Journal of Environmental Radioactivity 255 (2022) 107019

Contents lists available at ScienceDirect

Journal of Environmental Radioactivity

journal homepage: www.elsevier.com/locate/jenvrad

On the inclusion of forest exposure pathways into a stylized lake-farm scenario in a geological repository safety analysis

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ARTICLE INFO

Keywords:
Biosphere analysis
Farm
Forest
Lake
Radionuclide
Sensitivity analysis

ABSTRACT

Geological disposal of radioactive waste has been recognized as the 'reference solution' to ensure the safety required for the present and future society and environment. To study the possible exposure pathways from groundwater to humans, radioactive transport modelling is used. One of the ecosystems that may play a significant role when assessing the dose conversion factor (i.e. the dose resulting from a nominal release of 1 Bq/year of each radionuclide) for humans is forest. In this paper we have developed a model of a lake-farm system with a forest component. The biosphere system used in this study represents a typical agricultural scenario in Finland, amended with a typical forest. A lake is assumed to form due to post-glacial land uplift. The main features of this future lake have been obtained from our probabilistic shoreline displacement model. Both deterministic calculations and sensitivity analysis were carried out to simulate the model. The deterministic simulation demonstrates the behaviour of the studied radionuclides (^{36}Cl, ^{135}Cs, ^{129}I, ^{237}Np, ^{90}Sr, ^{99}Tc and ^{238}U) and the proportions of different exposure pathways to humans. Particularly for ^{135}Cs and ^{129}I, forest pathways make a notable contribution to the dose conversion factor. The sensitivity analysis was done using two methods: EFAST and Sobol'. With both methods, the parameters related to the farm contribute the most to the variance of the dose conversion factor for humans. The study demonstrates that the exposure pathways related to forest products may make a considerable contribution to the dose conversion factor in a lake–farm–forest system. It is also confirmed that an advanced sensitivity analysis for a radionuclide transport and dose assessment model on such a landscape scale is feasible even with moderate computational efforts.

1. Introduction

Safety management and disposal of radioactive waste is part of the life cycle of nuclear energy. For waste disposal, geological facilities have been recognized as the 'reference solution' (e.g., NAS, 1957; IAEA, 1997; NEA, 2008; ICRP, 2013); the higher the radioactivity level, the deeper the disposal, in order to offer the very long-term passive safety required for society and the environment in both present and future (IAEA, 2006; ICRP, 2013). The safety of geological disposal relies on man-made barriers and the host formation isolating, limiting and/or retarding releases of radioactivity from the repository to the surface ecosystems where people and other biota could be exposed to such contaminants. Nevertheless, it is also important to analyse the potential radiological impacts of geological waste disposal because of the large inherent uncertainties over the very long timescales involved, including the potential for various kinds of environmental changes and disturbances (e.g., ICRP, 2013; Posiva, 2013b; STUK, 2018).

Various methods of sensitivity analyses are commonly used to explore, evaluate and (partially) validate radiological impact assessment models (e.g., Capouet et al., 2009). In addition to so-called local sensitivity analyses exploring the impact of a change in a single or a few parameters at a time, more advanced probabilistic global sensitivity analyses are often employed also in this purpose on repository safety assessment and/or radiological impact models (Li and Mohanty, 2001; Avila et al., 2006; Rasmuson et al., 2007; Broed, 2008; Spiessl et al., 2012; Spiessl and Becker, 2015; SKB, 2019). Furthermore, they have been applied also to models connecting a few or several ecosystem-specific modules together (Reid and Corbett, 1993; Pröhl and Müller, 1996; Ekström and Broed, 2006; Broed, 2007; Olyslaegers et al., 2011; Avila et al., 2013; Kupiainen, 2014; Kupiainen and Nummi, 2017; Broed et al., 2021; Nicoulaud-Gouin et al., 2022), although this has often been attributed computationally intensive or otherwise inconvenient,

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https://doi.org/10.1016/j.jenvrad.2022.107019
Received 31 May 2022; Received in revised form 2 September 2022; Accepted 8 September 2022
Available online 27 September 2022

particularly with increasing size of the model (Avila et al., 2006, 2013; Broed, 2008; Kupiainen and Nummi, 2017).

Forest ecosystems may play a significant role in radionuclide transport modelling from groundwater to humans. Forest ecosystems consume a lot of water and the radionuclides that are transported via groundwater and taken up by the relatively deep roots are accumulated in the forest biomass during the evapotranspiration process. Humans consume berries, mushrooms and game meat and thus radionuclides may be transported to humans. Also, wood burning may generate radioactive soot and dust that could be inhaled during the ash removal process from fireplaces.

