air (Avila and Kovalets 2014).

Nevertheless, using Equation 8-11 for the canopy layer would lead to overestimation of the eddy diffusion coefficients and the turbulent upward fluxes of air. Instead, an exponential dependency of the eddy diffusion coefficients with height is commonly assumed (Shuttleworth and Gurney 1990). With an exponential dependency, the eddy diffusion coefficient of the canopy layer, D_{CA} [m^2 y^{-1}], becomes:

$$\mathrm{D_{CA}} = \frac{D(z_{CA}) \cdot Coeff_{ext}}{e^{Coeff_{ext}} - 1}$$

$$D(z_{CA}) = Karman_{const} \cdot Vel_{frict} \cdot (z_{CA} - height_{displ}) \quad (8\text{-}13)$$

where

$D(z_{CA})$ is the Eddy diffusion coefficient at the canopy height [m^2 y^{-1}], and

$Coeff_{ext}$ is the extinction coefficient [unitless].

The coefficient of extinction, $Coeff_{ext}$ [unitless], determinates the rate of decrease of the eddy diffusion coefficient with height in the canopy. It is calculated from canopy properties such as the leaf drag and the mixing length inside the canopy (Kustas 1990):

$$\mathrm{Coeff_{ext}} = \sqrt{\frac{dragCoef \cdot LAI \cdot z_{CA}}{L_m}} \qquad L_m = \sqrt{\frac{4 \cdot leaf_{width} \cdot z_{CA}}{\pi \cdot LAI}} \quad (8\text{-}14)$$

where

$dragCoef$ is the drag coefficient of the canopy [unitless],

LAI is the leaf area index of the canopy of mire vegetation or cultivated species [m^2 m^{-2}],

L_m is the mean mixing length inside the canopy [m], and

$leaf_{width}$ is the leaf width of mire-canopy-forming vegetation or cultivated species [m].

8.1.5 Advective flux

The recycling to the canopy air of C-12 (and C-14) by advection can be neglected, because substantial dilution takes place downstream. Hence, the total flux of carbon by advective transport can be used in Equation 8-1, instead of the net flux. The area specific total advective flux of carbon from the canopy layer, $Flux_{adv}$ [kgC m^{-2} y^{-1}], is the product of the area-specific horizontal advective flux of air and the concentration of stable carbon in the canopy air:

$$Flux_{adv} = Vel_{Adv,CA} \cdot conc_{C,atmos} \qquad (8\text{-}15)$$

where

$conc_{C,atmos}$	is the carbon concentration in the canopy layer [kgC m^{-3}], and
$Vel_{Adv,CA}$	is the air-exchange velocity of the canopy layer by advective transport [m y^{-1}].

The advective air-exchange velocity is defined as the advective flux of air through a plane perpendicular to the main wind direction, normalized by the area of the object. To parameterise the exchange velocity, the atmospheric layers are represented as a cube with an area equal to the area of the object that receives the releases and a height equal to the thickness of the corresponding atmospheric layer. For this geometry, the advective exchange velocities, Vel_{Adv} [m y^{-1}], of the different layers (Canopy, L1 and L2) can be calculated with the following equation:

$$Vel_{Adv,i} = \frac{Vel_{wind,i} \cdot z}{\sqrt{area_j}} \qquad i = \{CA, L1, L2\} \qquad (8\text{-}16)$$

where

Vel_{wind}	is the average wind speed in the three atmospheric layers [m y^{-1}],
i	is the index for the canopy, the first above and the second above canopy layers,
j	is the index for the mire (ter) or the cultivation system: infield-outland (IO), drained mire (DM) or garden plot (GP), and
area	is the surface area of the mire or the cultivated land (including *all* cultivated area irrespective of crop) [m^2]:

$$area_j = \begin{cases} area_{obj,ter} & j = ter \\ min(area_{obj,ter}, area_j\ N_{group,j}) & j = DM \\ area_j\ N_{group,j} & j = \{IO, GP\} \end{cases} \qquad (8\text{-}17)$$

For the above canopy layers (L1 and L2) the wind speed is assumed to increase as a logarithmic function of height above the displacement plane (*HAD*). Thus the average wind speed in these layers can be expressed as:

$$Vel_{wind,L1} = \frac{Vel_{frict}}{Karman_{const}}\left(\left(\frac{HAD_{L1}}{z_{L1}}\right)\left(\ln\left(\frac{HAD_{L1}}{z_0}\right) - 1\right) - \left(\frac{HAD_{CA}}{z_{L1}}\right)\left(\ln\left(\frac{HAD_{CA}}{z_0}\right) - 1\right)\right)$$

$$HAD_{CA} = height_{CA} - height_{displ} \qquad HAD_{L1} = height_{L1} - height_{displ}$$

$$Vel_{wind,L2} = \frac{Vel_{frict}}{Karman_{const}}\left(\left(\frac{HAD_{L2}}{z_{L2}}\right)\left(\ln\left(\frac{HAD_{L2}}{z_0}\right) - 1\right) - \left(\frac{HAD_{L1}}{z_{L2}}\right)\left(\ln\left(\frac{HAD_{L1}}{z_0}\right) - 1\right)\right) \qquad (8\text{-}18)$$

$$HAD_{L2} = height_{L2} - height_{displ}$$

$$height_{displ} = 0.75 \cdot height_{CA} \qquad z_0 = 0.1 \cdot height_{CA}$$

where

$Karman_{const}$	is the von Karman constant [unitless],
Vel_{frict}	is the friction velocity [m y^{-1}],
$height_{CA/L1/L2}$	is the height above the ground of the upper boundary of the canopy, the L1 and the L2 layers, respectively [m], and
$height_{displ}$	is the height of the displacement plane assumed to be 75% of the canopy height [m]
z	is the thickness of an atmospheric layer (canopy, first and second above the canopy) [m],
z_0	is the roughness length, assumed to be 10% of the canopy height [m].

For the canopy layer (CA) the wind speed is assumed to increase as an exponential function of height above the ground (Shutleworth and Gurney 1990). The average wind speed in the layer can be calculated as:

$$Vel_{wind,CA} = Vel_{wind}(z_{CA})\frac{1-e^{-Coeff_{ext}}}{Coeff_{ext}} \tag{8-19}$$

where

$Vel_{wind}\ (z_{CA})$ is the wind speed at the canopy height [m y^{-1}], and

$Coeff_{ext}$ is the extinction coefficient [unitless] (see Equation 8-14)

From the logarithmic wind profile above the displacement plane, assumed for the above canopy layers, the wind speed at the canopy height can be calculated as:

$$Vel_{wind}(z_{CA}) = \frac{Vel_{frict}}{Karman_{const}} \cdot ln\left(\frac{z_{CA} - height_{displ}}{z_0}\right) \tag{8-20}$$

8.1.6 Activity concentrations

The main outputs of the atmosphere sub-model are the C-14 specific activity in the canopy atmosphere (Equation 8-1), which is used to calculate plant uptake (Section 6.2.1) and the activity concentration in crop biomass (Sections 9.2.2 and 9.3.3). For this, the specific activity of the newly created biomass is assumed to equal the specific activity in the canopy atmosphere (Avila and Pröhl 2008). In addition, the C-14 activity concentration in air is calculated, which is used to estimate exposure from inhalation (Section 9.1). For ecosystems with a canopy height above 2 m, the activity concentrations in the canopy air can be used to calculate inhalation doses. However, as fens are expected to be open (Chapter 6) and cultivated crops have a limited height (< 1 m), the activity concentrations in the first above-canopy layer are used for this purpose. The C-14 activity concentration in atmospheric air, $AC^{14C}_{atmos,CA}$ [Bq m^{-3}] of the canopy layer is calculated as:

$$AC^{14C}_{atmos,CA,i} = SA^{14C}_{atmos,can,i} \cdot Conc_{C,atmos} \tag{8-21}$$

where

$AC^{14C}_{atmos,can}$ the C-14 specific activity in the canopy atmosphere, [Bq kgC^{-1}],

$Conc_{C,atmos}$ is the carbon concentration in air [kgC m^{-3}], and

i is the index standing for mire (ter) or crop within cultivation system ($cereal_{IO}$, $cereal_{DM}$, $tuber_{DM}$, $tuber_{GP}$, $vegetables_{GP}$, $fodder_{DM}$).

In this assessment, the same value of carbon concentration in air has been used in Equations 8-4 and 8-21. More rigorously, in Equation 8-4 the value corresponding to the first above-canopy layer should have been used and in Equation 8-21 the value for the canopy layer would have been more appropriate. However, the difference between these values is negligible for this assessment context (Avila and Kovalets 2014) and, for this reason, a general value of carbon concentration in air has been used in both equations.

The area normalized C-14 flux from the canopy layer to the first above-canopy layer equals the C-14 activity concentration in the canopy air (Equation 8-21) times the velocity or air exchange between these layers, which is the numerator in Equation 8-5. At the same time, the total area normalized C-14 flux from the first above-canopy layer equals the C-14 activity concentration in this layer times the total velocity of air exchange of this layer, which is the denominator in Equation 8-5. At steady state, these two fluxes equal each other. From this line of reasoning the following equation for the C-14 activity concentration in the atmospheric air of the first above canopy layer is obtained:

$$AC^{14C}_{atmos,L1,i} = AC^{14C}_{atmos,CA,i} \cdot RF_{L1ToCA,i} \tag{8-22}$$

where

$RF_{L1ToCA,i}$ is the carbon recycling factor from the first above canopy layer to the canopy layer [unitless].

8.2 Surface atmosphere above water bodies

The conceptual model of the near-surface atmosphere above water bodies includes two atmospheric layers and is similar to the model presented in (Figure 8-1), but without the canopy layer. C-14 is released to the layer closest to the surface (see below), and it is assumed that C-14 will be fully mixed with stable carbon (C-12) in all atmosphere layers. Both isotopes are transported vertically by turbulence and laterally with the advective flux of air, and steady-state conditions are assumed to apply at the time scale relevant for the safety assessment (i.e. over a year). The target endpoint of the model is the C-14 activity concentration in atmospheric air, used in the calculation of the reverse C-14 flux from the atmosphere to the surface water.

The first atmospheric layer, the L1 layer, extends from the water surface to a height of 1 m. This is the layer where gas exchange between air and water occurs. The second layer is required for representation of recycling of C-14 with turbulent transport. The assumed thickness of this layer has little impact on the prediction of the target endpoint (Avila and Kovalets 2014).

8.2.1 Activity concentrations in atmospheric air

The C-14 activity concentration in the first layer, L1, can be obtained by dividing the area normalized release of C-14 into this layer by the total velocity of air exchange of this layer, including the net air exchange with the second layer, L2 by turbulent transport and the lateral air exchange by advective transport:

$$AC^{14C}_{atmos,L1} = \frac{ReleaseRate}{Vel_{exch,L1,L2}(1 - RF_{L2ToL1}) + Vel_{Adv,L1}} \tag{8-23}$$

where

$RF_{L1ToCA,i}$	is the area normalized release rate of C-14 from the water to the L1 layer [Bq m^{-2} y^{-1}],
$Vel_{exch,L1,L2}$	is the velocity of air exchange between the first, L1, and second, L2, layers by turbulent transport [m y^{-1}],
$Vel_{Adv,L1}$	is the air exchange velocity of the L1 layer by advective transport [m y^{-1}], and
RF_{L2ToL1}	is the air recycling factor from the second layer to the first layer by turbulent transport [unitless].

8.2.2 Turbulent fluxes

The equation for the Recycling Factor, RF [unitless], from L2 to L1 is the same as Equation 8-6, which was used in Section 8.1.4 for the surface atmosphere above vegetated lands:

$$RF_{L2ToL1} = \frac{Vel_{exch,L1,L2}}{Vel_{exch,L2,Upp} + Vel_{exch,L1,L2} + Vel_{Adv,L2}} \tag{8-24}$$

where

$Vel_{exch,L1,L2}$	is the velocity of air exchange between the first and second layers by turbulent transport [m y^{-1}],
$Vel_{exch,L2,Upp}$	is the velocity of air exchange between the second and upper layers of the atmosphere by turbulent transport [m y^{-1}],
$Vel_{Adv,L2}$	is the air exchange velocity of the second layer by advective transport [m y^{-1}],

The upward turbulent exchange velocities, Vel_{exch} [m y^{-1}], are calculated as the inverse of the average resistance of the two adjacent layers, which follows from the definition of resistance (see Section 8.1.4). For simplicity, the resistance of the uppermost layer (above L2) has been neglected. This simplification is justified by the fact that the resistance decreases substantially with height and, as shown in Avila and Kovalets (2014) this simplification does not lead to significant overestimation of the exchange rates. The equations used for calculating the exchange velocities are:

$$Vel_{exch,L1,L2} = \frac{1}{0.5(res_{L1} + res_{L2})} \tag{8-25}$$

$$Vel_{exch,L2,Upp} = \frac{1}{0.5 \cdot res_{L2}} \tag{8-26}$$

where

res_{L1} is the resistance of the first layer to turbulent transport [y m^{-1}], and

res_{L2} is the resistance of the second layer to turbulent transport [y m^{-1}].

The resistance to turbulent transport, *res* [y m^{-1}], of an atmospheric layer is defined in the same way as in Section 8.1.4:

$$res_{L1} = \frac{z_{L1}}{D_{L1}} \qquad \mathrm{res}_{L2} = \frac{z_{L2}}{D_{L2}} \tag{8-27}$$

where

z is the thickness of an atmospheric layer (first and second above the canopy) [m], and

D is the eddy diffusion coefficient of corresponding layer [m^2 y^{-1}],

For both layers (L1 and L2), we assume that the eddy diffusion coefficient is proportional to the height above the water. Under this assumption, the eddy diffusion coefficients, *D* [m^2 y^{-1}], for these layers become (Avila and Kovalets 2014):

$$D_{L1} = \frac{Karman_{const} \cdot Vel_{frict} \cdot z_{L1}}{ln\left(\frac{z_{L1}}{z_0}\right)}$$

$$D_{L2} = \frac{Karman_{const} \cdot Vel_{frict} \cdot z_{L2}}{ln\left(\frac{z_{L1} + z_{L2}}{z_{L1}}\right)} \tag{8-28}$$

where

$Karman_{const}$ is the von Karman constant [unitless],

Vel_{frict} is the friction velocity [m y^{-1}], and

z_0 is the roughness length [m].

The friction velocity, Vel_{fric} [m y^{-1}], is calculated from the wind speed at a reference height, assuming that the wind speed increases logarithmically with height:

$$Vel_{frict} = \frac{Karman_{const} \cdot Vel_{wind,heigh,ref,ter}}{ln\left(\frac{height_{ref,ter}}{z_0}\right)} \tag{8-29}$$

$$z_0 = 0.1 \cdot height_{CA}$$

where

$Vel_{wind,heigh,ref,ter}$ is the wind speed reference height $height_{ref,ter}$ [m y^{-1}], and

$height_{ref,ter}$ is the reference height [m].

8.2.3 Advective fluxes

The advective fluxes, Vel_{Adv} [m y^{-1}], from the L1 and L2 layers are defined and calculated in the same way as in Section 8.1.5, with the following equation:

$$\mathrm{Vel}_{\mathrm{Adv,i}} = \frac{Vel_{wind,i} \cdot z_i}{\sqrt{area}} \qquad i = \{L1, L2\} \tag{8-30}$$

where

Vel_{wind} is the average wind speed in the two atmospheric layers [m y^{-1}],

z is the thickness of the *L1* and *L2* layers [m],

area is the surface area of water body [m^2], and

i is the index for the first and second layers.

For the L1 and L2 layers, the wind speed is assumed to increase as a logarithmic function of height. Consequently, the average wind speed in these layers, Vel_{wind} [m y^{-1}] can be expressed as:

$$Vel_{wind,L1} = \frac{Vel_{frict}}{Karman_{const}}\left(\left(\ln\left(\frac{z_{L1}}{z_0}\right) - 1\right) + \frac{z_0}{z_{L1}}\right)$$

$$Vel_{wind,L2} = \frac{Vel_{frict}}{Karman_{const}}\left(\left(\frac{z_{L1}+z_{L2}}{z_{L2}}\right)\left(\ln\left(\frac{z_{L1}+z_{L2}}{z_0}\right) - 1\right) - \left(\frac{z_{L1}}{z_{L2}}\right)\left(\ln\left(\frac{z_{L1}}{z_0}\right) - 1\right)\right) \tag{8-31}$$

where

Vel_{frict}	is the friction velocity [m y^{-1}] (see Equation 8-12),
$Karman_{const}$	is the von Karman constant [unitless],
z	is the thickness of the L1 and L2 layers [m], and
z_0	is the roughness length [m].

9 Dose calculations for human inhabitants

The assessment of dose to human inhabitants resulting from a release of radionuclides from the existing and the planned repository to the surface environment involves estimating activity concentrations of radionuclides in the environment as described in previous chapters (air, soil and water), and in plants and animals that may serve as foodstuffs. Assumptions on properties and habits of the humans that will inhabit or utilise the biosphere object are required to estimate exposure from external radiation, inhalation and intakes of radionuclides in food and water (land-use variants). The dose can then be computed using dose coefficients that account for retention in the human body and exposure from progeny radionuclides (Grolander 2013).

9.1 Dose to man

Radionuclides in the environment may lead to both external and internal exposure. Following the methods of Avila and Bergström (2006), the total dose from exposure to radionuclides in the environment is the sum from exposure through ingestion of water, ingestion of food, external exposure and inhalation:

$$Dose_{total} = Dose_{ing,water} + Dose_{ing,food} + Dose_{ext} + Dose_{inh} \tag{9-1}$$

where

$Dose_{ing,water}$ is the dose contribution from drinking water [Sv y^{-1}],

$Dose_{ing,food}$ is the dose contribution from eating foodstuff [Sv y^{-1}],

$Dose_{ext}$ is the dose contribution from external exposure [Sv y^{-1}], and

$Dose_{inh}$ is the dose contribution due to inhalation [Sv y^{-1}].

The internal exposure caused by a single radionuclide from water ingestion depends on water consumption [m^3 y^{-1}] and radionuclide concentration in consumed water [Bq m^{-3}]:

$$Dose_{ing,water} = AC_{ing,water} ingRate_{water} doseCoef_{ing,water} \tag{9-2}$$

where

$AC_{ing,water}$ is the activity concentration in drinking water [Bq m^{-3}],

$ingRate_{water}$ is the ingestion rate of water per year [m^3 y^{-1}], and

$doseCoef_{ing,water}$ is the dose coefficient from ingestion of water [Sv Bq^{-1}].

Similarly, the internal exposure caused by a single radionuclide from food ingestion is the product of the activity concentration in the food [Bq kgC^{-1}] and the amount of food consumed [kgC y^{-1}], summed over all food items in the diet. The corresponding dose is calculated by multiplication with the dose conversion factor for ingestion:

$$Dose_{ing,food} = ingRate_C doseCoef_{ing,food} \sum AC_{food,i} f_{cont,food,i}, \qquad \sum f_{cont,food,i} \leq 1 \tag{9-3}$$

where

$ingRate_C$ is the yearly energy demand for an adult expressed in units of carbon [kgC y^{-1}],

$AC_{food,i}$ is the activity concentration in food item i [Bq kgC^{-1}],

$f_{cont,food,i}$ is the fraction of food item i in the diet [kgC kgC^{-1}], and

$doseCoef_{ing,food}$ is the dose coefficient from ingestion of food [Sv Bq^{-1}].