One of the earliest forest radionuclide transport models was developed by following the propagation of ^{137}Cs injected into tulip poplar trees (Olson, 1965). Later this analysis was extended to include soil samples, rainfall and runoff estimations (Waller and Olson, 1967). Raines et al. (1969) developed two- and eight-compartment models to estimate the transport of 31 radionuclides in nuclear detonations near rainforests and seawater. The Chernobyl accident led to several developments in biosphere and forest modelling, testing and validation, mainly for ^{137}Cs (Antonopoulos-Domis et al., 1990, 1996; Belli et al., 1996; Mamikhin, 1995; Mamikhin et al., 1997; Mamikhin and Klyashtorin, 2000; Shaw et al., 1996, 2005; Schell et al., 1996a,b; Yoshida et al., 2004). Further, modelling forest contamination with ^{99}Tc related to a spent nuclear waste repository is addressed in Garten et al. (1986) and Garten (1987). Forest models have also been developed and examined in dissertations (e.g., Avila Moreno, 1998; Bostock, 2004) and in nuclear waste management company reports (Avila, 2006; Posiva, 2012; Kupiainen, 2014; Saetre et al., 2017). Also, the IAEA has proposed a methodology for modelling forest ecosystems (IAEA, 2002) with a later improved data set (Calmon et al., 2009; IAEA, 2010).

To address the contribution of forest pathways to the radiation dose rate for humans at landscape level in a reasonably simple manner, rather than for the forest alone, we developed our earlier models of a lake–farm system (Pohjola et al., 2016, 2019) to include a forest component. This was largely implemented by adapting the earlier models of Avila (2006) and Stenberg and Rensfeldt (2015), as described below. The application of a sensitivity analysis to this developed landscape was another of our aims.

In simulations, forests can be thought of as a single model for estimating the radionuclide dose rates for humans, although a more realistic radiation dose rate can be obtained when the forest model is combined with, for example, lake-farm scenario models. The initial model is a simple lake-farm model, where all household water is taken directly from the lake, rather than a well, located near a nuclear waste repository in bedrock (Pohjola et al., 2016). The model was extended to include the effect of lake bottom sediments in radionuclide transport in Pohjola et al. (2019). The extended model contains three sediment layers (till, mixed sediment and clay) which are assumed to continue under the forest area in the proposed model. This is the common stratigraphy in the region, as a result of the glacial erosion of the bedrock and redeposition of the material under glacial, proglacial, periglacial, marine and recent geological conditions (Tulkki, 1977; Winterhalter, 1992; Rantataro, 2001). The assumption comes from the idea that, after Weichselian glaciation, fine-grained bedrock material, mainly clays, was spread all over Finland and layered, on top of the till formations from the glacial abrasion of the bedrock, before the forest and overburden appeared. The model used in the current work contains both the scenarios presented in Pohjola et al. (2016, 2019) extended with a forest model based on Avila (2006), where game, berries and mushrooms are used for human food production. Household animal grass and hay consumption is already taken into account in Pohjola et al. (2016). The radionuclides released from wood burning are also included in the model. Throughout this paper, the BIOMASS-6 (IAEA, 2003), the interim version of its upgrade BIOMASS-2020 (Lindborg, 2018; Thorne et al., 2022), and IAEA (2020) approaches have been applied.

2. Assessment context

The current research aims at a stylized assessment, independent of the Olkiluoto spent nuclear fuel repository programme (e.g., Posiva, 2013b). The repository for spent nuclear fuel, a repository for low- and intermediate-level radioactive maintenance waste and three power stations are located on the island of Olkiluoto in Eurajoki, western Finland. However, the general context is the same, allowing comparisons of the differing modelling approaches, including sensitivities to different assumptions. In addition, our approach is generic so as to be directly applicable also to other geological radioactive waste repositories in Finland or more widely on the Fennoscandian Shield. As such repositories in Finland and Sweden have a coastal location with considerable post-glacial land uplift (at Olkiluoto, approximately 30 m vertically over the next ten millennia (Pohjola, 2014)), the regulatory guide (STUK, 2018) also generally prescribes "an assessment period, during which the radiation exposure of humans can be assessed with sufficient reliability, and which shall extend, at a minimum, over several millennia", which also allows sufficient time for radionuclides to migrate and accumulate in the modelled system. This has been interpreted here as the nominal time frame of 10 000 years used for our simulations, which is consistent with other assessments (e.g., Posiva, 2013a,b). This uplift means that the coastline will move westwards by about 10–15 km and several new lakes will be formed. Specifically, at Olkiluoto, it has been projected that a lake will form in 3000–4000 years' time because of post-glacial uplift. For recognizability, this future lake has been named Lake Liponjärvi (e.g., Posiva, 2013a,b). We too have used this lake as an assessment case because it is located near the radioactive waste repositories at Olkiluoto (e.g., Vieno and Suolanen, 1991; Posiva, 2013b) and because the estimated volume and expected productivity of the lake is large enough to satisfy the needs of a small hypothetical farmer-forager community relying on these resources. The main features of this future lake, such as its volume, area, in- and outflow, have been obtained from our probabilistic shoreline displacement model (Pohjola, 2014) and comprise a stylized representation of the sediments in three layers (compartments): till, glacio-aquatic mixed sediment and clay that also extend into the catchment soils (Rantataro, 2001; Posiva, 2012; Pohjola et al., 2019). The lake catchment also hosts relatively low-lying terrestrial areas upon which it is reasonable to assume that there would be forest areas receiving deep groundwater input. The addition of the forest ecosystem extends the model to include dose pathways to humans as a result of consuming game meat, berries and mushrooms and from burning wood. Because of the uncertainties inherent in the analysis of such a long period, the regulatory guide (STUK, 2018) advises that climate, ecosystems, human habits, diet and metabolism are assumed to remain similar to those of the present.

It is assumed that radionuclides released from the repository would enter the lake water and forest through the sediments/soils, including the retention potential in these layers. The release rate is nominally set at 1 Bq/year for each nuclide (^{36}Cl, ^{135}Cs, ^{129}I, ^{237}Np, ^{90}Sr, ^{99}Tc and ^{238}U), which facilitates easier comparison between models and assessments. The chosen suite of radionuclides covers a range of biogeochemical behaviour and includes nuclides both in earlier assessments for the site (e.g., Vieno and Suolanen, 1991; Posiva, 2012; Pohjola, 2014; Pohjola et al., 2016, 2019) and those of general interest (e.g., Chen et al., 2006). The ingrowth of ^{238}U progenies has not been included in the simulations here because the decay chain calculations would pose considerable practical challenges to the sensitivity analyses; ^{238}U as such has been included, however, for a wider spread of the types of biogeochemical behaviour.

3. Biosphere system identification and justification

The biosphere system used in this study represents a typical agricultural scenario in Finland with the addition of a typical forest. It is assumed that a lake will form because of the post-glacial land uplift west

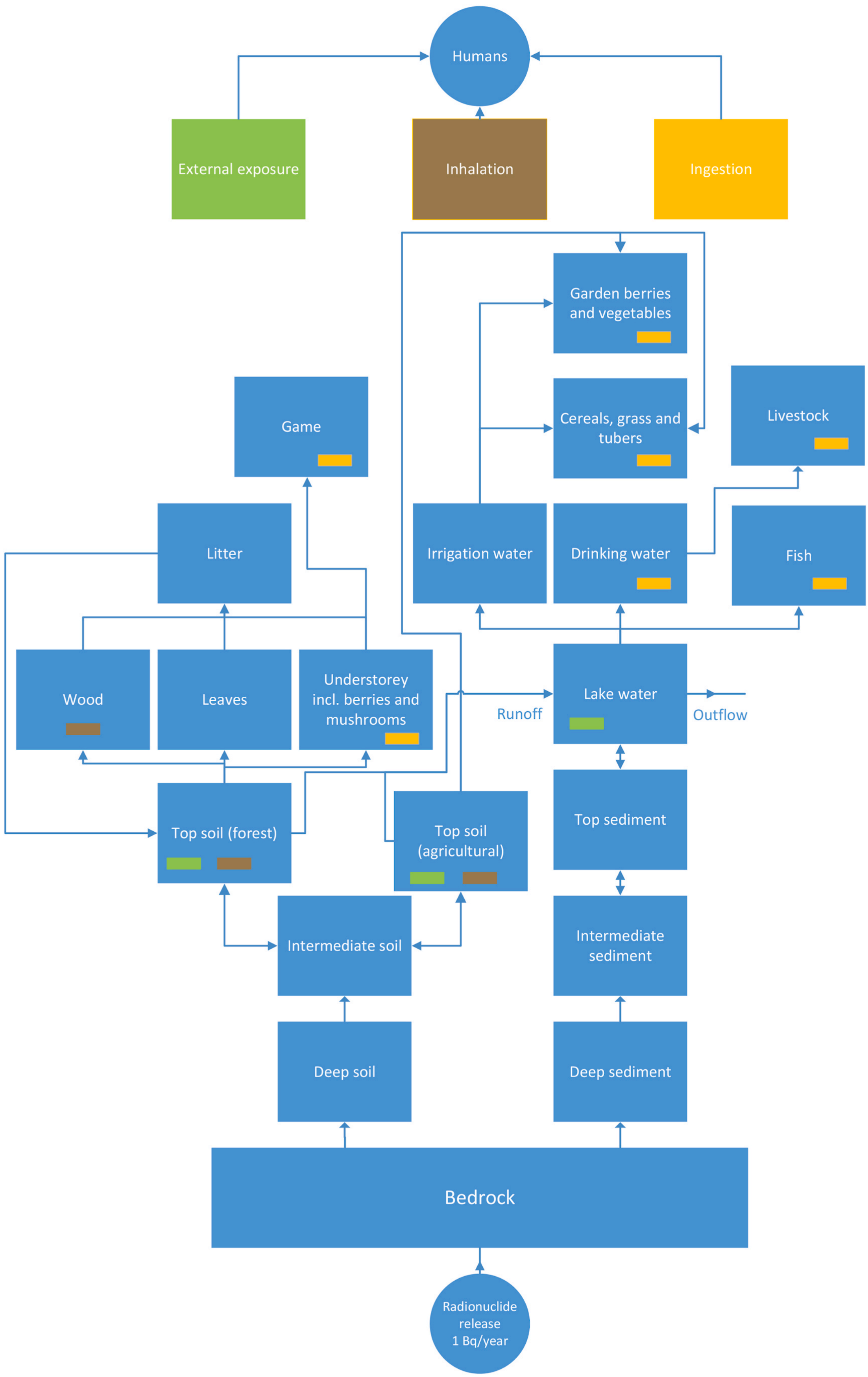

Fig. 1. Radionuclide transport paths in the model. The contributors to the three exposure pathways at the top have been identified with the respective colors in the lower part of the diagram. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