External exposure from the environment is calculated from the activity concentration in environmental media [e.g. Bq kg_{DW}^{-1}] and the time spent in contact with the media [h y^{-1}]. In this safety assessment, only exposure from the soil is considered, as this is the medium expected to be associated with both high activity concentrations and large exposure times, as compared with direct radiation from water , air, aquatic sediments or vegetation (SKB 2014a, e). The dose from direct radiation from the ground can thus be calculated by considering the exposure from a soil layer by multiplication with the appropriate dose coefficient:

$$Dose_{ext} = AC_{ground} dens_{ground}\, t_{exposure}\, doseCoef_{ext} \tag{9-4}$$

where

AC_{ground} is the activity concentration of the surface soil [Bq kg_{DW}^{-1}],

$dens_{ground}$ is the density of the top soil [kg_{DW} m^{-3}]

$t_{exposure}$ is the time spent in contact with the ground [h y^{-1}], and

$doseCoef_{ext}$ is the dose coefficient from external exposure [(Sv h^{-1})/(Bq m^{-3})].

The internal exposure from inhalation of air depends on the activity concentration in air from particles and radionuclides in gaseous form [Bq m^{-3}], the inhalation rate [m^3 h^{-1}] and the time spent in contact with air containing radionuclides [h y^{-1}]. The dose from inhalation can then be calculated by multiplying exposure with an appropriate dose coefficient:

$$Dose_{inh} = AC_{air} inhRate\, t_{exposure}\, doseCoef_{inh} \tag{9-5}$$

where

AC_{air} is the activity concentration in air [Bq m^{-3}],

$inhRate$ is the human inhalation rate of volume air [m^3 h^{-1}], and

$doseCoef_{inh}$ is the dose coefficient from inhalation [Sv Bq^{-1}].

Both, radionuclides in gaseous form (C-14) and radionuclides adsorbed to dust particles (other radionuclides) contribute to the activity concentration of air:

$$\begin{aligned} AC_{air}^{RN} &= AC_{ground}^{RN} conc_{Dust} \\ AC_{air}^{14C} &= AC_{atmos,L1}^{14C} + AC_{ground}^{14C}\, conc_{Dust} \end{aligned} \tag{9-6}$$

where

AC_{ground} is the activity concentration in the top soil [Bq kg_{DW}^{-1}]

$conc_{Dust}$ is the concentration of fine soil particles in the inhaled air [kg_{DW} m^{-3}], and

$AC_{atmos,L1}^{14C}$ is the activity concentration of C-14 in gaseous form in the inhaled air [Bq m^{-3}].

In land-use variants involving cultivation simplified ecosystem models are used (see Chapter 7). The solutions to the differential equations describing radionuclide accumulation in cultivated soils are relatively simple, and consequently algebraic expressions were used to calculate soil concentrations (without running dynamic simulations). However, the derived soil concentrations do not account for ingrowth of radionuclides. Instead the contribution of ingrowth to dose is accounted for by substituting the dose coefficients in Equations 9-3–9-5 by aggregated dose coefficients, $doesCoef_{i,agg}$ [Sv Bq^{-1}], that accounted for exposure from accumulated daughter radionuclides:

$$doesCoef_{i,agg} = doseCoeff_i \cdot doseIngrowth_{i,agri} \tag{9-7}$$

where

$doseIngrowth_{i,agri}$ is a scaling factor that accounts for exposure from progeny radionuclides accumulating in cultivated soil during 50 years [Bq Bq^{-1}] (Grolander 2013).

i indicate route of exposure, i.e. external, ingestion and inhalation.

9.2 Exposed populations and land-use variants

For the purpose of protection, the potential exposure is evaluated for a representative person from the more highly exposed individuals in the population (i.e. the most exposed group) (ICRP 2006). Due to the long time-scales considered for long-lived solid radioactive waste, the habits and characteristics of future inhabitants can only be based on a number of assumptions (cf. ICRP 1998).

Below we describe how the dose is calculated for four exposed populations, each corresponding to a unique land-use variant, namely hunter-gatherers, infield-outland agriculture, draining and cultivating a lake-mire complex, and small scale horticulture (Section 2.4.1). The first three land-use variants were chosen to serve as limiting cases with respect to exposure from groundwater discharge when ingestion of food is the primary source for exposure (Saetre et al. 2013, SKB 2014e). The forth variant was set up as a limiting case with respect to exposure from a well in bedrock, but was also used to quantify consequences of using seaweed for fertilisation and biofuel combustion for heating. The full analysis of exposure pathways is presented in SKB (2014e), and is summarised in SKB (2014a) (see also Section 2.4). An overview of exposure pathways included in the assessment is presented in Figure 9-1 and Table 9-1.

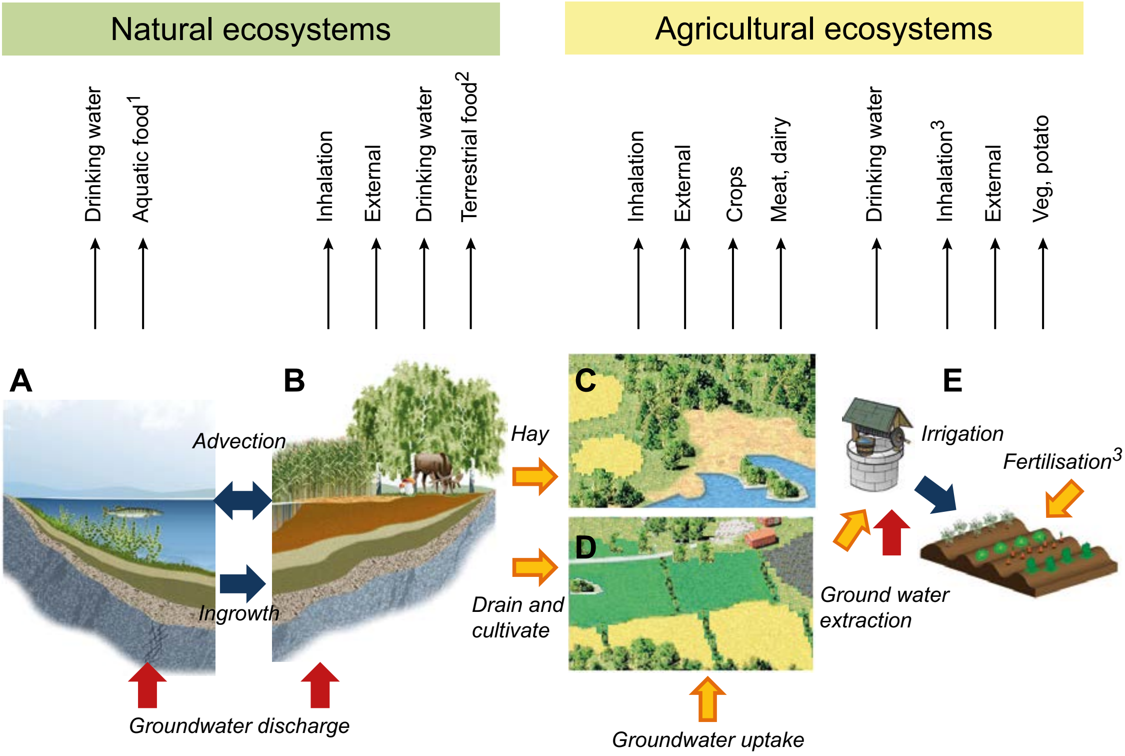

Figure 9-1. *Exposure pathways included in the dose calculations for exposed populations using natural resources and/or living in biosphere objects. Hunter-gatherers use natural aquatic (A) and mire (B) ecosystems, whereas the other three exposed populations represented different uses of arable land, namely infield-outland agriculture (C), draining and cultivating the mire (D) and small scale horticulture on a garden plot (E). Bold arrows represent input of radionuclides from the bedrock (red), from natural ecosystems or deep regolith deposits (orange), or water-bound transfer of radionuclides within the biosphere (blue). The thin arrows (top) represent exposure routes. 1 = fish and crayfish, 2 = game, berries and mushroom, 3 = inhalation and fertilisation includes radionuclides from combustion of biofuel.*

Table 9-1. Exposure pathways from source to potentially exposed population. Pathways have been organised with respect to the surface ecosystem in which the exposure point occurs or originates. Exposed populations (or land-use variants) are abbreviated: HG = hunting and gathering, IO = infield-outland farming, DM = draining and cultivating mire and GP = cultivating a garden plot.

Ecosyst.	Source		Exposure point[1]	Exposure route				Exposed popul.
	Type	From	(environmental medium)	Ingestion food	Ingestion water	Inhalation[2]	External irradiation	(land use)
Sea	Groundwater discharge or surface water	Bedrock or nearby basin	Sea water	Fish				HG
Lake/ stream	Groundwater discharge or surface water	Bedrock or upstream area	Lake/stream water	Fish & crayfish	Drinking water			HG
				Livestock[3]				IO, DM, GP
Mire	Groundwater discharge or surface water	Bedrock or upstream area	Surface peat	Game, berries & mushroom		Gas/ dust particles	Surface peat	HG
								IO
	Dug well	Soil ground-water	Deep regolith (Till)	Livestock[3]	Drinking water			IO, DM. GP
	Hay making	Surface peat	Mire vegetation	Livestock				IO
	Biofuel burning	Mire vegeta-tion or peat	Atmosphere[5]			Gas[4]/ dust particles		GP
Agricult.	Cultivation of mire	Regolith (Mire)	Cultivated soil	Cereal, pota-toes, livestock		Gas[5]/ dust particles	Cultivated soil	DM
	Groundwater uptake	Deep regolith (Mire)						
	Fertilization	Hay making (Mire)		Cereal				IO
		Biofuel ash (Mire)		Vegetables, potatoes				GP
		Seaweed						
		(Sea)						
	Irrigation	Contaminated water[6]						
Bedrock	Drilled well	Bedrock	Deep ground-water	Livestock[3]	Drinking water			DM, GP

1 = Endpoint of transport calculations.
2 = Exposure point (environmental medium) for inhalation is atmosphere above listed media
3 = Drinking water for livestock not included in the GP land use variant
4 = Atmosphere down-wind household
5 = Volatile radionuclides
6 = Surface water, or water from dug/drilled well.

9.3 Hunter-gatherers

The future landscape in Forsmark under weak human influence will most likely be a mosaic of forests, wetlands and lakes, with a decreasing influence by the sea as the shoreline transgress with time (Chapter 5 in SKB 2014a). The most exposed inhabitants would be the ones making full use of the areas affected by the discharge of deep groundwater from the repository. Therefore, if radionuclides were discharged to the sea, the most exposed group would be one with a marine diet, whereas if groundwater from the repository primarily reached a lake with surrounding wetland, a community living in proximity to, and foraging from, the wetland-lake ecosystem would be most exposed to radionuclides.

From the historical record of hunter-gatherers it is reasonable to assign a social structure that encompasses 30–40 members, and a corresponding local group area of 200 km^2 also to future foragers (see SKB 2014e and Saetre et al. 2013 for details). With these assumptions, most of the central investigation area of Forsmark (including all potential discharge areas for groundwater from the repository) would be within the home range of practically any settlement on the coast.

For hunter-gatherers, streams and lakes are the most likely source of drinking water (Werner et al. 2014), and consequently surface water is used to calculate the internal dose from ingestion of water. Note that seawater is not considered to be consumed as drinking water.

$$AC_{ing,water} = AC_{water,surface} \mid time > threshold_{isolation} \qquad (9\text{-}8)$$

where

$AC_{water,surface}$ is the activity concentration in surface water [Bq m^{-3}], and

$threshold_{isolation}$ is the point in time when the lake is isolated from the sea or when the biosphere object is completely dry at normal sea level.

Activity in drinking water includes radionuclides in solution, adsorbed on particles and stored in organic particulate matter. The yearly recharge from each biosphere object is sufficient to cover the demands of drinking water for a hunter-gatherer group (Werner et al. 2014). If the biosphere object does not have any lake or stream water, the peat pore water is used as a source of drinking water in the assessment (objects 121_2 and 157_2). This assumption is cautious from a water-supply perspective as, diffuse overland-water is difficult to collect and there will be other sources of water that are more appealing for consumption in nearby streams (Werner et al. 2014). The activity concentration in surface water is defined as:

$$AC_{water,surface} = \begin{cases} AC^{D}_{regoUp,ter}, & if\ object\ in\ \{121\_2, 157\_2\} \\ AC_{water} + AC^{PM}_{water,org}, & otherwise \end{cases} \qquad (9\text{-}9)$$

where

$AC^{D}_{regoUp,ter}$ is the activity concentration in the pore water of surface peat [Bq m^{-3}],

AC_{water} is the concentration of radionuclides in inorganic form in open water [Bq m^{-3}]. It includes radionuclides in solution and adsorbed on particulate matter.

$AC^{PM}_{water,org}$ is the activity concentration of radionuclides in suspended organic particulate matter [Bq m^{-3}]

As hunter-gatherers are expected to only have a weak influence on the landscape, external exposure from the ground is expected primarily during foraging in wetland ecosystems or from camp sites in the proximity to lake-wetland complexes. Therefore, the total activity concentration in surface peat, $AC^{PM}_{regoUp,tot,ter}$ [Bq kg$_{DW}^{-1}$], is used as the source for external exposure:

$$AC_{regoUp,tot,ter} = AC_{regoUp,ter} + AC_{regoUp,org,ter} \qquad (9\text{-}10)$$

where

$AC_{regoUp,ter}$ is the activity concentration in inorganic form in surface peat [Bq kg$_{DW}^{-1}$], and

$AC_{regoUp,org,ter}$ is the corresponding activity concentration in organic matter [Bq kg$_{DW}^{-1}$].

Similarly, exposure from inhalation of radionuclides in air is expected to occur during foraging in wetland ecosystems, or from camp sites in proximity to lake-wetland complexes. Thus, the activity concentrations in air and suspended dust above wetland vegetation are used to calculate the source for inhalation exposure:

$$AC_{air}^{RN} = AC_{regoUp,tot,ter}^{RN} conc_{Dust,ter}$$

$$AC_{air}^{14C} = AC_{atmos,ter,L1}^{14C} + AC_{ground}^{14C} conc_{Dust,ter} \tag{9-11}$$

where

$conc_{Dust,ter}$ is the concentration of fine soil particles in air above wetland vegetation [kg_{DW} m^{-3}],

AC_{ground}^{14C} is the activity concentration of C-14 in gaseous form in the air above the wetland canopy layer [Bq m^{-3}], and

$AC_{atmos,ter,L1}^{14C}$ is the activity concentration of C-14 in the atmosphere above the canopy of the field layer [Bq m^{-3}].

There are many factors affecting foraging behaviour in a patchy landscape. For example, potential landscape production, the distance to, and the harvesting/capturing time associated with, available food items. Moreover, storage efficiency will modify the yield, and cultural preferences and nutritional needs will ultimately determine the diet (Hassan 1975). A simple way to estimate an upper limit for consumption of food items containing radionuclides is to assume that the most exposed group consumes all food items they can reasonably harvest from the area affected by radionuclides from the repository (given constraints on human demand for energy and nutrients).

The production capacity of the biosphere object with the highest concentrations of radionuclides is typically not sufficient to cover the energy demand for a band of hunter-gatherers (Figure 9-2). The fraction of food from the object can therefore be calculated as the ratio of the sustainable production capacity of the area with respect to a specific food item [kgC y^{-1}] divided by the yearly energy demand expressed in carbon units [kgC y^{-1}]:

$$f_{cont,i} = \frac{area_{obj,aqu} prod_{edib,i}}{N_{group} ingRate_C}, \qquad i = \{crayfish, fish\}$$

$$f_{cont,j} = \frac{area_{obj,ter} prod_{edib,j}}{N_{group} ingRate_C}, \qquad j = \{berry, game, mush\} \tag{9-12}$$

where

$area_{obj,aqu}$ is the surface area of the aquatic part of the biosphere object [m^2] including the whole sea basin during submerged period,

$area_{obj,ter}$ is the surface area of the wetland/mire in the biosphere object [m^2],

$prod_{edib,i}$ is the area specific sustainable yield with respect to food item i [kgC m^{-2} y^{-1}].

N_{group} is the number of individuals in a hunter-gatherer group [–], and

$ingRate_C$ is the annual demand of energy (expressed in units of carbon) [kgC y^{-1}]

When the biosphere object is a large body of water (e.g. a sea basin) the production capacity for fish may cover a large fraction or even all of the most exposed group's demand of energy. However, as excessive protein consumption may lead to toxic effects, (causing nausea and diarrhea, Bilsborough and Mann 2006), there is an upper boundary for safe fish consumption (Saetre et al. 2013). Thus, the upper limit of safe consumption of protein has to be factored in when calculating the fraction of radionuclide containing fish in the diet:

$$f_{cont,fish} = \min\left(\frac{area_{obj,aqu} prod_{edib,fish}}{N_{group} ingRate_C}, f_{diet,fish,max}\right) \tag{9-13}$$

where

$f_{diet,fish,max}$ is the fraction of fish corresponding to the maximum healthy protein consumption.

If the production of fish of several biosphere objects exceeds the protein demand of the most exposed group, it is cautiously assumed that fish from the object with the highest radionuclide concentration is consumed first. Thereafter, the diet is complemented with fish from the object with the second highest concentration of radionuclides, and so on, until the total intake of fish equals the upper limit of safe protein consumption (see Figure 9-2).

Through adsorption of radionuclides (to e.g. gills) and consumption of plant material, the activity concentration in aquatic fauna is expected to reach equilibrium with the activity concentrations in the water (Tröjbom et al. 2013):

$$AC_i^{RN} = AC_{water}^{DRN}\, CR_{aqu,i},\ i = \{fish, cray\} \tag{9-14}$$

where

AC_{water}^{DRN} is the concentration of a dissolved radionuclide in the water [Bq m^{-3}], and

$CR_{aqu,i}$ is equilibrium concentration ratios for fish and crayfish [m^3 kgC^{-1}].

Similarly, in mire ecosystems the activity concentrations in plants and fungi (including berries and fruit bodies) are assumed to reach equilibrium with concentrations in the upper peat layer where nutrients and trace elements are taken up (Tröjbom et al. 2013). Moreover the activity concentrations in herbivores are assumed to be in equilibrium with the activity concentrations in their diet. Thus, the activity concentrations in food items can be calculated as follows:

$$\begin{aligned}
AC_{berry}^{RN} &= AC_{PP,ter}^{RN} \\
AC_{mush}^{RN} &= \left(AC_{regoUp,ter}^{RN} + AC_{regoUp,org,ter}^{RN}\right) CR_{ter,mush} \\
AC_{herbiv}^{RN} &= AC_{diet,herbiv}^{RN} CR_{food,herbiv} \\
AC_{diet,herbiv}^{RN} &= f_{mush,herbiv} AC_{mush}^{RN} + (1 - f_{mush,herbiv}) AC_{PP,ter}^{RN}
\end{aligned} \tag{9-15}$$

where

AC_i^{RN} is the activity concentration per unit biomass carbon for mire vegetation, and human food items (berries, mushrooms and herbivores) [Bq kgC^{-1}],

$AC_{regoUp,ter}^{RN}$ is the inorganic activity concentration in the upper peat layer [Bq kg_{DW}^{-1}],

$AC_{regoUp,org,ter}^{RN}$ is the organic activity concentration in the upper peat layer [Bq kg_{DW}^{-1}],

$CR_{ter,mush}$ is the equilibrium concentration ratio between edible mushrooms and soil [kg_{DW} kgC^{-1}],

$CR_{food,herbiv}$ is the equilibrium concentration ratio between herbivores and their diet [kgC kgC^{-1}], and

$f_{mush,herbiv}$ is the fraction of mushrooms in the diet of herbivores [kgC kgC^{-1}].