of the present Olkiluoto Island, as supported by the underlying landscape development modelling (Pohjola, 2014; Posiva, 2013b). When considering Finnish lakes formed due to shoreline displacement and isolation from the Baltic Sea, it has been found that they usually have a till layer above the bedrock overlaid by glacial, transition and post-glacial clays, and recent mud (Winterhalter, 1992). Groundwater finds the easiest path through the sediments into the lake, determined by the hydraulic properties of the sediment and the groundwater pressure field. Acoustic-seismic studies in the vicinity of Olkiluoto (Rantataro, 2001) have identified the different sediment layers and their thicknesses as well as the elevation of the bedrock surface. Based on these studies, deposits of till, glacio-aquatic mixed sediment, post-glacial and recent clay are modelled as sediment compartments in this work. The lake water is assumed to be used as drinking water for people and animals, and irrigation water for cereals, potatoes, vegetables, berries and animal fodder (pasture grass). Vegetables are further divided into leguminous vegetables, root vegetables and other vegetables. Regarding the forest, contaminated groundwater is assumed to travel through the soil layers originating from the former lake sediments into plants and mushrooms, and further into game; humans would then consume game meat, berries and mushrooms (Avila, 2006). The use of firewood is included in this model, based on the estimations presented in Stenberg and Rensfeldt (2015). The human diet used here is based on the national nutrition surveys conducted by Helldán et al. (2013) and Valsta et al. (2018). It is assumed to include the consumption of fish caught from the lake, livestock products contaminated through irrigated fodder and drinking water. This scenario is conceptually similar to that used by Kupiainen (2014). With the differences of the contaminated water source being a lake instead of a well and of adding fish and forest products to the diet, this scenario is similar to the agricultural well system presented in Hjerpe and Broed (2010). Our scenario also includes probabilistic modelling and sensitivity analysis in the calculations.

The radionuclide transport pathways to humans in the modelled biosphere system are illustrated in Fig. 1.

4. Definition of potentially exposed group

Exposure resulting from geological waste disposal will occur far in the future, if at all. Based on the uncertainties inherent in making predictions for the distant future, the International Commission on Radiological Protection (ICRP) recognizes that, in safety assessments of (deep) radioactive waste repositories, it would be adequate to estimate the dose conversion factor or risk to an adult person and to consider this person as representative of the whole population (ICRP, 2013). Therefore, as well as for the general sake of simplicity, only adult humans are considered in our model. The male dietary profile is selected from national statistics when exploring the potential sensitivity of the dose conversion factor to dietary assumptions because men generally have higher consumption rates, thus retaining the conservativeness of the model. A dose constraint of 0.1 mSv per year has been set for the most exposed individuals near a radioactive waste repository in Finland (STUK, 2018). However, in our simulations only dose conversion factors (i.e. the dose resulting from a nominal release of 1 Bq/year of each radionuclide) are calculated, which are not directly comparable with the dose constraint. As radiation exposure occurs through different pathways, the representative person can further be considered as an average individual in a self-sustaining family that uses the local resources at the repository site. In other words, in the present work too, the potentially exposed group is defined so that the dose conversion factor for the average individual implied by the regulations is equal to the dose rate calculated here for an adult male individual who satisfies all his needs for dietary items and drinking water from the resources contaminated through the release of radioactivity in the lake–farm–forest system analysed, according to statistical average rates of consumption and other types of exposure.

5. Biosphere model development and calculation

5.1. Model formulation

The multi-compartment model used in this study is presented in Fig. 1. The mathematical formulation of the model has been detailed in Supplement 2.

The bottom sediments of the lake have been divided into three compartments: deep sediment (till), intermediate sediment (glacio-aquatic mixed sediment) and top sediment (clay). The top sediment compartment is an aggregate of different types of clay for simplicity and because of the lack of data available in the literature for the different clay types. The mud layer above the clay layer has been omitted because its influence on radionuclide transport is negligible. The mud layer is typically thin and, for land areas, is removed by littoral exposure when rising from the sea. The thicknesses of the sediment layers in the area of interest were interpolated from seafloor acoustic-seismic data (Rantataro, 2001) using the thin plate spline method that has earlier been shown to be appropriate for estimating the topographical features at the Olkiluoto site (Pohjola, 2014). From these data, the mean value of the thickness of each sediment layer within the projected future Lake Liponjärvi was used for the reference-case simulations and the distribution ranges of the thicknesses for the sensitivity analysis. For the agricultural and forest soil compartments, the same types and thicknesses were used as for the lake bottom sediments. This follows the conceptual understanding of the site (e.g., Posiva, 2013). Lake volume, outflow and water-depth distributions were obtained from the UNTAMO simulation tool as described by Posiva (2013b) and Pohjola (2014). These distributions represent the temporal variation in the water depth, volume and turnover of Lake Liponjärvi over its span of development. Sedimentation from the catchment area will reduce the size of the lake according to the method described in BIOMOVS (1989) and this property of the model was calculated using the MATLAB tool (Mathworks, 2021). For simplicity, the method assumes that the depth profile of the lake is triangular.