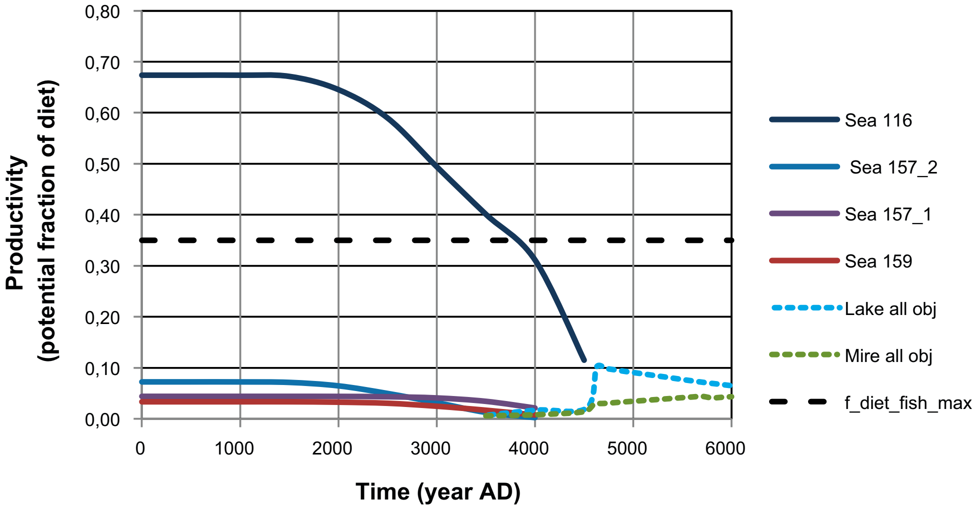

Figure 9-2. *Productivity of natural food items in biosphere objects expressed relative to the annual energy demand of a group of hunter-gatherers (N=30). During the submerged period fish is the only food item considered. During this period, the productivity of fish exceeds the upper level of safe protein consumption for object 116 only. During the land period the combined productivity of lakes and mires in all biosphere objects (116, 121_1, 121_2, 157_1, 157_2, 159, and 160) satisfies less than 20% of the annual energy demand of a hunter-gatherer group.*

As the discrimination of carbon species is expected to be small in synthesis and metabolism of carbon (Peterson 1980), the ratio of radioactive (C-14) to stable carbon (primarily C-12) is expected to be similar among primary producers and to be propagated throughout the food chains (Avila and Pröhl 2008). Ignoring dilution from consumption of organic carbon outside or entering the biosphere object, the specific activity in local primary producers sets the upper boundary for the specific activity in consumers (including detritivores).

If the contribution of carbon from primary producers to higher trophic levels is proportional to the net production of carbon (NPP), the specific C-14 activity in aquatic organisms (which may serve as food items for human inhabitants), can be expressed as a weighted average of macrobethic, microbentic and pelagic primary producer activity concentrations:

$$AC_{fish}^{14C} = AC_{cray}^{14C} = AC_{PP,aqu}^{14C} = \frac{\sum_i AC_{biom,i}^{14C} NPP_i}{\sum_i NPP_i}$$

$$i = \{plank,\ micro,\ macro\} \qquad (9\text{-}16)$$

where

$AC_{biom,i}^{14C}$ is the activity concentration per unit carbon for community i [Bq kgC^{-1}], and

NPP_i is the net primary production of community *i* in water per unit surface area [kgC m^{-2} y^{-1}].

Note that as radionuclide concentrations in organisms are expressed per unit carbon, the activity concentration equals the specific activity for all biota. The C-14 activity concentration in edible parts of terrestrial primary producers and in saprophytic and/or mycorhizza-forming fungi (that depend on terrestrial primary production for carbon) can all be approximated with the specific activity concentration in wetland primary producers:

$$AC_{berry}^{14C} = AC_{mush}^{14C} = AC_{herbiv}^{14C} = AC_{diet,herbiv}^{14C} = AC_{PP,ter}^{14C} \qquad (9\text{-}17)$$

where

$AC_{PP,ter}^{14C}$ is the activity concentration of carbon 14 in terrestrial primary producers per unit biomass carbon [Bq kgC^{-1}].

Assuming that the tritium (H-3) concentration is primarily in the form of heavy water in living tissue, the activity concentration in fish (Bq kg^{-1}) is described as the product of the liquid tritium concentration in fish (Bq kgH_2O^{-1}) and the liquid to carbon ratio in fish (kgH_2O kgC^{-1}):

$$AC_{fish}^{3H} = AC_{fish,wet}^{3H} f_{fish,wet}, \qquad (9\text{-}18)$$

where

$AC_{fish,wet}^{3H}$ is activity concentration in the liquid part of the fish [Bq $kg_{H_2O}^{-1}$], and

$f_{fish,wet}$ is the ratio of liquid to carbon in fish [kg_{H_2O} kgC^{-1}].

Moreover, assuming there is no discrimination with respect to H-3 when body fluids are formed from water in fish, the tritium activity concentration in the liquid part of the fish (Bq kg_{H2O}^{-1}) may be approximated as the activity concentration in water:

$$AC_{fish,wet}^{3H} = \frac{AC_{water}^{3H}}{dens_{water}}, \qquad (9\text{-}19)$$

where

AC_{water}^{3H} is activity concentration in the water [Bq m^{-3}], and

$dens_{water}$ is the density of water [kg_{H_2O} m^{-3}].

The ratio of liquid to carbon in fish can be described as the quotient of the fraction of liquid in the fresh weight of the fish (kg_{H_2O} $kgFW^{-1}$), i.e. the part that is not dry weight, and the fraction of carbon in fresh weight (kgC $kgFW^{-1}$):

$$f_{fish,wet} = \frac{1 - f_{DW,FW,fish}}{f_{C,fish} \cdot f_{DW,FW,fish}}, \qquad (9\text{-}20)$$

where

$f_{DW,FW,fish}$ is dry weight to fresh weight ratio [$\mathrm{kg_{DW}\ kg_{FW}^{-1}}$], and

$f_{C,fish}$ is carbon to dry weight ratio in fish [$\mathrm{kgC\ kg_{DW}^{-1}}$].

This simple approach is equivalent to calculating biomass activity concentration based on the specific activity of H-3 in the body fluids of fish. However it does not account for H-3 activity in dry tissue which under equilibrium conditions may contribute to approximately one sixth of the total activity concentration[6].

9.4 Self-sustained agriculture

Once the biosphere object has emerged out of the sea it develops into a lake-mire complex or directly into a mire, where harvesting of wetland grasses and sedges provide an exposure pathway for human settlers. Sustainable cultivation of peat is not possible in the young lake-mire or mire system. However when the discharge area has emerged sufficiently above the sea level to make salt water intrusions unlikely, draining and cultivation of peat is considered a relevant exposure pathway.

Saetre et al. (2013) suggested that a self-sustained Iron Age agricultural community of two households could be used to represent the most exposed group with respect to exposure from wetland hay. In an Iron Age infield-outland agricultural system, hay would be collected from wetlands to be used as winter fodder, and radionuclides would accumulate in cultivated soil as a result of fertilisation with manure. An upper boundary for exposure is to assume that the most exposed group fully depends on the wetland in the biosphere object with the highest concentration of radionuclides for winter fodder.

Moreover the same authors (Saetre et al. 2013) suggested that two small self-sustained family farms from the industrial-age would serve well as a model for the most exposed group with respect to cultivation of a wetland. Although peat soils were traditionally used for fodder production, cereals (oats, barley and, in particular, rye) and potatoes also grow well in organic soils. Consequently, it is assumed that the most exposed group uses organic soils for production of all human crops and fodder in this land-use variant.

9.4.1 Activity concentrations of water, soil and air

Water from a dug well, a drilled well or from a nearby lake or a stream are all plausible sources for drinking water in the drained mire land-use variant. The infield-outland agriculture has historically been associated with less sophisticated technology, and a drilled well was not considered as a source of drinking water for this land-use variant (Werner et al. 2014). For each group, the available water resource with the highest concentration of radionuclides was used to calculate the internal dose from ingestion of water:

$$AC_{ing,water,IO} = \max\left(AC_{water,well,dug} \mid time > threshold_{well}, AC_{water,surface} \mid time > threshold_{isolation}\right) \qquad (9\text{-}21)$$

$$AC_{ing,water,DM} = \max\left(max\left(AC_{water,well,dug}, AC_{water,well,drilled}\right) \mid time > threshold_{well},\ AC_{water,surface} \mid \mathrm{time} > threshold_{stop}\right) \qquad (9\text{-}22)$$

where

$AC_{water,well,dug}$ is the activity concentration of dissolved radionuclides in a well dug into the regolith layers of a biosphere object [$\mathrm{Bq\ m^{-3}}$],

$AC_{water,surface}$ is the activity concentration in surface water (lake or stream) [$\mathrm{Bq\ m^{-3}}$],

$threshold_{isolation}$ is the point in time when the lake is isolated from the sea or when the biosphere object is completely dry at normal sea level [y],

$threshold_{well}$ is the point in time when it is feasible to construct a well [y], and

$threshold_{stop}$ is the point in time when it is possible to drain and cultivate the mire [y].

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[6] As white fish biomass primarily is made up of proteins, a H to C ratio for dry tissue can be estimated to ~0.16 from common amino acids. With fractions of dry weight to fresh weight, and carbon to dry weight, of 0.21 and 0.44, respectively (Grolander 2013), dry mass will contribute to approximately 16% of total fresh weight H content.

The activity concentration in a dug well can be approximated as that of the pore water in till (terrestrial RegoLow) (Werner et al. 2014):

$$AC_{water,well,dug} = AC^{D}_{regoLow,ter}, \; if\ time > threshold_{well} \tag{9-23}$$

The activity concentration in a drilled well is defined as the sum of release from the repository [Bq y^{-1}], scaled by the fraction reaching the well [unitless], and divided by the water extraction rate [m^3 y^{-1}]:

$$AC_{water,well,drilled} = \frac{\sum_i f_{well,agri,i} Release_i}{q_{well}}, \qquad if\ time > \ threshold_{well} \tag{9-24}$$

where

$Release_i$	is the release of radionuclides to biosphere-geosphere interface [Bq y^{-1}],
$f_{well,agri,i}$	is the fraction of the release going to the well [Bq y^{-1} per Bq y^{-1}],
q_{well}	is the rate of water extraction from a well [m^3 y^{-1}], and
i	is the index for the waste vaults *Silo*, *BRT*, *1BTF*, *2BTF*, *1BMA*, *2BMA*, *1BLA*, *2BLA*, *3BLA*, *4BLA* and *5BLA*.

External exposure from the ground is primarily expected from working on arable land, but may also occur from collecting hay from the wetland in a biosphere object. Thus, the activity concentrations in the agricultural soil and wetland peat are used as the sources for external exposure in the infield-outland land-use variant. The fractions of time spent in the two environments are used to weight the radionuclide concentrations (and the external dose is calculated by summing the contributions of the two sources).

$$\begin{aligned} AC_{ground,IO,agri} &= f_{time,agri}\,(AC_{regoUp,IO} + AC_{regoUp,org,IO}) \\ AC_{ground,IO,mire} &= f_{time,hay}\,(AC_{regoUp,ter} + AC_{regoUp,org,ter}) \end{aligned} \tag{9-25}$$

where

$AC_{regoUp,org,IO}$	are the activity concentrations in cultivated soil [Bq kg_{DW}^{-1}],
$AC_{regoUp,org,ter}$	are the activity concentrations in surface peat [Bq kg_{DW}^{-1}],
$f_{time,agri}$	is the fraction of time spent working on arable land [y y^{-1}], and
$f_{time,hay}$	is the fraction of time spent collecting wetland hay [y y^{-1}].

For agriculture on a drained mire, it is assumed that the time spent working with each crop is proportional to the surface area of the fields. The weighted average of radionuclide concentration in soil associated with different crops is therefore used to calculate the soil concentration [Bq kg_{DW}^{-1}] causing external exposure:

$$AC_{ground,DM} = \sum_i f_{area,i}\left(AC_{regoUp,DM,i} + AC_{regoUp,org,DM,i}\right), \qquad i = \{cereal, tuber, fodder\} \tag{9-26}$$

where

$f_{area,i}$	is the fraction of productive arable land (non-fallow) that is used for crop i [m^2 m^{-2}],
$AC_{regoUp,DM,i}$	is the activity concentration in inorganic form in cultivated peat soils [Bq kg_{DW}^{-1}]
$AC_{regoUp,org,DM,i}$	is the corresponding concentration in organic form [Bq kg_{DW}^{-1}], and
i	is the type of crop cultivated.

Exposure from inhalation of air is expected in the same environments as the ones resulting in direct external exposure (see above). The activity concentrations in the air above agricultural soil and wetland [Bq m^{-3}] are used as the sources for inhalation exposure in the infield-outland land use variant, and the total dose is calculated by summing the contributions from the two environments.

$$\begin{aligned}
AC_{air,IO,agri}^{RN} &= f_{time,agri}\left(AC_{regoUp,IO}^{RN} + AC_{regoUp,org,IO}^{RN}\right)conc_{dust,IO} \\
AC_{air,IO,mire}^{RN} &= f_{time,hay}(AC_{regoUp,org,ter}^{RN} + AC_{regoUp,ter}^{RN})conc_{dust,ter} \\
AC_{air,IO,agri}^{14C} &= f_{time,agri}\left(\left(AC_{regoUp,IO}^{14C} + AC_{regoUp,org,IO}^{14C}\right)conc_{dust,IO} + AC_{atmos,L1,\mathrm{IO}}^{14C}\right) \\
AC_{air,IO,mire}^{14C} &= f_{time,hay}((AC_{regoUp,ter}^{14C} + AC_{regoUp,org,ter}^{14C})conc_{dust,ter} + AC_{atmos,L1,\mathrm{ter}}^{14C})
\end{aligned} \tag{9-27}$$

where

$conc_{dust,IO}$ is the concentration of fine soil particles in air above cereal field [$\mathrm{kg_{DW}\ m^{-3}}$],

$conc_{dust,ter}$ is concentration of fine soil particles in air above wetland vegetation [$\mathrm{kg_{DW}\ m^{-3}}$],

$AC_{atmos,L1,ter}^{14C}$ is the activity concentration in air above the mire field layer [$\mathrm{Bq\ m^{-3}}$], and

$AC_{atmos,L1,IO}^{14C}$ is the activity concentration in air above crop canopy [$\mathrm{Bq\ m^{-3}}$].

Similarly, air activity concentrations for the drained mire land-use variant are described as:

$$AC_{air,DM}^{RN} = AC_{ground,DM}^{RN}\, conc_{dust,DM}, \tag{9-28}$$

$$AC_{air,DM}^{14C} = \sum_i f_{area,i}\left(AC_{ground,DM,i}^{14C} conc_{dust,DM,i} + AC_{atmos,L1,DM,i}^{14C}\right), \quad i = \{cereal, potato, fodder\}$$

where

$conc_{dust,DM,i}$ is the concentration of fine soil particles in air above cultivated peat soils [$\mathrm{kg_{DW}\ m^{-3}}$], and

$AC_{atmos,L1,DM,i}^{14C}$ is the activity concentration in air above the canopy of crop i [$\mathrm{Bq\ m^{-3}}$].

9.4.2 Activity concentration in food from arable land and potential dilution

Assuming that agriculture supplies the bulk of the diet for a farm household, typical dietary fractions of food items can be calculated from land use and productivity (Saetre et al. 2013, Grolander 2013). For infield-outland agriculture *all* food items are assumed to contain radionuclides. However, the supply of hay from the biosphere object may affect the activity concentrations of winter fodder and organic fertilisers (see below).

In the case of cultivating a drained mire, the production from the biosphere object is likely to have the capacity to support a few farming households. However, if the area is not sufficiently large to support the energy demand of the most exposed group, then food items outside the biosphere object will also be consumed. Also, the time spent on land outside the biosphere object can be taken into account, reducing the external exposure as well as exposure due to inhalation radionuclides. This potential dilution can be described by modifying the total dose equation (Equation 9-1) with a coefficient α:

$$Dose_{total} = Dose_{ing,water} + \alpha_{drained,mire}\,(Dose_{ing,food} + Dose_{ext} + Dose_{inh}) \tag{9-29}$$

where

$\alpha_{drained,mire}$ is a dilution coefficient that takes the value between 1 (all ingested food originates from the biosphere object and 0 (no food from the biosphere object is consumed):

$$\alpha_{drained,mire} = \begin{cases} 1, & \text{when } area_{obj,ter} \geq N_{group}area_{support} \\ \dfrac{area_{obj,ter}}{N_{group}area_{support}}, & \text{otherwise} \end{cases} \tag{9-30}$$

where

$area_{obj,ter}$ is the area of the wetland in the biosphere object [$\mathrm{m^2}$], and

$area_{support}$ is the area required to support the energy demand of one individual [$\mathrm{m^2}$], and

N_{group} is the number of individuals in the most exposed group.

As the discrimination of C-14 is expected to be limited during photosynthesis, the specific activity of C-14 will reflect the specific activity of inorganic carbon (CO_2) in the canopy atmosphere and in soil pore water. Therefore, the carbon specific C-14 activity in crop [$\mathrm{Bq\ kgC^{-1}}$] is approximated with a pathway (canopy vs. root) weighted average of specific activity in environmental media. The specific activity in cereal for infield-outland agriculture is then:

$$AC_{cereal,IO}^{14C} = \left(1 - f_{rootUptake}\right) SA_{atmos,CA,IO}^{14C} + f_{rootUptake} \frac{AC_{regoUp,IO}^{D14C}}{conc_{DIC,regoUp,IO}} \tag{9-31}$$

where

$f_{rootUptake}$	is the fraction of carbon in biomass that originates from root uptake (as opposed to uptake from the atmosphere) [kgC kgC^{-1}].
$SA_{atmos,CA,IO}^{14C}$	is the specific activity concentration in cereal canopy air [Bq kgC^{-1}],
$AC_{regoUp,IO}^{D14C}$	is the activity concentration of dissolved C-14 in soil pore water [Bq m^{-3}], and
$conc_{DIC,regoUp,IO}$	is the concentration of dissolved inorganic carbon in soil pore water [kgC m^{-3}].

Similarly, the specific activity in crop type *i* in for the drained and cultivated mire is:

$$AC_{i,DM}^{14C} = \left(1 - f_{rootUptake}\right) SA_{atmos,CA,DM,i}^{14C} + \left(f_{rootUptake}\right) \frac{AC_{regoUp,DM,i}^{D14C}}{conc_{DIC,regoUp,DM}}, \quad i = \{cereal, tuber, fodder\} \tag{9-32}$$

Accumulation of other radionuclides in edible parts of a crop is primarily a result of root uptake and plant allocation. Plant uptake of radionuclides can be active or passively follow water mass flow; the uptake rate will depend on the activity in soil, availability to plants, transpiration rate, root distribution and various processes in the root-rhizosphere interface. For the safety assessment, it is assumed that on the time scale of yearly yield the activity concentration in plant tissue will be in equilibrium with that in the biologically active part of soil, and this equilibrium can be described by a constant for a given cultivation system (Tröjbom et al. 2013).