Water fluxes from the bedrock to the deep sediment compartments and from the agricultural and forest soil to lake water compartments were adopted from Posiva (2014) (see also Table S3.32 in Supplement 3). These fluxes have been obtained from a surface hydrological and near-surface geohydrological model (Posiva, 2012) which uses the same description of the sediment layers (Rantataro, 2001). The whole of the lake water volume is represented by a single compartment, assuming that the radionuclide release will mix 'instantly' and homogeneously throughout the lake (because its water turnover time is 0.8 years). A constant release rate of 1 Bq/year of each radionuclide from the bedrock repository is assumed to enter the deep sediment compartment and deep soil compartment uniformly. Annual precipitation, evapotranspiration and catchment area water fluxes have been taken into account when estimating the outflow flux (cf. Fig. 1). Runoff and interflow fluxes between the rest of the lake catchment and the forest soils, agricultural soils and the lake are assumed to be negligible for conservativeness in respect of the radionuclide-diluting fluxes. Mathematical descriptions of the interactions affecting the sediment and agricultural soil layers and the lake water (advection, bioturbation and diffusion, sedimentation, resuspension from the top sediment, and the outflow from the lake) have been taken from Posiva (2014); Karlsson and Bergström (2000) and Vieno and Suolanen (1991), as already described by Pohjola et al. (2016, 2019). The transfer of radionuclides to crops caused by root uptake and retention of irrigation water, the transfer to livestock from contaminated fodder and drinking water, and the transfer of radionuclides via the intake of drinking water by humans are modelled using the transfer functions described in Hjerpe and Broed (2010) and Posiva (2014). The equation for bioaccumulation from water to fish is taken from Vieno and Suolanen (1991). The interactions in the forest compartments were adopted from Avila (2006). The different radionuclide transport coefficients of forest soil to game were determined using the ERICA

Table 1
References for the parameter values in dose conversion factor calculations. The parameters described by a probability distribution were included in the first phase of the sensitivity analysis (Morris).

Deterministic parameters	
Diffusion coefficients for water	Ohlsson and Neretnieks, 1997
Dose coefficients for external exposure	Eckerman and Ryman, 1993
Dose coefficients for ingestion and inhalation	ICRP, 2012
Forest properties	Avila, 2006
Leaf area indexes	Hjerpe and Broed, 2010
Miscellaneous parameters	Hjerpe and Broed, 2010; Vieno and Suolanen, 1991
Radionuclide half-lives	Avila et al., 2003
Soil to animal concentration ratios (forest)	Brown et al., 2016
Total irrigation amounts	Hjerpe and Broed, 2010
Water diffusion coefficients	Ohlsson and Neretnieks, 1997
Wood combustion activity concentrations	Stenberg and Rensfeldt, 2015
Yield values for crops	Hjerpe and Broed, 2010
Parameters described by a probability distribution	
Food intake rates	Helldán et al., 2013; Valsta et al., 2018
Lake properties	Pohjola, 2014
Sediment/soil layer properties	Rantataro, 2001
Soil to plant concentration ratios (agricultural)	IAEA, 2009
Soil to plant concentration ratios (forest)	Avila, 2006
Solid-liquid distribution coefficients	Karlsson and Bergström, 2000; Sheppard et al., 2009
Transfer coefficients (animal intake)	Karlsson and Bergström, 2000
Translocation factors (root crops)	Karlsson and Bergström, 2000
Water to fish concentration ratios	Karlsson and Bergström, 2000
Water fluxes	Posiva, 2014

software (Brown et al., 2016). Firewood is assumed to be obtained from the local forest near the dwelling site and the dose rates from burning it have been calculated based on Stenberg and Rensfeldt (2015).

5.2. Data selection

The data sources used in the calculations are presented in Table 1. The parameter values are presented in more detail in Supplement 3.

The average Finnish daily food intake rates presented by Helldán et al. (2013) and Valsta et al. (2018) were converted to annual values and were assumed to follow a normal distribution. Since the updated values of Valsta et al. (2018) did not contain standard deviations, they were assumed to be the same as those reported by Helldán et al. (2013). For the soil to plant transfer factors, log-normal distributions were assumed, and geometric means as well as geometric standard deviations were used in the calculations. When these were not available, the arithmetic mean and standard deviation were used. For the data sets that did not have probabilistic data available, such as some translocation factors from animal intake to animal products, log-normal distribution and a geometric standard deviation of 3.2 based on Sheppard (2005) was assumed for the sensitivity analysis. The water fluxes between the sediment compartments (mean, maximum and minimum values) were taken from Posiva (2014), and are based on surface and near-surface hydrological modelling (Posiva, 2013). Triangular distributions determined by these mean, maximum and minimum values were used in the sensitivity analysis.