For infield-outland scenario the concentration of radionuclides (other than C-14) in crops [Bq kgC^{-1}] is simply calculated by the product of the total activity concentration in soil and an element specific concentration ratio:

$$AC_{cereal,IO}^{RN} = (AC_{regoUp,IO} + AC_{regoUp,org,IO})\ CR_{agri,cereal} \tag{9-33}$$

where

$AC_{regoUp,IO}$	is the activity concentration in inorganic form in cultivated soils [Bq kg$_{DW}^{-1}$]
$AC_{regoUp,org,IO}$	is the corresponding concentration in organic form [Bq kg$_{DW}^{-1}$], and
$CR_{agri,cereal}$	is the concentration ratio between soils and cereal [kg$_{DW}$ kgC^{-1}].

Similarily, the activity concentration of crops and fodder in the drained mire land-use variant is described using:

$$AC_{i,DM}^{RN} = (AC_{regoUp,DM,i} + AC_{regoUp,org,DM,i})\ CR_{agri,i},\ i = \{cereal, tuber, fodder\} \tag{9-34}$$

Livestock that graze in biosphere objects (or eat hay from biosphere objects) and/or drink surface water from a biosphere object will be exposed to radionuclides, which accumulate in body tissues. The degree of accumulation depends on the activity in ingested water, fodder and soil, and to what degree that activity is transferred to tissues and milk. As the discrimination of carbon isotopes is (assumed to be) marginal in the synthesis of biomass or milk, the specific C-14 activity in meat and milk (including dairy products) at steady state can be approximated from that in ingested organic matter (i.e. fodder). For other radionuclides, the activity in water may also contribute to exposure, and the degree of long-term bioaccumulation in meat and milk can be described by transfer coefficients.

In the infield-outland agriculture, where the source of radionuclides is winter fodder, the C-14 activity concentration in meat and milk [Bq kgC^{-1}] is a function of the specific activity in hay:

$$AC_{meat,IO}^{14C} = AC_{milk,IO}^{14C} = AC_{hay}^{14C}\, \alpha_{hay} f_{meadow}\ and\ \{AC_{hay}^{14C} = AC_{PP,ter}^{14C}\}, \tag{9-35}$$

where

AC_{hay}^{14C}	is the specific activity in hay (or primary producers) from the mire [Bq kgC^{-1}]

α_{hay}	is the contribution of hay from the biosphere object relative the total demand for winter fodder [(kgC y^{-1})/(kgC y^{-1})] (Equation 7-6), and
f_{meadow}	is the contribution of hay (winter fodder) to the yearly fodder demand.

The corresponding expression for the activity concentration of other radionuclides [Bq kgC^{-1}] includes exposure from drinking water in addition to that from hay:

$$AC^{RN}_{i,IO} = \left(AC_{hay}\alpha_{hay}\, f_{meadow}\, ingRate_{C,cattle} + AC^{RN}_{ing,water,cattle}\, ingRate_{water,cattle}\right)\frac{TC_i}{conc_{C,i}}, \qquad (9\text{-}36)$$
$$i = \{meat, milk\}$$

where

$AC^{RN}_{ing,water,cattle}$	is the activity concentration in drinking water for livestock [Bq m^{-3}],
$ingRate_{C,cattle}$	is the daily energy demand for cattle expressed in units of carbon [kgC d^{-1}],
$ingRate_{water,cattle}$	is the daily water demand for cattle [m^3 d^{-1}],
TC_{milk}	is the transfer coefficient from intake of radionuclides in fodder and water to the milk of cows [d l^{-1}],
$conc_{C,milk}$	is the concentration of carbon in milk [kgC l^{-1}],
TC_{meat}	is the transfer coefficient from intake of radionuclides in fodder and water to the meat of cattle [d kg_{FW}^{-1}], and
$conc_{C,meat}$	is the concentration of carbon in meat [kgC kg_{FW}^{-1}],

For agriculture on the drained mire, where arable land is used for both grazing and production of winter fodder, the specific C-14 activity in meat and milk is also approximated by the specific activity in fodder:

$$AC^{14C}_{meat,DM} = AC^{14C}_{milk,DM} = AC^{14C}_{fodder,DM} \qquad (9\text{-}37)$$

where

$AC^{14C}_{fodder,DM}$	is the specific activity in fodder (see Equation 9-32) [Bq kgC^{-1}]:

However, grazing in the biosphere object adds an extra exposure route for the livestock in the drained mire land use variant, namely ingestion of soil. Thus, the activity concentration in meat and milk [Bq kgC^{-1}] is a function of ingestion of water, fodder and soil in this land-use variant:

$$\begin{aligned} AC^{RN}_{i,DM} = \frac{TC_i}{conc_{C,i}}\Big(&\left(AC^{RN}_{regoUp,fodder,DM} + AC^{RN}_{regoUp,org,fodder,DM}\right) ingRate_{soil,cattle} \\ &+ AC^{RN}_{fodder}\, ingRate_{C,cattle} + AC^{RN}_{ing,water,cattle}\, ingRate_{water,cattle}\Big), \end{aligned} \qquad (9\text{-}38)$$
$$i = \{meat, milk\}$$

Following the discussion on future water resource management in Forsmark (Werner et al. 2014), it is assumed that livestock are watered from the same source used for human drinking.

$$AC^{RN}_{ing,water,cattle,i} = AC^{RN}_{ing,water,i},\ i = \{IO, DM\} \qquad (9\text{-}39)$$

9.5 Garden plot household

There are several possible pathways of exposure that may affect a small group of future permanent residents. These exposure pathways include irrigation, biomass fuel burning and soil improvement through fertilisation with seaweed (see Chapter 2). Present-day horticultural practice involves irrigation of potatoes and vegetables on outdoor garden plots. Moreover, biomass and peat are natural resources that may be used to produce heat and energy, and ash and marine algae have been valuable sources for fertilisers and mulch in Baltic coastal communities. However, the area affected by radionuclide release is limited, and both biomass fuel production in mires and algal production in near-shore basins are also limited. Therefore it is reasonable to evaluate exposure from irrigation, biomass fuel burning and seaweed fertilisation on a household scale (SKB 2014e).

Fertilisation with seaweed (i.e. macrophytes) is only possible in the submerged period (SKB 2014e). After isolation, wood or peat from the biosphere object can be used as fuel, and freshwater can be consumed and used to irrigate the garden plot. Therefore, the dose from ingestion of food, inhalation and direct external exposure is calculated using exposure as a consequence of fertilisation by seaweeds before the emergence/isolation of the biosphere object. After this point in time, the dose to inhabitants is calculated combining exposure from burning biomass fuel (wood or peat) and from utilising freshwater in the biosphere object:

$$Dose_i = \begin{cases} Dose_i^{fert,sea,macro}, & before\ threshold_{isolation} \\ \max(Dose_i^{comb,peat}, Dose_i^{comb,wood}) + Dose_i^{irrig}, & after\ threshold_{isolation} \end{cases}, \quad i = \{ingfood, inh, ext\} \tag{9-40}$$

A dug well, a drilled well or water from nearby lake or stream are all plausible sources for drinking and irrigation water in the garden plot land-use variant (Werner et al. 2014). In the safety assessment, the available water resource with the highest concentration of a radionuclide was used to calculate the internal dose from ingestion of water (and to irrigate the garden plot). The activity concentration in drinking water was calculated in a similar way as in the drained mire land-use variant:

$$AC_{ing,water,GP} = \max\big(max\big(AC_{water,well,dug}, AC_{water,well,drilled}\big) | \ time > threshold_{well}, AC_{water,surface} | \text{time} > threshold_{isolation}\big) \tag{9-41}$$

where

$AC_{water,well,dug}$	is the activity concentration of dissolved radionuclide in a well dug into the regolith layers of a biosphere object [Bq m^{-3}],
$AC_{water,surface}$	is the activity concentration in surface water (lake or stream) [Bq m^{-3}],
$threshold_{well}$	is the point in time after which it is feasible to construct a well [y], and
$threshold_{isolation}$	is the point in time when the lake is isolated from the sea or when the object is fully emerged at normal sea level [y].

External exposure from the ground in this land-use variant is primarily expected from cultivating the garden plot, and it is assumed that the time spent working with each crop is proportional to the cultivated area. Thus, the activity concentration on the ground causing external exposure can be expressed as the sum of the soil activity concentrations below each crop, weighted by the fractional area occupied by the crop:

$$AC_{ground,GP} = \sum_i f_{area,i}\big(AC_{regoUp,\text{GP},i} + AC_{regoUp,orgGP,i}\big), \quad i = \{vegetable, tuber\} \tag{9-42}$$

where

$f_{area,i}$	is the fraction of arable land that is used for crop i [m^2 m^{-2}],
$AC_{regoUp,GP,i}$	is the activity concentration in inorganic form in garden plot soils [Bq kg$_{\text{DW}}^{-1}$],
$AC_{regoUp,orgGP,i}$	is the corresponding concentration in organic form [Bq kg$_{\text{DW}}^{-1}$], and
i	is the type of garden plot produce.

As the discrimination of carbon isotopes is small during photosynthesis, the specific activity of C-14 in crops will reflect the specific activity of inorganic carbon (CO$_2$) in the canopy atmosphere and in soil pore water. Consequently, the specific C-14 activity can be approximated with a weighted average of the two uptake pathways (i.e. the canopy and root pathways). For the garden plot land-use variant the specific activity in crops is:

$$AC_{i,GP}^{14C} = \big(1 - f_{rootUptake}\big) SA_{atmos,CA,GP,i}^{14C} + \big(f_{rootUptake}\big) AC_{regoUp,GP,i}^{D14C} / conc_{DIC,regoUp,GP} \quad i = \{veg., \text{tuber}\} \tag{9-43}$$

where

$f_{rootUptake}$	is the fraction of carbon in biomass originating from root uptake [kgC kgC^{-1}].
$SA_{atmos,CA,GP,i}^{14C}$	is the specific activity concentration in crop canopy air [Bq kgC^{-1}],

$SA^{14C}_{regoUp,GP,i}$ is the activity concentration of dissolved C-14 in soil pore water [Bq m^{-3}], and

$conc_{DIC,regoUp,GP}$ is the concentration of stable DIC in soil pore water [kgC m^{-3}].

Cultivation of a garden plot is assumed to provide *all* potatoes and vegetables consumed by a household (see Grolander 2013 for details on diet fractions). Accumulation of radionuclides in edible parts of a crop is primarily a result of root uptake and plant allocation (see Section 8.2.2), and for the safety assessment it is assumed that the activity concentration in plant tissues will be in equilibrium with that in the upper soil layer. Moreover, radionuclides retained on the leaf surfaces, due to interception of irrigation water, are assumed to contribute independently to the activity concentrations of vegetables. The activity concentration in produce of the garden plot is defined as:

$$\begin{aligned} AC^{RN}_{vegetable,GP} &= \left(AC^{RN}_{regoUp,GP,veg} + AC^{RN}_{regoUp,org,GP,veg} \right) CR_{agri,\mathrm{veg}} + AC_{LeafRet}, \\ AC^{RN}_{tuber,GP} &= \left(AC^{RN}_{regoUp,GP,tub} + AC^{RN}_{regoUp,org,GP,tub} \right) CR_{agri,\mathrm{tuber}}, \end{aligned} \tag{9-44}$$

where

$CR_{agri,i}$ is the concentration ratio between soil and crop i [kg_{DW} kgC^{-1}], and

$AC_{LeafRet}$ is the activity concentration on the leaves of vegetables [Bq kg_{DW}^{-1}].

Inhalation of air results in dose due to exposure from small soil particles. The concentration of radionuclides in air above the garden plot is simply the product of the activity concentration in the soil and the concentration of fine soil particles in the air (Equation 8-6). In the case of C-14, gas (CO_2) from decomposition of organic fertilisers (i.e. seaweed) will also contribute to the activity concentration of inhalation air:

$$AC^{14C}_{air,GP} = AC^{14C}_{ground,GP} conc_{dust,GP} + \sum_i f_{area,i} AC^{14C}_{atmos,L1,GP,i}, \qquad i = \{tuber, vegetables\} \tag{9-45}$$

where

$conc_{dust,GP}$ is the concentration of fine soil particles in air above cultivated peat soils [kg_{DW} m^{-3}], and

$AC^{14C}_{atmos,L1,GP,i}$ is the activity concentration in air above the canopy of crop i [Bq m^{-3}].

After the biosphere object has emerged out of the sea, burning of biomass fuel causes additional exposure due inhalation of fly ash and combustion gases. In the safety assessment the fuel type yielding the highest inhalation dose is used to represent the garden plot land use scenario.

$$\begin{aligned} Dose_{inh,GP} &= Dose_{inh,air,GP} + Dose_{inh,comb} \\ Dose_{inh,comb} &= \max \begin{cases} AC_{fuel,peat} f_{combust} doseCoef_{combPeat} \\ AC_{fuel,wood} f_{combust} doseCoef_{combWood} \end{cases}, \end{aligned} \tag{9-46}$$

where

$Dose_{inh,air,GP}$ is the dose from inhalation of air above the garden plot, where radionuclides originate from fertilisation and/or irrigation [Sv y^{-1}],

$Dose_{inh,comb}$ is the dose contribution from inhalation of radionuclides originating from combustion of biomass fuel [Sv y^{-1}],

$AC_{fuel,type}$ is the activity concentration in bottom ash resulting from combustion of peat or wood [Bq kg_{DW}^{-1}], and

$f_{combust}$ is the fraction of the fuel inventory that ends up in fly ash and gas after combustion of wood or peat [kg_{DW} kg_{DW}^{-1}], and

$doseCoef_{combType}$ is the dose coefficient used for combustion of peat or wood [Sv y^{-1} per Bq kg_{DW}^{-1}].

Note that the dose coefficient for combustion ($doseCoef_{comb}$) represents exposure to an individual staying within 200 m from a release resulting from household scale heat production (Grolander 2013). That is, it factors in the consumption of fuel needed for heat production, the atmospheric dispersion, the outdoor exposure time and the dose coefficient for inhalation.

10 Dose-rate calculation for non-human biota

10.1 Background

There has been a shift from the long-held anthropocentric view of protection of the environment (e.g. ICRP 1977) to one which expressly considers biota. There is now a widely held view that there is a need to demonstrate, explicitly, that the environment can and will be protected from the effects of radiation. This issue has not been driven by any particular concern over environmental radiation hazards; rather, it has been developed to fill a conceptual gap in radiological protection (ICRP 2007).

ICRP (2007) suggests that the aim should be a negligible effect on the maintenance of biological diversity, the conservation of species, and the health and status of natural habitats, communities, and ecosystems; aims that are reflected in the specific regulations concerning protection of the environment issued by SSM (SSMFS 2008:37, Section 6): "*final management of spent nuclear fuel and nuclear waste must be implemented so that biodiversity and the sustainable use of biological resources are protected against the harmful effects of ionising radiation*." Further SSM states (SSMFS 2008:37, Section 7) that an assessment *"shall take particular account of the existence of genetically distinctive populations such as isolated populations, endemic species and species threatened with extinction and in general any organisms worth protecting.".* The general advice from SSM (SSMFS 2008:37) also states that "*the analysis of consequences for organisms in 'today´s biosphere' ... should be used for the assessment of environmental consequences in a long-term perspective*".

The organisms currently found at the site may therefore form a basis for the characterisation of the site, representing future assemblages in terms of the range of biological complexity, life-styles, size classes, and habitat uses exhibited by species, as well as the diversity, function and productivity of the ecosystem. Jaeschke et al. (2013) analysed the importance and benefit of using site-specific values for non-human biota, concluding that the use of the ERICA reference organisms (Brown et al. 2008), with some modifications and additions, was suitable for representing non-human biota at the site, at present and in the future. The selection of organism types included and the parameters used in the SR-PSU safety assessment is discussed in SKB (2014e).

10.2 ERICA dosimetric models incorporated in Ecolego

The safety assessment of SR-PSU with respect to non-human biota is based on the ERICA assessment approach (Brown et al. 2008, Beresford et al. 2008). The dosimetric models of ERICA were incorporated into the biosphere assessment model, and are further described below. Here, a brief description of the ERICA approach is given, for further details see also Jaeschke et al. (2013).
In brief, the ERICA Tool estimates absorbed dose rates from both external irradiation resulting from radionuclides in environmental media (soil, sediment, water) and from the incorporation of radionuclides within the organism. Geometrical data and habitat occupancies for the organism types considered, together with data on the decay properties of individual radionuclides, are used to calculate dose conversion coefficients (DCC) that are then applied as a means of converting activity concentrations in environmental media or within the organism itself to an absorbed dose rate for each radionuclide. Internal and external dose rates across all radionuclides of interest are then summed for each organism. The assessment approach is depicted in Figure 10-1.

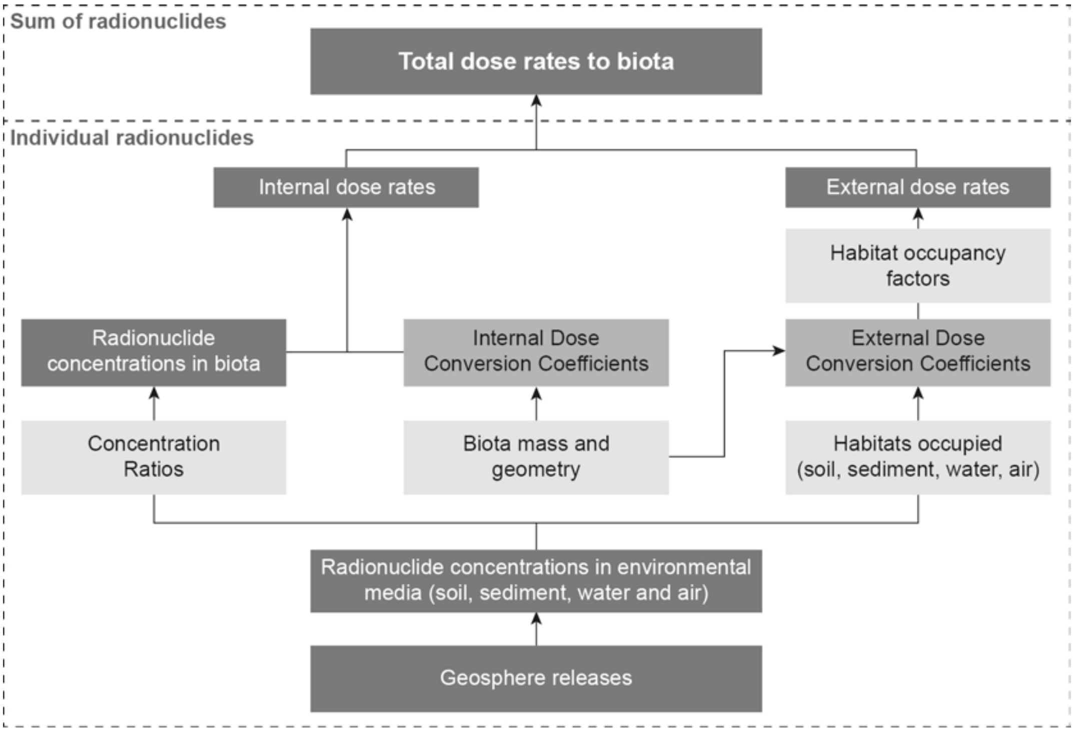

Figure 10-1. *Assessment of the impact of radionuclide releases on non-human biota. Darkest grey boxes indicate modelling with the radionuclide transport model. Lighter grey boxes indicate input parameters calculated with the ERICA Tool, whereas lightest grey boxes indicate other input parameters. The uptake of C-14 and H-3 in biota is simulated in a different way (see text).*

10.2.1 Basic concepts

The geometrical data of each considered organism are used within the ERICA Tool, together with radionuclide-specific radioactive decay properties and information on the habitat occupancies of the organism types considered to calculate dose conversion coefficients (DCC) (Ulanovsky and Pröhl 2006, Ulanovsky et al. 2008). These are quantities that link radionuclide activity concentrations in environmental media, or within organisms, to dose rates. Within ERICA, two sets of DCC are defined (Pröhl 2003):

- those used for calculating exposure resulting from internal radiation; DCC_{int} is defined as the internal absorbed dose rate (µGy/h) per unit activity concentration in an organism (Bq/kg fw); and
- those used for calculating exposure due to external exposure, DCC_{ext} defined as the external absorbed dose rate (µGy/h) per unit concentration in an environmental medium (Bq/kg or Bq/l fw).

As part of the work within the SR-PSU project, the ERICA dosimetric models for generic ecosystems (freshwater, marine and terrestrial), were implemented as Ecolego models. The radionuclide, radiation type and organism-specific DCCs were calculated within the ERICA Tool and incorporated in the biosphere assessment model as input parameters together with occupancy factors (ERICA default values with some changes as described in SKB 2014e) and radiation weighting factors. The latter were used to take account of the relative biological effectiveness of different radiations (α, low energy β and high energy β+γ), giving weighted biota dose rates. Throughout this assessment, the default weighting factors in ERICA (β+γ radiation = 1; low energy β radiation = 3; and α radiation = 10) have been applied (Grolander 2013).

Non-human biota dose rates are calculated for each biosphere object using local activity concentrations in environmental media as the source. In aquatic ecosystems, internal exposure is calculated using the concentration of the dissolved fraction of radionuclides in the water compartment whereas the total inventory of radionuclides in the water (dissolved as well as particulate fractions) have been used for calculating external exposure. The latter is a modification of the ERICA method (Brown et al. 2008), where the filtered water concentration is used for external exposure estimates. As it is difficult to justify why external exposure from activity in water surrounding an organism should only account for activity in solution, the exposure from radionuclides on or in suspended particulate matter was cautiously included in our exposure model. The activity concentration in aquatic surface sediments is used as input for external exposure for organisms living in or on the sediments. In terrestrial ecosystems, internal and external exposures are calculated using the radionuclide concentrations in surface peat.

The main biosphere object (157_2) has no defined lake state, as it evolves from a marine bay into a wetland area (SKB 2014a). However, as shallow surface water habitats (ponds) are likely to form in the object during the land period (Werner et al. 2014), the pore water concentrations of the mire have been used to calculate dose rates to limnic biota in this object. External exposures of freshwater organisms to sediment are calculated using the radionuclide concentrations in the surface peat.

10.2.2 Calculation of internal exposure

Weighted internal dose rates [µGy h^{-1}] for biota in all ecosystems are calculated by weighting all dose conversion coefficients before multiplying with the radionuclide activity concentration inside the organism:

$$DoseRate_{int,j} = w_{low\beta}\, DCC_{int,low\beta,j} + w_{\beta\gamma}\, DCC_{int,\beta\gamma,j} + w_{\alpha}\, DCC_{int,\alpha,j})\, AC_j \tag{10-1}$$

where

$w_{low\beta}$	is the weighting factor of internal low energy β-radiation [unitless],
$w_{\beta\gamma}$	is the weighting factor of internal (high energy) β- and γ-radiation [unitless],
w_{α}	is the weighting factor of internal α-radiation [unitless],
$DCC_{int,low\beta,j}$	is the dose conversion coefficient of internal low beta radiation for organism *j* [µGy h^{-1} per Bq kg_{FW}^{-1}],
$DCC_{int,\beta\gamma,j}$	is the dose conversion coefficient of internal beta gamma radiation for organism *j* [µGy h^{-1} per Bq kg_{FW}^{-1}],
$DCC_{int,\alpha,j}$	is the dose conversion coefficient of internal alpha radiation for organism *j* [µGy h^{-1} per Bq kg_{FW}^{-1}], and
AC_j	is the radionuclide activity concentration in the organism *j* whole body [Bq kg_{FW}^{-1}], see below.

Internal organism radionuclide concentrations are calculated using element- and organism-specific concentration ratios (CR), relating to concentrations in water (aquatic ecosystems) or soil (terrestrial ecosystems) (Tröjbom et al. 2013, Grolander 2013).

The internal whole body activity concentration, AC_j^{RN} [Bq kg_{FW}^{-1}], of aquatic organism j is:

$$AC_j^{RN} = CR_j^{RN}\, AC_{water}^{D,RN} \tag{10-2}$$

where

CR_j^{RN}	is the concentration ratio of radionuclide *x* for organism *j* [m^3 kg_{FW}^{-1}], and
$AC_{water}^{D,RN}$	is the activity concentration of the dissolved radionuclide in water [Bq m^{-3}] (Equation 5-4)

where the pore water concentration in surface peat, $AC_{regoUp,ter}^{D}$ [Bq m^{-3}] (Equation 6-7), is used for freshwater organisms living in shallow waters of biosphere objects where no lake or stream water is present:

$$AC_{water}^{D} = AC_{regoUp,ter}^{D} \tag{10-3}$$

Similarly, the whole body internal activity concentration, AC_j^{RN} [Bq kg_{FW}^{-1}], of terrestrial organism j is:

$$AC_j^{RN} = CR_j^{RN}\, AC_{regoUp,ter,tot}^{RN} \tag{10-4}$$

where

CR_j^{RN}	is the concentration ratio of a radionuclide for organism *j* [kg_{DW} kg_{FW}^{-1}], and
$AC_{regoUp,ter,tot}^{RN}$	is the total activity concentration of a radionuclide in the upper peat layer [Bq kg_{DW}^{-1}], including organic and inorganic activity (Equation 9-9).

Other concepts are used for the uptake of C-14 and H-3 in organisms. For C-14, the specific activity concentration in newly synthesised biomass of aquatic primary producers is approximated as the specific activity concentration of inorganic carbon in water (see plant uptake Equation 5-16). As the turnover time of aquatic organisms is relatively short, it is assumed that the specific activity in the whole body of primary producers is similar to that in water, and that the ratio of C-14 to stable carbon is conserved throughout the aquatic food chain. Thus the whole-body radionuclide activity concentration in aquatic biota, AC_j^{14C} [Bq kg_{FW}^{-1}], can be expressed in terms of the radionuclide activity concentration in water, the concentration of inorganic carbon in water and the fraction of carbon in the organism:

$$AC_j^{14C} = SA_{water}^{14C} f_{C,j} = \frac{AC_{water}^{D,14C}}{conc_{DIC,aqu}} f_{C,j} \tag{10-5}$$

where

SA_{water}^{14C} is the specific activity of inorganic carbon in water in the solute phase [Bq kgC^{-1}],

$AC_{water}^{D,14C}$ is the activity concentration of dissolved inorganic carbon-14 in water [Bq m^{-3}],

$conc_{DIC,aqu}$ is the concentration of dissolved inorganic carbon in water [kgC m^{-3}], and

$f_{C,j}$ is the fraction of carbon in the whole body of organism j [kgC kg_{FW}^{-1}].

Similarly, the whole body specific activity in terrestrial fauna, AC_j^{14C} [Bq kg_{FW}^{-1}], is assumed to be the same as in mire primary producers [Bq kgC^{-1}]. Consequently, the activity concentration in terrestrial organisms can be expressed in terms of the radionuclide activity concentration in mire vegetation and the fraction of carbon in the organism:

$$AC_j^{14C} = AC_{PP,ter}^{14C} f_{C,j} \tag{10-6}$$

where

$AC_{PP,ter}^{14C}$ is the activity concentrations of C-14 in mire primary producers [Bq kgC^{-1}] (Equation 6-1), and

$f_{C,j}$ is the fraction of carbon in the whole body [kgC kg_{FW}^{-1}].

To calculate the H-3 radionuclide activity concentration in aquatic biota it is assumed that the major H-3 source in organisms is within tissue fluids, and that these have the same H-3 concentration as the surrounding water. Thus, the whole body radionuclide activity concentration in the organism, AC_j^{3H} [Bq kg_{FW}^{-1}], can be calculated as the product of the activity concentration in water, and the fraction of body weight that is made up of fluids:

$$AC_j^{3H} = \frac{AC_{water}^{D,3H}}{dens_{water}} (1 - f_{DW,j}) \tag{10-7}$$

where

$AC_{water}^{D,3H}$ is the activity concentration of tritium in water [Bq m^{-3}],

$dens_{water}$ is the density of water [kg m^{-3}] (parameter value 1,000 kg m^{-3}) and

$f_{DW,j}$ is the fraction of dry weight in biota whole body [kg_{DW} kg_{FW}^{-1}].

For freshwater organisms living in the shallow water of biosphere objects where no lake or stream water is present, the pore water concentration in surface peat was used as the environmental water concentration (Equation 6-7).

Similarly, it is assumed that the major H-3 source in terrestrial organisms is tissue fluids, and that these are in equilibrium with H-3 concentration of dissolved radionuclides in the drinking water. As lake, stream and shallow surface waters in the biosphere objects are regarded as relevant sources for drinking water, the equation describing the internal whole body H-3 activity concentration in aquatic organisms (10.7) is also applicable to terrestrial organisms living in the biosphere object.

10.2.3 Calculation of external exposure

Due to the short range of the emitted particles and their absorption in superficial tissues, alpha emitters are not, in general, harmful to life unless the source is ingested or inhaled. Alpha particles cannot penetrate the skin or other organism surface layers and, therefore, alpha radiation does not contribute to external radiation exposure. Therefore, weighting factors are only applied to low β and β+γ radiation when considering external radiation exposure. Also, in terrestrial ecosystems, exposure from low β radiation has been assumed to be negligible (all DCCs in the ERICA are set to 0) and this exposure has therefore not been considered in this safety assessment.

The weighted external dose rate for biota in aquatic ecosystems can be calculated as:

$$DoseRate_{ext,j}^{aqu} = (0.001\ \mathrm{m^3 l^{-1}})\left(w_{low\beta}\ DCC_{ext,low\beta,j} + w_{\beta\gamma}\ DCC_{ext,\beta\gamma,j}\right)AC_{occup,j}, \tag{10-8}$$

where

$0.001\ \mathrm{m^3\ l^{-1}}$	is a unit transformation factor,
$w_{low\beta}$	is the radiation weighting factor for external low energy beta radiation [unitless],
$w_{\beta\gamma}$	is the radiation weighting factor for external beta gamma radiation [unitless],
$DCC_{ext,low\beta,j}$	is the dose conversion coefficient of external low beta radiation for organism type *j* [μGy h^{-1} per Bq l^{-1}]
$DCC_{ext,\beta\gamma,j}$	is the dose conversion coefficient of external beta gamma radiation for organism type *j* [μGy h^{-1} per Bq l^{-1}], and
$AC_{occup,j}$	is the occupancy corrected activity concentration for organism type *j* [Bq m^{-3}].

The occupancy corrected activity concentration is a weighted average of the activity concentration in water and in sediments. For organisms living on the sediment surface the weights are adjusted with a factor one half to reflect the reduced exposure from water and sediments. The occupancy corrected activity concentration for organism *j* is calculated as:

$$\begin{aligned} AC_{occup,j} = {} & \left(v_{water,j} + \frac{v_{wat.surf,j}}{2} + \frac{v_{sed.surf,j}}{2}\right) AC_{water,surf} \\ & + \left(\frac{v_{sed.surf,j}}{2} + v_{sed,j}\right) AC_{regoUp,tot,aqu} \end{aligned} \tag{10-9}$$

where

$v_{water,j}$	is the occupancy factor for the organism *j* in habitat water [unitless],
$v_{wat.surf,j}$	is the occupancy factor for the organism *j* in habitat water at its surface [unitless],
$v_{sed.surf,j}$	is the occupancy factor for the organism *j* in habitat sediment at its surface [unitless],
$v_{sed,j}$	is the occupancy factor for the organism *j* in habitat sediment [unitless],
$AC_{water,surf}$	is the total radionuclide activity concentration in water [Bq m^{-3}], (Equation 9-9) and
$AC_{regoUp,aqu,tot}$	is the total radionuclide activity concentration in surface sediments [Bq m^{-3}].

where the total activity concentration in aquatic surface sediments is the sum of contributions in organic and inorganic form:

$$AC_{regoUp,tot,aqu} = AC_{regoUp,aqu} + AC_{regoUp,org,aqu} \tag{10-10}$$

where

$AC_{regoUp,aqu}$	is the activity concentration in inorganic form in surface sediments [Bq kg_{DW}^{-1}],
$AC_{regoUp,org,aqu}$	is the corresponding activity concentration in organic matter [Bq kg_{DW}^{-1}].

For biota in terrestrial ecosystems, the weighted external dose rate is calculated from the activity concentrations in top soil, using occupancy factors corresponding to time in contact with this medium, and dose conversion coefficients that are unique for each habitat:

$$DoseRate_{ext,j}^{ter} = \big(w_{\beta\gamma} DCC_{ext,\beta\gamma,OnSoil,j} v_{OnSoil,j} + w_{\beta\gamma} DCC_{ext,\beta\gamma,InSoil,j} v_{InSoil,j} + w_{\beta\gamma} DCC_{ext,\beta\gamma,InAir,j} v_{InAir,j}\big) AC_{regoUp,tot,ter} \tag{10-11}$$

where

$w_{\beta\gamma}$ is the radiation weighting factor for external beta gamma radiation [unitless],

$DCC_{ext,\beta\gamma,k,j}$ is the dose conversion coefficient of external beta gamma radiation in habitat k (on soil, in soil or in air) for organism type j [μGy h^{-1} per Bq kg_{DW}^{-1}],

$v_{k,j}$ is the occupancy factor for the organism type j in habitat k [unitless], and

$AC_{regoUp,tot,ter}$ is the total radionuclide activity concentration in surface peat [Bq kg_{DW}^{-1}]

10.2.4 Total dose rate calculation

Finally the weighted total dose rate [μGy h^{-1}] for each organism type in each of the ecosystems is the sum of the weighted external and internal dose rates:

$$DoseRate_{total,j} = DoseRate_{ext,j} + DoseRate_{int,j} \tag{10-12}$$

where

$DoseRate_{ext,j}$ is the weighted external dose rate in organism type j [μGy h^{-1}] (Equation 10-8 or 10-11), and

$DoseRate_{int,j}$ is the weighted internal dose rate in organism type j [μGy h^{-1}] (Equation 10-1).

10.2.5 Transition of ecosystems in biosphere objects

In the dose rate calculations, the biosphere object is considered to be under marine conditions until the sea level reaches the lake threshold, or for objects without a proper lake stage, when the object is fully emerged at normal sea level (see parameter *threshold_isolation* in Grolander 2013). In the marine period, dose rates are estimated to marine organisms. Thereafter, conditions in the aquatic part of the object are considered to be suitable for freshwater organisms (and dose rate calculations are performed for limnic organisms). Terrestrial (i.e. wetland species) are assumed to colonize the terrestrial part of the object one year after the first land has emerged within the object (see parameter *threshold_land* in Grolander 2013) and dose rates to terrestrial organisms are calculated from this point in time (until the end of the simulation period). For each biosphere object, dose rates for marine and terrestrial organisms are estimated simultaneously for a short transitional period, followed by simultaneous dose rate estimations for freshwater and wetland biota.

11 Concluding remarks

The presented radionuclide model for the biosphere is used to calculate radionuclide transport and radiation doses and dose rates from a release of radionuclides from SFR to the Forsmark area for a time period of 100,000 years. The model is the current endpoint of SKB biosphere/dose assessment modelling since the 80ies, and it is in many ways similar to the model used in the latest SKB safety assessment SR-Site.

Given the requirements of the safety assessment, and the features of a developing coastal area, the model is capable of (i) handling the release of radionuclides to the surface with groundwater discharges and its dependence on landscape development and climate, (ii) incorporating development of ecosystems driven by e.g. land rise, shoreline displacement and ecosystem succession, (iii) accounting for transport of radionuclides between different areas of the landscape (driven by e.g. advection or surface runoff), (iv) handling transport, accumulation and decay of radionuclides (including progeny) with different geochemical behaviour, and (v) assessing radiation exposures of humans and other organisms.

The compartments and fluxes included in the model were identified from conceptual models for marine, freshwater and terrestrial ecosystems, from the description of the Forsmark landscape and its development, and from biosphere components identified to be important in safety assessments for radioactive waste disposal by SKB. The final set of model compartments includes the transport pathways from the bedrock to the surface areas, and environmental media with a potential to expose future inhabitants and other organisms to radionuclides from the repository (i.e. surface water, surface sediments, surface peat, cultivated soil and air). Fluxes of radionuclides were primarily associated with mass fluxes of water, solid matter and gas, and with transitions between inorganic and organic matter, between compartments that were in proximity to (or in physical contact with) each other.

The compartment modelling approach assumes that the system can be sufficiently represented by a finite number of homogenous compartments interconnected with each other. Due to the complexity of the surface ecosystems and the intrinsic uncertainties with regard to the development over large spatial and temporal scales, it is not feasible to calculate realistic dose estimates by fine grained mechanistic modelling. Instead a relatively small number of compartments and parameters are used to describe the outcome of the underlying processes at larger scales. Thus at the scales of the safety assessment it is assumed that ecosystem states can be sufficiently represented by average conditions, and fluxes of water, solid matter and gas can be adequately described as functions of aggregated empirical parameters, which capture the outcome of the underlying processes.

In SR-PSU, quantitatively relevant exposure pathways have been identified from a comprehensive range of potential pathways. For each of these pathways, combinations of environmental media and exposure routes are mapped to one (or more) exposed populations. Each exposed population represents a unique use of resources that is reasonable and sustainable from a historical perspective. Moreover, dose calculations are constrained by the physical and biological properties of the biosphere objects and by human requirements for energy and nutrients. A set of organisms living in potentially affected ecosystems (i.e. marine, liminc and wetland ecosystems) were also identified in SR-PSU. The selected organisms represent present and future assemblages in terms of the range of biological complexity, life-styles, size classes, and habitat uses exhibited by species, as well as the diversity, function and productivity of the ecosystem. Exposed populations and representative organisms should be seen as upper bounding cases for a combination of pathways when doses and dose rates are evaluated.

As the radionuclide model for the biosphere has been founded in knowledge of features, events and processes important for radionuclide transport from a geological repository, as well as in understanding of ecosystems and their development in Forsmark, and has the required model capabilities, we are confident that the model is fit for purpose. That is, that it can provide robust calculations of projected doses to evaluate if the long-term safety of the extended SFR repository complies with regulatory criteria. In the assessment, sets of parameters representing the geosphere and the biosphere have been combined with appropriate release scenarios in several calculation cases. These cases have then been used to evaluate potential impact on inhabitants and the environment for temperate conditions, as well as for a considerably colder or warmer climate than today. Thus given an appropriate parametrisation the radionuclide model for the biosphere is applicable also to other areas with a similar climate, geology and natural history.