5.3. Modelling platform

The modelling platform used for the deterministic dose conversion factor calculations and the sensitivity analysis was the Ecolego software tool (AFRY AB, version 8.0.6; (e.g., Avila et al., 2003)). Ecolego can be used for creating dynamic compartment-based models and running simulations deterministically or probabilistically. The built-in database of radionuclides was used here so the radioactive decay was taken care of automatically in the dose conversion factor assessment calculations. The computations in this study used an implicit multistep solver of a variable order between 1 and 5, which is embedded in the Ecolego software. The multistep solver is based on the numerical differentiation formulas described by Shampine and Reichelt (1997). The Mathworks Matlab (version R2021A) simulation platform (Mathworks, 2021) was used for auxiliary data processing and visualization.

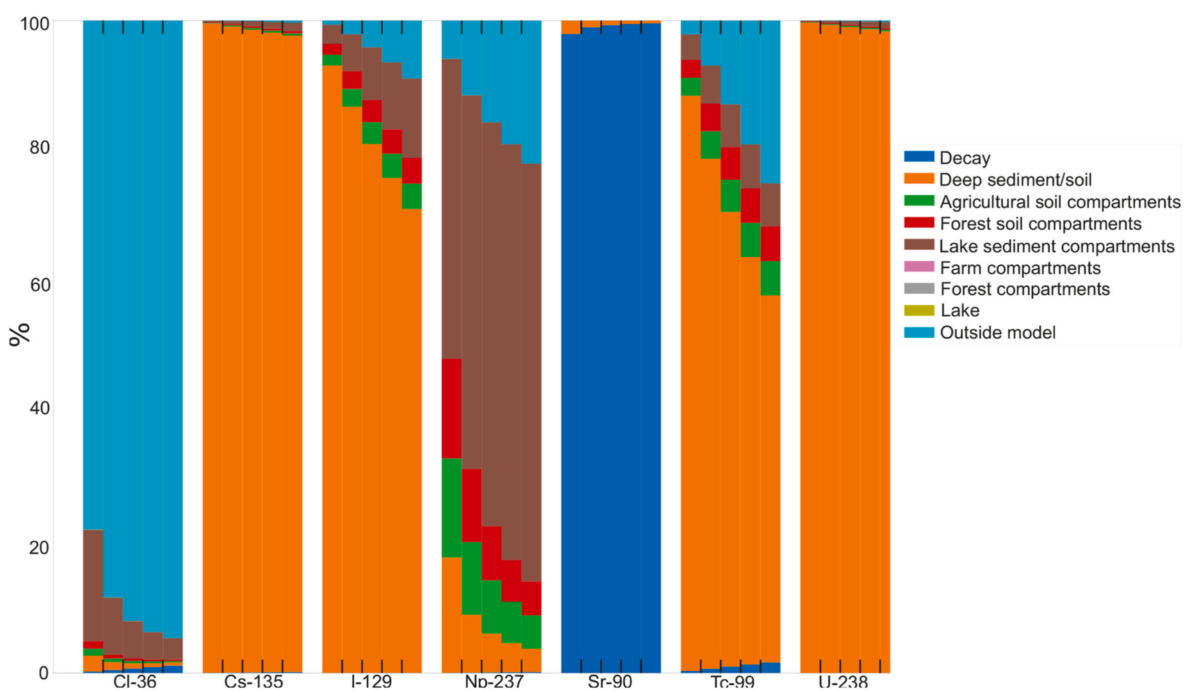

Fig. 2. Inventory showing how the radionuclide release accumulates in different compartments in the next 2 000 − 10 000 years (interval is in 2 000 years' steps).

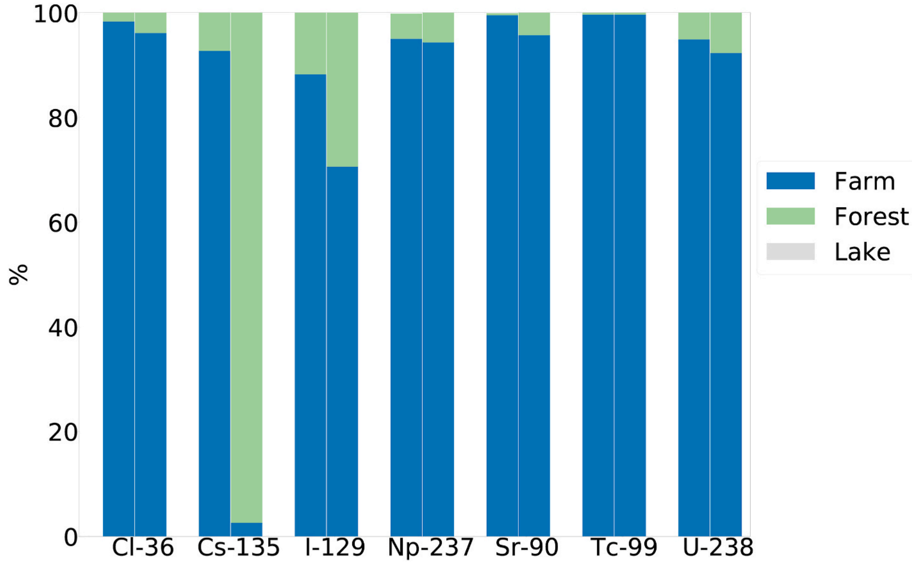

Fig. 3. Relative contribution to the dose conversion factor to humans of the pathways through the farm, forest and lake ecosystems in two cases: 1) irrigation using lake water and applied to farm crops (left-hand column), 2) no irrigation (right-hand column).