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Appendix A

Principles of compartment modelling

The mathematical approach used in the radionuclide model of the biosphere is that of compartment modelling. This approach has been described in detail by for example Sheppard (1962), Jacquez (1972) and Andersson (1983). It assumes that a system can be adequately represented by a finite number of compartments or pools, each of which is homogeneous and connected to other compartments. A variable that represents the inventory of radionuclide activity (or the amount of matter) at a given time is associated with each compartment. This variable is affected by the fluxes into and out of the compartments, and a differential equation is used to describe the dynamic change in the activity in each compartment. If the system is simple, these equations can be solved analytically to describe the dynamic change of the variable over time.

A.1 Basic concepts and properties of compartment models

The starting point for modelling transport and accumulation of matter is the principle of conservation of mass, which states that 'the rate of change of mass in a specified region of space equals the rate at which mass enters that region minus the rate at which the mass leaves' (Denn 1986).

As an example for illustrating a single compartment model we can think of a lake that receives radionuclides from a steady source of contaminated surface water (Figure A-1). In order to describe the dynamics of the activity concentration in the Lake (AC_L, Bq m^{-3}) with a single compartment model we have to make some assumptions

Thus we assume that

1) volume (V, m^3) of the lake is constant,
2) surface water reaches the lake at a constant rate (q_{in}, m^3 day^{-1}),
3) input water has a constant activity concentration of a radionuclide (AC_I, Bq m^{-3}), and
4) water of the lake is well mixed.

Moreover, we assume that the contaminant is of low enough concentration to make up a negligible fraction of the volume in the lake, inflow water and outflow water, and for simplicity we neglect effects of precipitation and evaporation on the water balance of the lake.

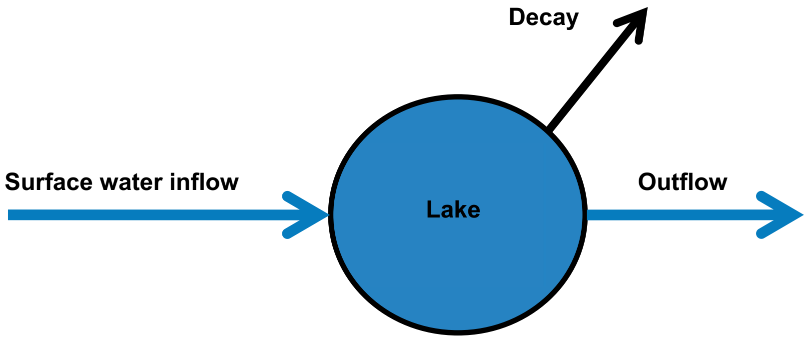

Figure A-1. Box and arrow representation of a hypothetical lake.

The total activity in the lake is $AC_L \cdot V$. The law of conservation now states that the rate of change of activity in the lake = rate at which activity enters – rate at which activity leaves. Taking radioactive decay into account, we can write this in symbolic terms:

$$\frac{dAC_L\,V}{dt} = AC_\mathrm{I}\,q_{in} - AC_L\,q_{out} - AC_L\,V\,\lambda \qquad \text{(A-1)}$$

where

$d(AC_L \cdot V)/dt$ represents the rate of change of the total lake activity concentration, and

λ is the rate of decay [day^{-1}].

From assumption 1) and mass balance with respect to water, it follows that the inflow of water is equal to the outflow of water ($q_\mathrm{in} = q_\mathrm{out} = q$). Equation A-1 is linear with respect to $AC_L \cdot V$, and the quantities AC_I, q, V and λ are all constants. Thus the equation can be solved to obtain the solution:

$$AC_L\,V(t) = \frac{AC_\mathrm{I}\,q}{\frac{q}{V}+\lambda}\left(1 - e^{-\left(\frac{q}{V}+\lambda\right)t}\right) + AC_{L,0}\,V\,e^{-\left(\frac{q}{V}+\lambda\right)t} \qquad \text{(A-2)}$$

where

$AC_{L,0}$ represents the initial the activity concentration in the lake at time t=0.

After division with V we obtain the corresponding solution in terms of activity concentration:

$$AC_L(t) = \frac{AC_\mathrm{I}\,\frac{q}{V}}{\frac{q}{V}+\lambda}\left(1 - e^{-\left(\frac{q}{V}+\lambda\right)t}\right) + AC_{L,0}\,e^{-\left(\frac{q}{V}+\lambda\right)t} \qquad \text{(A-3)}$$

The expression ($q/V+\lambda$), the exponential coefficient of time, sets the time scale of the dynamics of the activity and the activity concentration through time, and is frequently referred to as the *eigenvalue*. A first-order linear differential equation has one eigenvalue, and the inverse of this eigenvalue is referred to as the *return time to equilibrium* (T_R)[7].

T_R can be interpreted in terms of system perturbation, and it is the time that it takes for the system to return to the fraction e^{-1} of the original perturbation. Thus for a system with the steady-state equilibrium AC_L^*, T_R is the time it takes for the system to return to $(1-1/e)AC_L^*$, when perturbed to $AC_{L,0} = 0$. The return time can be scaled to any fraction of the original perturbation. For example $T_R \cdot \ln(2)$ corresponds to the time it takes to reach half the initial perturbation (or the half-life of perturbation).

The solution (Equation A-2) contains many of the basic properties that are also of interest in multi compartment models. For example as $t \to \infty$ the activity concentration approach a constant value

$$AC_L{}^* = \frac{AC_\mathrm{I}\,\frac{q}{V}}{\frac{q}{V}+\lambda} \qquad \text{(A-4)}$$

where

the asterisk (*) indicate that the activity concentration is at *steady-state equilibrium*.

In particular, we note that if the half-life of the radionuclide is large, (i.e. λ is much smaller than q/V), then the steady-state activity concentration equals the activity concentration AC_I in the influx water. And in the special case when the influx ($AC_I \cdot q$) is expressed as a unit release rate (1 Bq year^{-1}), $AC_L^* = 1/q$ (Bq). However, when $\lambda > 0$ the steady state activity concentration will always be a fraction of the concentration in the input water (AC_I).

The part of the solution that depends on time is called the *transient* component. For the activity concentration in this example it equals $(AC_{L,0} - AC_L^*) \cdot e^{-(q/V+\lambda)t}$. When the time scales of interest are large compared with the return time (T_R), the absolute value of the exponent in the expression becomes large, and the contribution of the transient component small, as compared to the to the contribution of the steady-state term.

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[7] The first term of the T_R expression (q/V) is frequently referred to as the residence time (or removal time) and represents the average amount of time that a particle (not subject to decay) spends in a particular system.

Figure A-2 illustrates the points made above by comparing the activity concentration in water of two hypothetical radionuclides in a large water body with a long residence time (54 years). Radionuclides reach the water body with inflowing water which has a concentration of 1 Bq m^3. Although the difference in decay rate between the two radionuclides is only one order of magnitude there is a clear difference in both the dynamics and the steady-state concentrations. That is, for the (relatively) fast decaying radionuclide the steady state activity concentration is significantly lower than in the incoming water (~a factor five) and the time to approach equilibrium is primarily determined by the decay rate (timescale ~11 years). On the other hand, the dynamics of the system, and the steady state activity concentration of the slowly decaying radionuclide, are primarily determined by the residence time of the water in the lake and the concentration of the inflowing water (see above).

A.2 Adsorbed and solved radionuclides

Radionuclides from an underground repository are expected to reach the sediment layers in the biosphere via deep groundwater. Once dissolved radionuclides reach sediments they will equilibrate with the solid state through different chemical and biological processes (e.g. precipitation, complex formation or immobilization by microorganisms) collectively referred to as sorption. Thus, a fraction of the radionuclides entering the sediments in solution will be retained in the sediment solid phase.

The usual approach in radionuclide transport modelling, where time scales are long and information on physical and chemical states and processes is uncertain, is to treat the sorption process as being fast and independent of radionuclide concentrations. Thus the net outcome from retention processes can be described a constant solid/liquid distribution coefficient K_d:

$$Kd(X) = \frac{[X]_{\text{solid}}}{[X]_{\text{solution}}} \tag{A-5}$$

where

$K_d(X)$ [m^3 $kgDW^{-1}$] is defined as the ratio between the radionuclide activity concentrations in the solid, $[X]_{\text{solid}}$ [Bq $kgDW^{-1}$] and solute, $[X]_{\text{solute}}$ [Bq m^{-3}], phases, respectively.

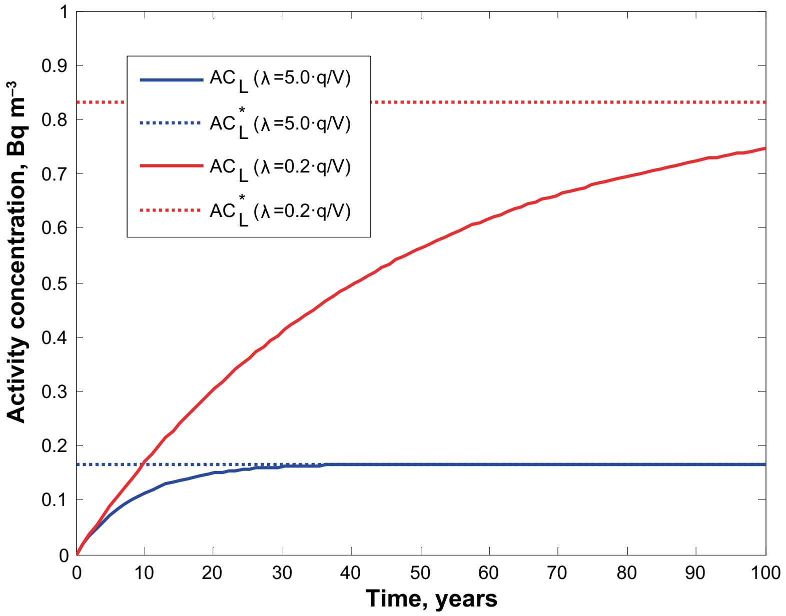

Figure A-2. *Activity concentration (AC_L) in a hypothetical lake as a function of time and decay rate (λ). Dotted lines represent equilibrium activity concentrations (AC_L*).*
$AC_{L,0} = 0$ Bq m^{-3}, $AC_I=1$ Bq m^{-3}, $V=1.4\times10^8$ m^3, $q=2.6\times10^6$ m^3 $year^{-1}$ and $\lambda = 5$ q/V (red), or 0.2 q/V (blue) $year^{-1}$.

The fraction of the total amount of radionuclides that are in the solute, f_{pore} (Bq Bq^{-1}), will also be constant, and can be described in terms of the K_d and soil physical properties:

$$f_{pore}(X) = \frac{1}{1 + \frac{Kd(X)\rho}{\theta S}} \tag{A-6}$$

where

ρ is the sediment density [kg$_{DW}$ m^{-3}], and

θ is the porosity of the sediment [m^3 m^{-3}].

S is the degree of water saturation (i.e. the fraction of pore space filled by pore water).

The K_d concept can also be applied to partitioning of radionuclides between surface water and suspended matter. Using mass balance, the fraction of the radionuclide inventory in surface water that is partitioned to the water, f_{water}, can be described in terms of K_d and the concentration of suspended matter:

$$f_{water}(X) = \frac{1}{1 + Kd(X)conc_{PM}} \tag{A-7}$$

where

$conc_{PM}$ is the concentration of suspended matter [kg$_{DW}$ m^{-3}] in for example sea or lake water.

Figure A-3 shows how the percentage of total activity in the water column that is adsorbed to suspended particulate matter varies as a function of the distribution coefficient.

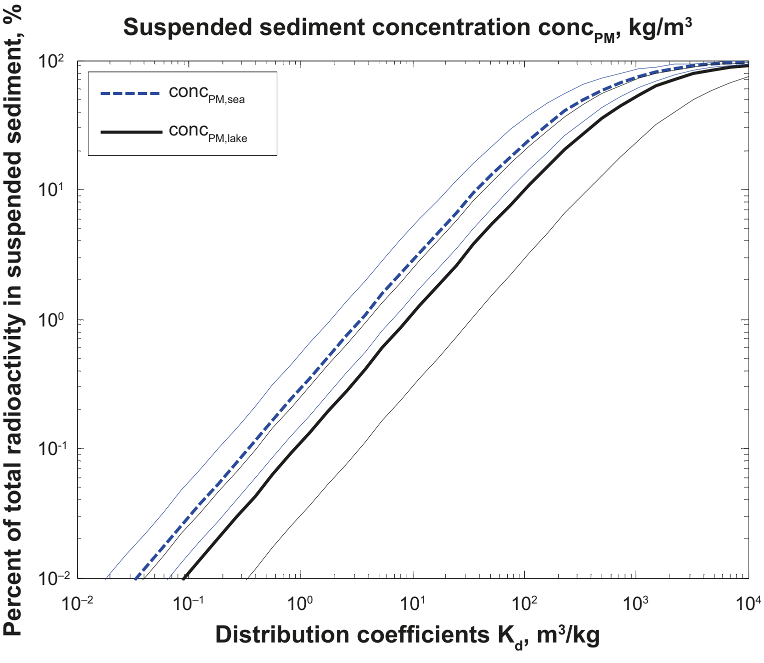

Figure A-3. *Percentage of total activity adsorbed onto suspended particulate matter as a function of the distribution coefficient. Thick lines represent marine (blue dashed) and freshwater (black solid) conditions with respect to the concentration of suspended matter (0.003 and 0.001 kg$_{DW}$ m^{-3}), and thin lines represents corresponding minimum and maximum values (Grolander 2013).*

A.3 One compartment representing two states

If sorption processes are fast the dynamics in a two state solid/liquid system, like soil or water with suspended particular matter, can be described by one inventory. For example, if we consider a water saturated soil compartment of total volume V and assume complete mixing, then we can rewrite Equation A-1 to represent the rate of change of the total activity I (Bq) in the soil compartment:

$$\frac{dI}{dt} = AC_I \, q - AC_{pore} \, q - I \, \lambda \qquad \text{(A-8)}$$

where

AC_{pore} is the activity concentration in pore water [Bq m^{-3}], i.e. the fraction of the inventory partitioned into the pore water (f_{pore}), divided by the sediment pore volume $V \cdot \theta$ [m^3].

For convenience, radionuclide decay (λ, year^{-1}) and water flux (q, $m^3 \text{ year}^{-1}$) are expressed on the time scale of years in this example.

$$\frac{dI}{dt} = AC_I \, q - I \left(\frac{f_{pore} \, q}{V\theta S} + \lambda \right) \qquad \text{(A-9)}$$

The solution of the linear differential equation becomes:

$$I(t) = \frac{AC_I \, q}{\frac{f_{pore} \, q}{V\theta S} + \lambda} \left(1 - e^{-\left(\frac{f_{pore} \, q}{V\theta S} + \lambda\right)t} \right) + I_0 e^{-\left(\frac{f_{pore} \, q}{V\theta S} + \lambda\right)t} \qquad \text{(A-10)}$$

The return time to equilibrium is the inverse of the exponential coefficient of time. If f_{pore} is expressed in terms of K_d, ρ, S and θ, this return time becomes:

$$T_R = \left[\frac{q}{V\theta S} \left(1 + \frac{Kd \, \rho}{\theta S} \right)^{-1} + \lambda \right]^{-1} \qquad \text{(A-11)}$$

Similarly the steady state solution for I can be written:

$$I^* = \frac{AC_I \, q}{\frac{q}{V\theta S} \left(1 + \frac{Kd \, \rho}{\theta S} \right)^{-1} + \lambda} \qquad \text{(A-12)}$$

and the activity concentration in the pore water at steady state is:

$$AC^*_{pore} = AC_I - I^* \frac{\lambda}{q} \qquad \text{(A-13)}$$

For a radionuclide with a slow rate of radioactive decay ($\lambda \rightarrow 0$) the time to approach equilibrium, T_R, will increase by a factor ($1+K_d \cdot \rho/\theta S$) when compared to a non-sorbing conditions (A-11). The steady state inventory, I^*, will increase by the same factor (A-12). Thus the inverse of the pore water fraction f_{pore} is frequently referred to as the *retardation factor* or *the storage capacity*. As there is no loss through radioactive decay, the steady-state concentration of pore water will equal that in the input water (A-13) (see Table A-1 $\lambda = 10^{-8}$).

For radionuclides with a medium rate of decay, (compared to the product of pore water turnover rate and f_{pore}, i.e. $\lambda \approx (q/V\theta S) \cdot (1+K_d \cdot \rho/\theta S)^{-1}$), soil physical and chemical characteristics (ρ, θ, S, K_d) will affect both the time to reach steady state from a perturbation and the steady state inventory of the radionuclide. For such conditions, the steady-state of pore water concentration will be approximately half of that in the input water. (Note that the radioactive decay will be an internal source term for the progeny product which may be radioactive, but that this activity contribution is not considered in this simple example).

However for radionuclides with a fast rate of decay ($\lambda >> (q/V \cdot \theta) \cdot (1+K_d \cdot \rho/\theta S)^{-1}$) the return time to equilibrium will only depend on radionuclide half-life. Under such conditions the steady state inventory will approach the ratio between the input rate of radionuclides ($AC_I \cdot q$, Bq year^{-1}) and the radionuclide decay rate (λ, y^{-1}) ($\lambda = 10^{-4}$ and $K_d \geq 1$ in Table A-1), and the flux of the original radionuclides from the compartment through pore water will only make up a small fraction of the flux into the compartment.

Table A-1. Return time (T_R) and steady-state conditions with respect to the total inventory (I*) and the pore water activity concentration (AC*) in an inorganic water saturated regolith layer as a function of radionuclide (specific) properties (λ and K_d). The calculations are based on a layer with a depth of 4 m and an area-specific vertical groundwater flux rate of 45 mm year^{-1}. The concentration in the inflowing water is 1 Bq year^{-1}. Typical values for till were used for soil physical properties (q = 0.2 and ρ = 2,100 kg_{DW} m^{-3}).

K_d $\text{m}^3/\text{kg}_{\text{DW}}$	λ year^{-1}	T_R year	I^* Bq	AC_{pore}^* Bq/m^3	$(q/V\cdot\theta)\cdot(1+K_d\cdot\rho/\theta)^{-1}$ year^{-1}
0	10^{-4}	4×10^2	0.2	1	2×10^3
0	10^{-6}	5×10^2	0.2	1	2×10^3
0	10^{-8}	5×10^2	0.2	1	2×10^3
10^{-5}	10^{-4}	5×10^2	0.2	1	2×10^3
10^{-5}	10^{-6}	5×10^2	0.2	1	2×10^3
10^{-5}	10^{-8}	5×10^2	0.2	1	2×10^3
0.01	10^{-4}	8×10^3	20	0.8	2×10^5
0.01	10^{-6}	5×10^4	20	1	2×10^5
0.01	10^{-8}	5×10^4	20	1	2×10^5
0.1	10^{-4}	1×10^4	70	0.3	2×10^6
0.1	10^{-6}	3×10^5	200	1	2×10^6
0.1	10^{-8}	5×10^5	200	1	2×10^6
1	10^{-4}	1×10^4	100	0.05	2×10^7
1	10^{-6}	8×10^5	2,000	0.8	2×10^7
1	10^{-8}	5×10^6	2,000	1	2×10^7
10	10^{-4}	1×10^4	100	0.005	2×10^8
10	10^{-6}	1×10^6	7,000	0.3	2×10^8
10	10^{-8}	3×10^7	20,000	1	2×10^8

Appendix B

Fraction of CO_2 in pore gas

The fraction of the radionuclide inventory that is partitioned to the gas phase in a given volume of soil is by definition:

$$f_{gas} = \frac{I_{gas}}{I_{tot}} \quad \text{(B-1)}$$

where

I_{gas} [Bq] is the inventory in the gas phase and

I_{tot} [Bq] is the total inventory in the sediment.

Considering all fractions of radionuclides in unsaturated soil, mass balance gives:

$$I_{tot} = I_{pore} + I_{gas} + I_{solid} \quad \text{(B-2)}$$

where

I_{pore} [Bq] is the inventory of dissolved radionuclides in the pore water, and

I_{solid} [Bq] is the inventory of radionuclides in the solid state.

For elements with a sufficiently small *Kd* the solid phase can be neglected and the total inventory approximated by the sum over the inventories in pore water and pore gas.

The inventory in the pore water can be expressed as the product of the activity concentration and the volume of the pore water, and the inventory of the gas state can be expressed as the product of the activity concentration and the gas volume, respectively:

$$l_{pore} = AC_{solution}\, V\, \theta\, S_w \quad \text{(B-3)}$$

$$l_{gas} = AC_{gas}\, V\, \theta\, (1–S_w) \quad \text{(B-4)}$$

where

$AC_{solution}$	is the activity concentration in pore water volume [Bq $m^{–3}$],
AC_{gas}	is the activity concentration in pore gas [Bq $m^{–3}$],
V	is the soil volume [m^3],
θ	is the soil porosity [$m^3\ m^{–3}$], and
S_w	is the degree of soil saturation (the fraction of pore space filled with water) [$m^3\ m^{–3}$].

Henry's law states that the solubility of a gas in a liquid is directly proportional to the partial pressure of the gas above. That is, at equilibrium, the concentration in water can be expressed as a function of the atmospheric concentration and a solubility constant:

$$AC^*_{solution} = AC^*_{gas}\, solubilityCoef_{ter} \quad \text{(B-5)}$$

where

$solubilityCoef_{ter}$ is a solubility coefficient [mol $m^{–3}$ per mol $m^{–3}$].

Now, combining Equations B-2 to B-5 the fraction of gas can be expressed in terms of soil saturation and the solubility coefficient:

$$f_{gas} = \frac{1 - S_w}{1 - S_w(1 - solubilityCoef_{ter})} \quad \text{(B-6)}$$

For biologically active soils, the carbonate system is driven towards H_2CO_3 due to constant production of CO_2 (Zabowski and Sletten 1991). For high pH soils, some of the dissolved inorganic carbon will still be in the form of carbonate (HCO_3^-), and thus for such a soil the above fraction will over-estimate the fraction of the DIC pool that is in the gas phase. However, experimental studies on the fractionation of radiocarbon, suggests that bulk pH has a limited effect on the fraction of gas in cultivated soil at pH 7 or below (Ishii et al. 2010). Thus for the purpose of a safety assessment, the expression in B5 is a reasonable approximation for the gas fraction of DIC in the pore space of cultivated soils.

Appendix C

Fraction of radionuclides in crop

The fraction of the radionuclide inventory for a soil-plant system that is immobilised in crop, f_{crop}, is by definition:

$$f_{crop} = \frac{I_{crop}}{I_{tot}} \tag{C-1}$$

where

l_{crop}	is the inventory in crop biomass [Bq], and
l_{tot}	is the total inventory in the soil and crop system [Bq].

Considering only two fractions of radionuclides in the soil plant system, mass balance gives:

$$l_{tot} = l_{crop} + l_{regoUp} \tag{C-2}$$

where

l_{regoUp} is the inventory of radionuclides in the biologically active part of the soil [Bq].

The inventory in the crop, l_{crop} [Bq KgC^{-1}], can be expressed as the product of the activity concentration in crop and the crop biomass, and the inventory of soil, l_{regoUp} [Bq kg$_{DW}$$^{-1}$], can be expressed as the product of the total activity concentration in soil and soil mass, respectively:

$$l_{crop} = AC_{crop}\ biom_{crop} area_{agri} \tag{C-3}$$

$$l_{regoUp} = AC_{regoUp}\ area_{agri}\ Z_{regoUp}\ \rho_{regoUp} \tag{C-4}$$

where

AC_{corp}	is the activity concentration in crop [Bq kgC^{-1}]
AC_{regoUp}	is the activity concentration in soil [Bq kg$_{DW}$$^{-1}$]
$biom_{crop}$	is the area-specific crop biomass in carbon units [kgC m^{-2}],
$area_{agri}$	is the surface area of cultivated land [m^2],
Z_{regoUp}	is the soil depth [m], and
ρ_{regoUp}	is the soil density [kg$_{DW}$ m^{-3}].

Now, if we assume that the activity concentrations in plant and soil are in equilibrium, we can insert the right hand expression of Equations C-2 to C-4 into Equation C-1, substitute AC_{regoUp} with AC_{corp}/ CR_{crop}, and rearrange the equation:

$$f_{crop} = \frac{biom_{crop}}{biom_{crop} + \frac{z_{regoUp}\ \rho_{regoUp}}{CR_{crop}}} \tag{C-5}$$

where

CR_{crop} is the ratio between concentration in crop and soil at equilibrium [kg$_{DW}$ kgC^{-1}].

Appendix D

Degassing from unsaturated soil under steady-state concentrations

Fick's 2nd law predicts how diffusion causes the concentration to change with time. If we only consider one spatial dimension (z), then the change of concentration of an element with time is:

$$\partial_t c = -\partial_z (D \partial_z c) \tag{D-1}$$

where c is the concentration (mol m^{-1}),

z is the position along a gradinent (m), and

D is the diffusion coefficient in unsaturated soil (m^2 y^{-1})

If we now consider a soil column of depth Z, we can use Fick's 2nd law to describe the soil concentration. When the concentration has reached steady state, we get:

$$-\partial_z (D \partial_z c) = -Dc'' = 0 \tag{D-2}$$

Moreover, if we consider a source at the bottom of the column (z=0), and assume that the atmosphere above the soil (z=Z) acts as a sink (due to extensive dilution), we get the boundary conditions:

$$-D\partial_z c|_{(z=0)} = -Dc'(0) = IR$$

$$c(Z) = 0 \tag{D3}$$

where

IR is the input rate of an element to the bottom (mol time^{-1}).

The solution of this (ordinary) differential equation is:

$$c(z) = \frac{IR}{D}(Z - z) \tag{D-4}$$

That is, at steady state the concentration changes linearly from the bottom of the column to the soil surface.

By integrating over the air-filled pore space along the soil profile (from the bottom to the top) we get the total inventory within the soil column (mol):

$$Inventory = \int_0^Z c(z)\theta(1 - S_w)\, dz = \theta(1 - S_w)\, \frac{1}{2}\frac{IR}{D} Z^2 \tag{D-5}$$

where

θ is the soil porosity (m^3 m^{-3}), and

S_w is the degree of water saturation, i.e. the fraction of pore space filled by pore water.

By dividing the flux IR (mol time^{-1}) by the inventory I (mol), we get the turn-over rate of the inventory (time^{-1}), or the proportional rate of degassing:

$$k_{degassing} = \frac{IR}{I} = \frac{2D}{\theta(1 - S_w)\, Z^2} \tag{D-6}$$

It should be noted that the concentrations above are to be understood as one dimensional concentrations (mol m^{-1}), which can be multiplied by a constant cross section of the soil column (m^2) to get conventional volumetric concentrations (mol m^{-3}). Moreover, if the there are other process affecting the inventory of an element in the soil column (e.g. radioactive decay or leaching), then the rate in Equation D-6 will overestimate the rate of transfer due to degassing.

However, the degassing rate of a the cultivated soil layer (~0.25 m), with a typical soil diffusion coefficients for CO_2 (~6 m^2 year^{-1}) will be two orders of magnitude larger than the rate associated with leaching (K_d=0, percolation 0.4 m year^{-1}, soil porosity 0.6, degree of saturation 0.6) (parameter values for cultivated soils from Grolander 2013), and several orders of magnitude larger than those associated with radioactive decay (the half-life of C-14 is 5,730 years). Thus for the purpose of this safety assessment, the rate in D-6 appears to be a reasonable approximation for degassing of CO_2 from unsaturated cultivated soils.

Appendix E

Biosphere process audit

Out of the original 45 SKB FEP processes identified as being important to consider in a safety assessments for radioactive waste disposal in a geological repository (SKB 2014e), twenty five were explicitly represented by at least one flux in the radionuclide transport model (Table E-1).

The effects of approximately half of the remaining twenty-one FEP processes were accounted for in parameter values. For example, the effect of habitat supply on radionuclide uptake by plants was accounted for by scaling NPP with parameters describing the photic zone, and the surface area of aquatic and terrestrial ecosystems, respectively. The effect of sorption/desorption was handled by the Kd-parameter, and the effect of wind stress on gas exchange was considered in parameter selection of the piston velocity. For further examples see table below.

The remaining FEP processes were disregarded for various reasons. The effect of three FEP processes (i.e. species interactions, species introduction, and weathering) was handled by assuming that biotic and abiotic properties of the ecosystem were in approximate equilibrium, and that effects of these processes thus could be sufficiently represented by average conditions at the site. Exposure of humans and non-human biota was handled outside the transport modelling (see Chapter 9) and loss of radionuclides through human material use and food harvesting was cautiously ignored. Acceleration, (i.e. the change in rate or direction of velocity), was not considered relevant on the temporal resolution of a year.

Table E-1. Audit of biosphere processes which may affect transport of radionuclides in the surface ecosystems. Processes that are represented by a flux (or affects an inventory) of radionuclides in the ecosystem transport models are marked with Y (yes), and the corresponding flux is listed in italics in the comments field. Processes that are not represented in the transport model are left blank. The effects of these processes on transport are primarily captured by site specific parameter values, and to a lesser extent by cautious or simplifying assumptions (see comments field and text above).

Process		**Radionucl. flux**			
FEP #	**Name**	**Aqua**	**Ter**	**Agri**	**Comment.**
Biological processes					
1	Bioturbation	Y		Y	Bioturbation in aquatic ecosystems. (Limits on bioturbation cause *burrial)*. In the drain/cultivate scenario, inventories from upper regolith layers are initially mixed into one biologically active cultivated layer.
2	Consumption	Y	Y		On a yearly basis there is no accumulation of radionuclides in primary producers. Instead labile primary producer biomass is consumed by grazers or decomposers, and radionuclides are released to the environment (*litter respiration/release*). Refractory dead organic matter (consumed or not) is deposited in the upper regolith layer (*litter production*).
3	Death	Y	Y		See Consumption above. On a yearly basis net primary production is balanced by death, through grazing and sencense (*litter respiration/release, litter production*).
4	Decomposition	Y	Y	Y	Decomposition of refractory organic matter is represented in all organic regolith layers (*mineralisation*)
5	Excretion	Y	Y		Non-mineralised net primary production is excreted (if consumed), and subsequently deposited to surface regolith layers (*litter production*)
6	Food supply				See Consumption (2) above. Losses of radionuclides associated with harvest are cautiously ignored. Food supply (productivity) is included in dose calculations
7	Growth				Effects of growth on radionuclides deposited by irrigation is included in the weathering/wash off rate from leaves (*leaf retention*).
8	Habitat supply				Effects of habitat avilability on primary producer is included in parameter values for the photic/surface area of the contaminated object, which in turn is used to scale net primary production (NPP) and biomass.
9	Intrusion			Y	Consequences of an intrusion well are considered as a source of radionuclides due to *irrigation.*

Process		**Radionucl. flux**			
FEP #	**Name**	**Aqua**	**Ter**	**Agri**	**Comment.**
10	Material supply				Losses of radionuclides through export of material (wood, peat, or regolith) for human use are cautiously ignored.
12	Particle release /trapping	Y	Y		Refractory organic matter deposited on aquatic surface sediments (litter production) is assumed to be fragmented and become part of top sediments. That is, it is implicitly included in litter production flux.
13	Primary production	Y	Y		Fixation of carbon is included in *plant uptake.* Net primary production also drives *uptake, litter respiration*/*release* and *litter production.*
14	Stimulation / inhibition				Biological communities treated as being in equilibrium. (Level of detail not compatible with model resolution).
15	Uptake	Y	Y	Y	*Plant uptake* in all ecosystems. *Root uptake* considered in Mire. Animal uptake included in dose calculations.
Processes related to human behaviour					
16	Anthropogenic release			Y	Release of radionuclides due to *irrigation* and *fertilization* included as source terms. Release to the atmosphere due to combustion of peat or firewood is included in dose calculations.
17	Material use			Y	Human utilisation of wetland hay, seaweeds and peat or wood ash for *fertilization* is included as source terms.
18	Species introduction/ extermination				Biological communities treated as being in equilibrium.
19	Water use			Y	Effects of irrigation considered in the garden plot scenario.
Chemical, mechanical and physical processes					
21	Consolidation			Y	Effect of subsidience considered when wetland/lake regolith layers are drained and cultivated. Rock formation not relevant within assessment context.
22	Element supply				Element concentration considered as part of site characteristics (e.g. CR values). Radionuclides released to the biosphere from a repository are assumed to be small relative the abundance of naturally occurring elements, and have no effect on supply of elements.
24	Phase transitions	Y	Y	Y	Phase transition included in the gas exchange between surface water/soil pore water and the atmosphere (*degassing* and *gas uptake*).
25	Physical properties change			Y	Soil properties (porosity and density) change when lake and mire regolith is drained and cultivated. (Model parameters describe how the density and porosity of regolith change in the transition between natural ecosystems).
26	Reactions				The equlibium concentration ratio between soild and liquid phases (kd) includes the outcomeof chemical reactions.
27	Sorption / desorption				The equlibium concentration ratio between soild and liquid phases (*Kd*) includes the outcomeof sorption/desorption.
28	Water supply				Human and livestock consumption/use of water is considered to be an insignificant sink for surface water. (The water extraction rate from a well is included as a model parameter).
29	Weathering				Element concentrtion in soil solution and surface water considered part of site characteristics (as reflected in *Kd* values). Radionuclide release to the biosphere is assumed to be small in mass relative to the abundance of naturally occuring elements (and to have no effect on concentrations of elements or their analogues).
30	Wind stress				Effect of windspeed on gas exchange considered in parameter selection (piston velocity)

Process		**Radionucl. flux**			
FEP #	**Name**	**Aqua**	**Ter**	**Agri**	**Comment.**
Transport processes					
31	Acceleration				Acceleration is not considered relevant on the temporal resolution of year.
32	Convection	Y	Y	Y	Included in water exchange between sea basins and down-stream transport (*advection horizontal*), vertical groundwater advection (*advection vertical*), pore water *diffusion* and *degassing.* Implicitly in expression for atmospheric equilibrium concentrations.
33	Covering	Y	Y		Included in transition of lake to mire (*ingrowth*). Effect of ice-cover accounted for in parameter values for *degassing* and *gas uptake.*
34	Deposition	Y	Y		*Sedimentation* in aquatic ecosystem. Implicit in direction of *litter production* flux.
35	Export	Y	Y	Y	Radionuclide fluxes associated with fluxes of water and gas, including: Sea basin exchange and down-stream fluxes (*Adv_water_out)*, *leaching and degassing.* Dilution of radionuclides through movement of solid matter or biological populations out of the model area, is cautiously neglected.
36	Import	Y	Y	Y	Radionuclides enter the biosphere object through: Groundwater discharge (*release*) or *groundwater uptake*, sea basin exchange and surface water from upstream object (*Adv_water_in*), and *gas uptake*. See Antropogenic release for additional source terms to agricultural ecosystems.
37	Interception			Y	Effects of intercepted radionuclides from irrigation is included in the Garden plot scenario (*leaf retention).*
38	Relocation			Y	Relocation of radionuclides through *fertilization* (hay-making, sea weeds, ash) is source term in infield-outland and garden plot cultivation.
39	Resuspension	Y			*Resuspension* of radionuclides from upper sediment explicitly included. Transport of radionuclides stored in, or adsorbed to, fine particular matter included in advective water fluxes. Resuspension of fine particular matter in dust considered in exposure/dose calculations.
40	Saturation				Regolith layers of sea, lake and mire ecosystems are assumed to be saturated. Degree of saturation affects *leaching* and *degassing* in cultivated soil.
Radiological and thermal processes					
41	Decay	Y	Y	Y	Decay is treated as a sink for mother and a source term for the daughter radionuclides.
42	Exposure				Internal and external exposures are calculated from the concentrations of radionuclides in food and the environment (Dose calculations)
43	Heat storage				Heat storage is a property of regolith layers and surface water of the site. Included in parameters that have been derived for the site, accounting for seasonal and climatic variation.
45	Light related processes				The fate of light (e.g. the degree of absorption, scattering and reflection) is determined by site-specific properties. The radiation balance affects the local climate (e.g. temperature and humidity), which in turn is reflected in site-specific parameters.
47	Radionuclide release	Y	Y	Y	Source term for models. See Import and Anthropogenic release above.
Landscape development processes					
48	Change in rock surface location				The isostatic rebound due to the last glaciation is the cause of a continuous to shoreline displacement in Forsmark. The effects of land upheaval on geometries, regolith depth and associated processes are accounted for in several time-dependent site-specific parameters.
49	Sea level change				For sea level change caused by land upheaval see 48) above. Natural fluctuations in sea level change due to e.g. frequency and magnitude of storms are included in parameter values describing the start and end of threshold effects at lake isolation.
50	Thresholding				Aquatic parameters are interpolated between sea and lake values when thresholding effects are expected but not complete (see Section)