5.4. Sensitivity analysis

The sensitivity analysis was implemented using the Sensitivity Analysis Toolbox in the Ecolego software suite (Ekström, 2005; Ekström and Broed, 2006). The sensitivity analysis was done in two phases using three sensitivity analysis methods: the Morris method and the variance-based EFAST and Sobol' methods. In order to find out which parameters had the greatest influence on the dose conversion factor, the method presented by Morris (1991) was used. The Morris method is described as a screening method that can be used to isolate the set of the most important parameters. The Morris sensitivity analysis was performed in Ecolego using 100 realizations of the parameters described by a probability distribution (see Table 1 and Supplement 3). Based on the Morris sensitivity analysis, a set of 20 parameters having the highest influence was selected for the second phase of the sensitivity analysis for each radionuclide in question. Using the results from the first phase of the sensitivity analysis, the second phase was conducted using the two variance-based methods mentioned above. The sample size was set to 100 for both methods.

6. Results

Firstly, the results of the deterministic calculations are presented; one case with and one case without the irrigation of crops with lake water. In this way the effect of irrigation on the contribution of the farm/forest ecosystems to the dose conversion factor can be assessed. After that, the sensitivity analysis results are presented for the case where irrigation occurs. Sensitivity analysis was performed only for the case involving irrigation as we consider this more realistic and because the sensitivity analysis is of high computational complexity.

6.1. Deterministic simulations

In Fig. 2, an inventory of the total radionuclide activity for ^{36}Cl,^{135}Cs, ^{129}I, ^{237}Np, ^{99}Tc,^{90}Sr and ^{238}U is presented. Radionuclides accumulated in the various compartments in 2 000 – 10 000 years from the present day, taking into account radioactive decay, are presented in the columns of the figure (one column represents 2000 years for each radionuclide). Because of its relatively short half-life, the majority of the released ^{90}Sr will have decayed. For ^{36}Cl, the majority of the release ends up outside the model because of its high mobility and the discharge from the lake. The majority of the concentrations of (^{135}Cs, ^{129}I,^{99}Tc and ^{238}U) are held in the 'deep sediment' or 'soil' compartments. ^{237}Np will accumulate in the 'lake sediment' compartment.

Table 2
Dose conversion factor (DCF) at 10 000 AP and proportions (%) of the various contributing pathways for the case with irrigation. The contribution of pathways related to forest having the highest impact on the dose conversion factor among the considered radionuclides are in bold.

DCF (Sv/year)/(Bq/year)	^{36}Cl	^{135}Cs	^{129}I	^{237}Np	^{90}Sr	^{99}Tc	^{238}U
	1.2×10^{-6}	8.7×10^{-12}	5.0×10^{-11}	1.1×10^{-9}	8.5×10^{-12}	1.7×10^{-7}	5.8×10^{-12}
Beef	0.7	6.1	1.1	0.0	1.0	0.0	0.0
Berries (garden)	0.7	0.0	4.6	1.7	0.1	0.2	1.8
Berries (forest)	1.7	0.1	**11.8**	4.3	0.4	0.4	4.5
Cereal	36.5	10.3	3.3	4.6	8.1	0.8	15.5
Drinking water	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eggs	1.1	0.1	0.1	0.0	0.1	0.0	0.3
External	0.0	0.0	3.0	5.4	0.0	0.0	0.0
External_from water	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fish	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Inhalation	0.0	0.1	0.0	31.9	0.0	0.0	19.6
Game	0.0	**0.4**	0.0	0.0	0.0	0.0	0.0
Leguminous vegetables	1.2	1.5	4.7	2.8	10.9	0.3	0.6
Milk	24.0	37.9	10.3	0.0	14.4	0.0	2.5
Mushrooms	0.0	**6.8**	0.0	0.0	0.0	0.0	0.0
Mutton	0.1	1.2	0.0	0.0	0.0	0.0	0.0
Other vegetables	22.1	17.9	28.5	36.1	47.1	92.5	42.1
Pork	0.4	2.0	0.5	0.0	0.0	0.0	0.4
Potato	7.2	11.8	23.9	5.4	7.0	0.1	7.5
Poultry	1.8	0.8	0.0	0.0	0.0	0.0	0.3
Root vegetables	2.5	3.0	8.2	7.1	10.8	5.7	4.3
Wood burning	0.0	0.1	0.0	0.5	0.0	0.0	**0.6**

In Fig. 3, the contribution to the dose conversion factor of three main subsystems – farm, forest and lake – is presented for the case where crops are irrigated using the lake water contrasted with the case with no irrigation. It can be seen that, when irrigation is involved, the forest ecosystem contributes only slightly to the dose conversion factor; however, when the contaminated lake water is not used for irrigation and thus the dose conversion factor obtained from the farm system is

Table 3
Dose conversion factor (DCF) at 10 000 AP and the proportions (%) of the various contributing pathways for the case without irrigation. The contribution of pathways related to forest having the highest impact on the dose conversion factor among the considered radionuclides are in bold.