Appendix F

Glossary of terms and acronyms used in biosphere model description for SR-PSU

Term/acronym	Definition/description
abiotic	Non-living physical or chemical component or process.
autotroph	Organism that utilises photosynthesis or chemosynthesis to build up organic carbon.
basin	In the SR-PSU terminology, a basin is the drainage area of a biosphere object (cf. below) minus the drainage area of any upstream object. When the basin is below sea level, the basin equals the biosphere object.
BCC	Biosphere calculation case. The BCCs are used for the biosphere part of the radionuclide transport model in the SR-PSU safety assessment.
biosphere	The part of the environment normally inhabited by living organisms. In the context of safety assessments usually more constrained to the surface (see *surface ecosystem*).
biosphere object	A part of the landscape that potentially will receive radionuclides released from a repository.
biotic	Living ecosystem component or process involving living organisms.
CC	Calculation case. CCs are used in the main safety assessment calculations. Biosphere calculation cases (*BCC*, cf. above) are mapped to these main calculation cases.
climate case	SR-PSU describes a set of climate cases, which are possible future climate developments at Forsmark.
climate domain	A climatically determined environment with a specific set of characteristic processes of importance for repository safety.
CR	Concentration ratio. CR values are used to calculate uptake of radionuclides by biota and are defined as the element-specific concentration ratios between the concentrations in biota and in the surrounding media (soil or surface water).
conceptual model	A qualitative description of important components and their interactions.
DEM	Digital elevation model. The DEM describes the topography and bathymetry of the modelled area. It is a central data source for the site characterisation, and is used as input to most of the descriptions and models produced for the surface system.
deterministic analysis	Analysis using single numerical values for key parameters (taken to have a probability of one), which leads to a single value for the result.
discharge points/locations	Locations/areas where groundwater reaches the ground surface. In the safety assessment context, these terms refer to discharge of groundwater that has passed though the repository volume in the bedrock and hence could carry radionuclides to the surface.
DM	Drained-mire farmer. DM refers to a self-sustained industrial agriculture in which wetlands are drained and used for agriculture (both crop and fodder production). It is one of four land use variants considered in SR-PSU for assessment of the most exposed group.
dose	Dose, as used in SR-PSU, refers to the mean annual dose of the most exposed group. The calculated dose accounts for retention of radionuclides in the human body and exposure from daughter radionuclides, as well as radiation sensitivities of different tissues and organs.
dose rate to biota	Dose rate to biota represents mean absorbed dose rates in the whole body of a given radionuclide and is expressed in µGy h^{-1}.
Ecolego tool	Computer software used to model radionuclide transport.
ecosystem model	Conceptual or mathematical representation of an ecosystem, divided into compartments, and its included processes.
effective dose	Effective dose or effective dose equivalent is a measure of dose designed to reflect the risk associated with the dose. It is calculated as the weighted sum of the dose equivalents in the different tissues of the body.
ERICA tool	Computer software used to obtain activity concentrations and radiological effects on different types of non-human biota.
exposure	The act or condition of being subject to irradiation. (Exposure should not be used as a synonym for *dose,* which is a measure of the consequences of exposure.)
	External exposure. Exposure to radiation from a source outside the body.
	Internal exposure. Exposure to radiation from a source within the body.
functional group	A group of organisms with a common function in the ecosystem, e.g. primary producers and filter feeders.
GP	Garden plot household. GP refers to a household that is self-sustained with respect to vegetables and root crops produced through small scale horticulture. It is one of four land use variants considered in SR-PSU for assessment of the most exposed group.
geosphere	Those parts of the lithosphere not considered to be part of the *biosphere*. In safety assessments usually used to distinguish the subsoil and bedrock below the depth affected by normal human activities, in particular agriculture, from the soil that is part of the *biosphere*.

glacial cycle	Used in climate descriptions to denote a period of c 120,000 years that includes both a glacial, e.g. the Weichselian, and an interglacial.
heterotroph	Organism that uses organic compounds produced by autotrophs.
HG	Hunters and gatherers. HG refers to a community that uses the undisturbed surface ecosystems as living space and to obtain food. It is one of four land use variants considered in SR-PSU for assessment of the most exposed group.
hydrodynamic model	In SR-PSU, the hydrodynamic model is the flow model of the sea part of the considered model area, and gives outputs of annual mean flows between adjacent marine basins and water retention times for each individual basin.
hydrological model	The SR-PSU hydrological modelling includes conceptual and mathematical modelling of surface, near-surface and bedrock water flows. The SR-PSU hydrological modelling utilises GIS, as well as the MIKE SHE and DarcyTools numerical modelling tools.
IM	Interaction matrix. The IM is a tool used to ensure that all relevant processes affecting transport and accumulation of radionuclides in the biospehre are considered in the assessment.
infilling	Infilling describes the combined process of sedimentation and organogenic deposition, which turns lakes into wetlands.
IO	Infield outland farmer. IO refers to a self-sustained agriculture in which inland farming of crops is dependent on nutrients from wetlands for haymaking (outland). It is one of four land use variants considered in SR-PSU for assessment of the most exposed group.
K_d	Element-specific soil/liquid partition coefficient defined as the ratio between the elemental concentrations in the solid and liquid phases.
LDF	Landscape dose conversion factor. The LDF is a radionuclide-specific dose conversion factor, expressed in Sv/y per Bq/y, which represents the mean annual effective dose to a representative individual from the most exposed group, resulting from a unit constant release rate to the biosphere of a specific radionuclide. This means that the LDF relates a unit release rate to a dose rate.
LDM	Landscape development model. The LDM is a model at landscape level that describes the long-term development of a landscape. The model is used to describe time-dependent properties of the biosphere objects that are input parameters to the *Radionuclide model of the biosphere*.
mass balance model	The mass balance model calculates the total sum of major sources and sinks for individual chemical elements in the landscape.
most exposed group	In SR-PSU, the most exposed group refers to the group of individuals subjected to the highest exposure during any time period.
NEP	Net ecosystem production. NEP is the sum of gross primary production and ecosystem respiration.
NHB	Non-human biota. SR-PSU includes an assessment of the effects of potential future radionuclide releases on NHB.
NPP	Net primary production. The balance between gross primary production and plant respiration.
probabilistic analysis	Mathematical analysis of stochastic (random) events or processes and their consequences. Since the input is described in stochastic terms, also the output is stochastic (e.g. in the form of probabilities or distributions).
Radionuclide model	Model used to calculate radionuclide inventories in different compartments of the biosphere, radionuclide fluxes between the compartments and radionuclide concentrations in environmental media (soil, water, air and biota). The radionuclide model utilises the Ecolego modelling tool.
regolith	All matter overlying the bedrock is collectively denominated regolith. This includes both minerogenic and organogenic deposits, as well as anthropogenic landfills.
RDM	Regolith depth and stratigraphy model. The RDM interpolates observation points of analysed vertical distribution of regolith into a three-dimensional model of regolith extension.
RLDM	Regolith-lake development model. The RLDM is divided into a marine module that predicts the sediment dynamics caused by waves, and a lake module that predicts infilling of lakes. The model output is the regolith distribution and thickness of different strata at the studied time steps.
SDM	Site descriptive model. The SDM is a multi-disciplinary description of the studied site, including both qualitative and quantitative information, which is based on both direct observations and modelling studies.
SFR	The existing final repository for short-lived radioactive waste.
SR-PSU	The safety assessment of the existing SFR facility (SFR 1) and its planned extension (SFR 3).
sub-catchment	In the present context, defined as the drainage area of a biosphere object minus the drainage area of the inlets to the object.
surface ecosystem	In the safety assessment context, surface ecosystem refers to the part of the analysed system that is above the bedrock, with all its abiotic and biotic processes and features.
terrestrialisation	The transformation of an aquatic ecosystem (marine or limnic) to a terrestrial ecosystem.
watershed	In the present context defined as the drainage area of a biosphere object.

Appendix G

Ecosystem properties and fluxes of surface water during transition periods of natural landscape succession

All biosphere objects are presently submerged under the sea. However, as a consequence of the isostatic rebound the bottom of coastal sea basins will ultimately rise above the sea level in the Forsmark area. Thus, the central parts of sea basins will be transformed into lake and/or wetland areas, and physical, chemical, biological and hydrological properties of the area may change in response to the landscape development.

For illustrative purposes, the continuous development of a biosphere object has been divided into two periods, namely the submerged period (when the object is covered by the sea) and the land period (when the object is above sea level). For biosphere objects where the isolation of a coastal bay results in a lake (116, 121_1,157_1, 159, 160) the submerged period ends (and the land period starts) when the lake threshold reaches the average sea level (Figure G-1A, left panel). For objects that lack a lake threshold (121_2 and 157_2), and thus develop into land areas without a lake stage, the submerged period ends when the area is fully emerged above the average sea level (Figure G-1B, left panel). Moreover, the land period can be divided into an early period when the wetland is expanding in the object, and a late stage when wetland has reached its full area extension and lakes has been completely filled by mire vegetation.

The continuous development of the individual biosphere objects is partly described with dynamic (i.e. time-dependent) model parameters. For example, the depth and volume of water bodies, the area of mire, the potential depth of the peat layer, and the rates of horizontal surface water fluxes, of sedimentation, of resuspension, and of aquatic primary production, are described with time-dependent model parameters (see Parameter report). However, for other properties that are affected by ecosystem transition, parameter values are only available for periods with stable conditions (i.e. stable sea, lake or mire conditions).

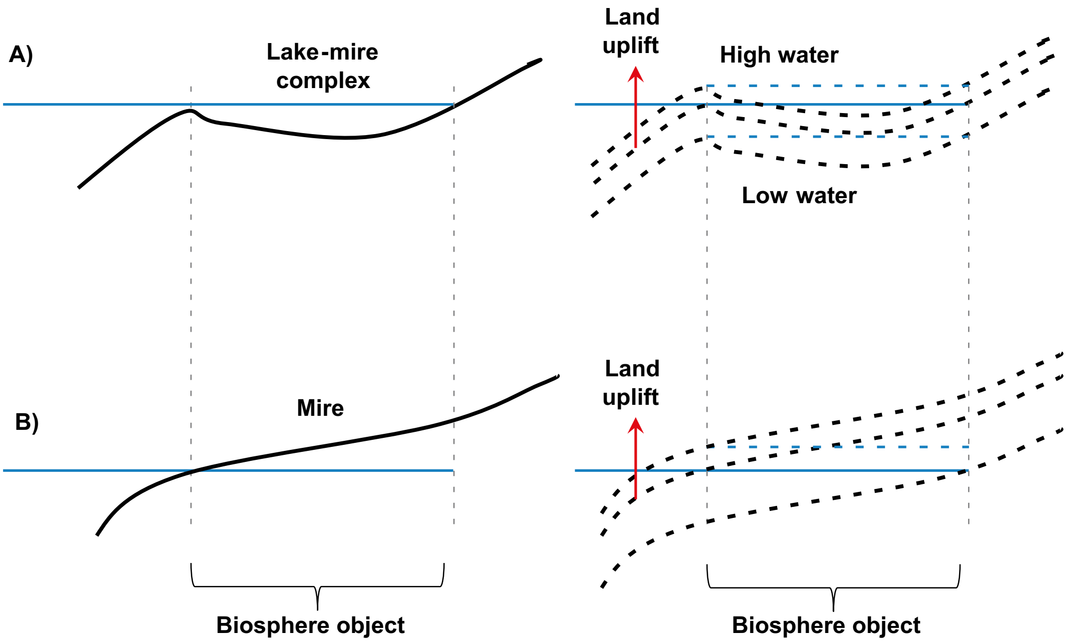

Figure G-1. *Biosphere objects in transition. A) The transition of a sea bay into a lake-mire complex. The submerged period ends, and the land period starts, when the lake threshold is at the sea level (threshold_isolation) (left). Aquatic ecosystem conditions are assumed to be intermediate in the period between isolation at extreme low water (threshold_start) and isolation at extreme high water (threshold_stop) (right). B) The transition of a sea bay into a mire area without a lake basin. The submerged period ends, and the land period starts, when the object is fully emerged at normal sea level (threshold_isolation) (left). Aquatic ecosystem properties are assumed to be intermediate in the period between the time when the first land emerges and the time when the object is completely emerged even at extreme high water levels (threshold_stop) (right).*

Thus, for the assessment purpose we assume that aquatic ecosystem properties during the transition from a sea basin to a fully isolated lake (or a mire) will be intermediate to those of marine and freshwater conditions, respectively. The start of the transition period was defined as the point in time when the lake threshold emerges above sea level during periods of extreme low water (threshold_start), whereas lake isolation was considered to be completed when the threshold is above sea level during periods of extreme high water (threshold_end). For objects that develop into wetland areas without a lake stage, the transition period for the aquatic ecosystem starts when the first land emerges out of the sea (threshold_start) and ends when the object is completely emerged, even at extreme high water levels (threshold_stop).

Aquatic parameters that are affected by the transition from a sea to a lake ecosystem include: fish and crayfish production (*prod_edib*), the minimum depth for fish production (*z_min_prod_edib_fish*), the dry weight content of fish biomass (*f_DW_FW_fish*), the equilibrium concentration in biota (*CR* for fish and primary producers, respectively), the fraction of litter that is refractory (*f_refrac* for macrophyte, microphyte and plankton, respectively), the mineralization rate of particulate matter in water and in aerobic and anaerobic organic sediments (*minRate*), the depth, density and porosity of the biologically active sediments (*z_regoUp_aqu, dens_regoUp_aqu, poro_regoUp_aqu*), the concentrations and sorption coefficients of particulate matter (*conc_PM, Kd_PM*) and dissolved inorganic carbon (*conc_DIC_aqu*), the fraction of DIC in the form of CO_2/H_2CO_3 (*f_H2CO3*), the solubility coefficient of CO_2 in water (*solubilityCoef_aqu*) and the piston velocity (*piston_vel_aqu*) (Grolander 2013). Consequently, the values of these parameters were linearly interpolated in the transition period between marine and freshwater conditions:

$$X_{trans}\,(t) = X_0 + (\Delta X_{trans}/T_{trans})t, \qquad \text{(G-1)}$$

where

X_{trans}	is the parameter value in the transition period.
X_0	is the parameter value at the start of transition, e.g. during the submerged period
ΔX_{trans}	is the difference between the typical parameter values before and after transition (i.e. the difference between values during freshwater and sea conditions)
T_{trans}	is the length of the transition period [y], and
t	is the time from the start of the transition period (i.e. simulation time less the time of transition start) [y].

For the transition of aquatic ecosystem X_0 equals the parameter value during submerged conditions (parameter name with subscript sea), ΔX_{trans} equals the difference in the parameter value between fresh water (parameter name with subscript sea) and submerged conditions, T_{trans} equals the transition period for lake isolation (threshold_stop – threshold_start).

Parameter values for ecosystem properties of the mire (e.g. net primary production, the fraction of litter that is refractory, the mineralization rates, the depth of peat layers, the water concentrations of DIC, the fraction of DIC in form of CO_2, the solubility coefficient of CO_2 in water and the piston velocity) are used for the terrestrial part of the object when land first appears above average sea level (threshold_land). This happens either when the lake threshold reaches the average sea level (Figure H-1.A, left panel), or when the first land emerges above average sea level (Figure G-1.B, right panel lower dotted line).

The vertical and horizontal fluxes of water in a biosphere object are strongly influenced by the development of the landscape. For example, near-surface vertical fluxes increase when the shoreline is passing a discharge area and the exchange between the water column and surface sediments may be an order of magnitude larger in the lake than in the original sea basin (Werner et al. 2014). On the other hand, the lake thresholds reduces the horizontal flux of water, which may be two or three orders of magnitude less in the isolated lake as compared to the water exchange in the marine bay (Werner et al. 2014).

The exchange of water between marine basins were estimated with MIKE 3 every thousand years, and combined with MIKE SHE estimates of surface water flow at isolation, to form a time-dependent parameter for each biosphere object (see Grolander et al. 2013). For the time between estimates, the flux rates were estimated with linear interpolation (Equation G-1). Vertical groundwater fluxes between regolith layers were modelled for stationary submerged and land conditions. That is, groundwater fluxes representing 1) the submerged period, 2) conditions following lake isolation, and 3) the period when mire expansion is completed, respectively, were modelled with MIKE SHE (Werner et al. 2014).

For the assessment, it was assumed that vertical ground water fluxes are stable during the submerged period and during the land period after the mire development has been completed, and intermediate during the transitions between stable states. Consequently aquatic fluxes were linearly interpolated (Equation G-1) over the time between the start and completion of lake isolation (for objects developing into a lake). For objects that did not go through a lake stage, vertical hydrological fluxes for both the aquatic and the mire ecosystems (and the exchange of surface water between the two) were interpolated between the time when land first emerged and the time when the object was fully emerged out of the sea (Table G-1). Moreover, vertical groundwater fluxes in the mire part of the object, and the exchange of surface water between ecosystems, were linearly interpolated in the period between lake isolation/land emergence and the time when the mire expansion was complete and the object was fully covered by mire vegetation with or without a stream (Table G-1). The parameters describing the start and end of transition periods are described in Grolander et al. (2013).

In biosphere objects that develop a lake, all horizontal advective transport out of the object is directed from the water body (lake or stream, see Chapter 4). However for biosphere objects that lack a lake stage, this simplifying assumption yields unrealistic flow paths and water turnover rates when the water body is disappearing (i.e. the area and depth are approaching zero). Thus, for these objects (157_2, 121_2) it was assumed that a fraction of the flux of water from the terrestrial part of the object will reach the downstream object directly (without passing through the water body) during the transition from submerged to land conditions. The fraction of land in the object (f_{ter}) was used to partition the advective transport from the mire object to the water body (within the object) and to the downstream object. Thus the advective transport from surface peat in object 157_1 during this period is expressed as:

$$Adv_{ter,downstream} = Adv_{regoUp,157\_1} = WF_{157\_1} f_{ter} AC^{D}_{regoUp,ter}$$

$$Adv_{Up,\ ter,\ Water} = WF_{157\_1} AC^{D}_{Up,ter} (1-f_{ter}) \tag{G-2}$$

where

WF_{157_1} is the vertical advective flux of water from upstream to downstream object (Equation 4-10) [$\text{m}^3\ \text{year}^{-1}$]

$AC^{D}_{regoUp,ter}$ is the pore water activity concentration of the terrestrial upper regolith [Bq m^{-3}], and

f_{ter} is the fraction of the object that is mire.

The total horizontal transport from the upstream object to the downstream object is then simply the sum of the transport from the terrestrial and the aquatic parts of the object:

$$\begin{aligned} Adv_{downstream} = {} & Adv_{ter,downstream} + Adv_{aqu,downstream} = \\ & = WF_{157_1}(AC^{D}_{Up,ter}\ (f_{ter}) + (1-f_{ter})(AC_{Water,} \\ & + AC_{Water,org})) \end{aligned} \tag{G-3}$$

where

AC_{Water} is the activity concentration in water in inorganic form [Bq m^{-3}], and

$AC_{Water,org}$ is the activity concentration in organic particulate matter [Bq m^{-3}].

Table G-1. Transition periods during which vertical flux rates of groundwater were interpolated. During each transition period the rates of a number of groundwater flux (column q_i) in the biosphere object changed linearly between the values at the start (column Start) and end (column End) of the transition period (Equation G-1, for details see text).
1 = biosphere object developing through a lake period. The object specific parameters describing the start and end of the transition periods are: 2 = *threshold_start*, 3 = *threshold_isolation,* and 4 = *threshold_end* (Grolander 2013).

		Transition Period		**Water flux q_i (m year^{-1})**		
Ecosyst	**Transition**	**Start**	**End**	**Start**	**End**	**q_i**
Aquatic	Sea to Lake[1]	Lake isolation at low water[2]	Lake isolation[3]	$q_{i,\mathrm{sea}}$	$q_{i,\mathrm{lake,iso}}$	Low_GL_aqu, GL_low_aqu, GL_PG_aqu, PG_GL_aqu, PG_Up_aqu, Up_PG_aqu, Up_water_aqu Water_Up_aqu
Aquatic and Mire	Sea to Mire	First land above water[2]	All land above water[3]	$q_{i,\mathrm{sea}}$	$q_{i,\mathrm{ter,iso}}$	Low_GL_aqu/ter, GL_Low_ aqu/ter, GL_PG_ aqu/ter, PG_GL_ aqu/ter, PG_Peat_ aqu/ter, Peat_PG_ aqu/ter, Up_water_aqu, Water_Up_aqu, Up_Peat_ter, Peat_Up_ter
Mire	Mire expansion	Lake isolation[3]/ All land above water[3]	Mire expansion completed[4]	$q_{i,\mathrm{ter,iso}}$	$q_{i,\mathrm{ter,end}}$	GL_Low_ter, Up_Peat_ter, Peat_Up_ter, Peat_PG_ter, PG_Peat_ter, PG_GL_ter, Low_GL_ter, GL_PG_ter, Water_Up_ter, Up_Water_ter