	^{36}Cl	^{135}Cs	^{129}I	^{237}Np	^{90}Sr	^{99}Tc	^{238}U
DCF (Sv/year)/(Bq/year)	1.2×10^{-6}	2.4×10^{-13}	4.0×10^{-11}	1.1×10^{-9}	8.1×10^{-12}	1.7×10^{-7}	5.7×10^{-12}
Beef	0.7	0.2	0.8	0.0	1.0	0.0	0.0
Berries (garden)	0.6	0.0	3.7	1.7	0.1	0.2	1.7
Berries (forest)	1.6	0.0	**9.5**	4.3	0.3	0.4	4.4
Cereal	35.6	0.3	2.6	4.6	7.8	0.8	15.1
Drinking water	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Eggs	1.1	0.0	0.1	0.0	0.1	0.0	0.3
External	0.0	0.0	2.4	5.4	0.0	0.0	0.0
External from water	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Fish	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Inhalation	0.0	0.0	0.0	31.6	0.0	0.0	19.1
Game	0.0	**0.3**	0.0	0.0	0.0	0.0	0.0
Leguminous vegetables	1.1	0.0	3.7	2.8	10.4	0.3	0.6
Milk	23.4	1.1	8.3	0.0	13.8	0.0	2.4
Mushrooms	2.2	**97.1**	20.0	0.9	3.9	0.0	2.7
Mutton	0.1	0.0	0.0	0.0	0.0	0.0	0.0
Other vegetables	21.7	0.5	22.8	35.8	45.3	92.5	41.0
Pork	0.4	0.1	0.4	0.0	0.0	0.0	0.4
Potato	7.1	0.3	19.1	5.3	6.7	0.1	7.3
Poultry	1.8	0.0	0.0	0.0	0.0	0.0	0.3
Root vegetables	2.4	0.1	6.5	7.1	10.4	5.7	4.2
Wood burning	0.0	0.0	0.0	0.5	0.0	0.0	**0.6**

decreased, the forest subsystem becomes the dominant pathway for ^{135}Cs and a major contributor for ^{129}I. The dose conversion factors can be compared with those for the lake–farm system alone (detailed results presented in Pohjola et al. (2016; 2019)).

Detailed results for the deterministic calculations, indicating the contribution of the 20 most significant parameters, are presented in Table 2 (involving irrigation from the contaminated lake) and 3 (no irrigation). The relative contribution of forest-related parameters increases if irrigation is not involved, especially for ^{135}Cs and ^{129}I. Mushrooms are the most significant pathway for both of these two radionuclides, whilst berries play a significant role for ^{129}I (see Table 3).

6.2. Sensitivity analysis

Sensitivity analysis was performed for the case involving irrigation from the contaminated lake using the Morris method to screen the most significant parameters and then comparing the results obtained with the Sobol' and EFAST methods. The results for the 10, 000 year time scope are presented in Fig. 4. The parameters are grouped into main categories for clarity of visualization. The columns in this figure indicate to what extent the variability in the dose conversion factor is explained by the variability of a certain parameter group. It appears that uncertainty in the agricultural soil properties, deep soil or sediment properties and soil to plant concentration ratios contribute most to the uncertainty of the dose conversion factor. While the Sobol' and EFAST methods mostly rank the parameter groups in a similar manner in terms of sensitivity (with some exceptions), the proportion of the parameter groups can be quite different.

More detailed results of the sensitivity analysis of the dose conversion factor to the variability of individual model parameters is presented in Tables 4 and 5. It can be seen that the variability of the solid-liquid distribution coefficient of agricultural top soil contributes significantly to the variability of the dose conversion factor with both methods, Sobol' and EFAST, for most of the radionuclides. Also, the sensitivity to deep soil properties is common for all radionuclides and both analysis methods. However, there are differences between the Sobol' and EFAST methods. The Sobol' method emphasizes sensitivity to the most significant parameters more (cf. for example ^{135}Cs in Table 5 to ^{135}Cs in Table 4). Forest-based products (game meat, berries, mushrooms and wood) cannot be seen in either of the tables. This is because of the minor amount of game meat, mushrooms and forest berries in the average human diet. When the average consumption of forest-based food is small, the standard deviation is also small and will be dropped to the least significant parameter in the sensitivity analysis.

7. Discussion and conclusions

This study is a continuation of Pohjola et al. (2016, 2019), with the purpose of examining the contribution of food originating from the forest and burning wood to the dose conversion factor for humans. Although the radioactive concentration in top forest soil is twice as big as that in agricultural top soil for all radionuclides, the amount of game meat, mushrooms and forest berries in the human diet is so small that it compensates the higher accumulation in the top forest soil. For

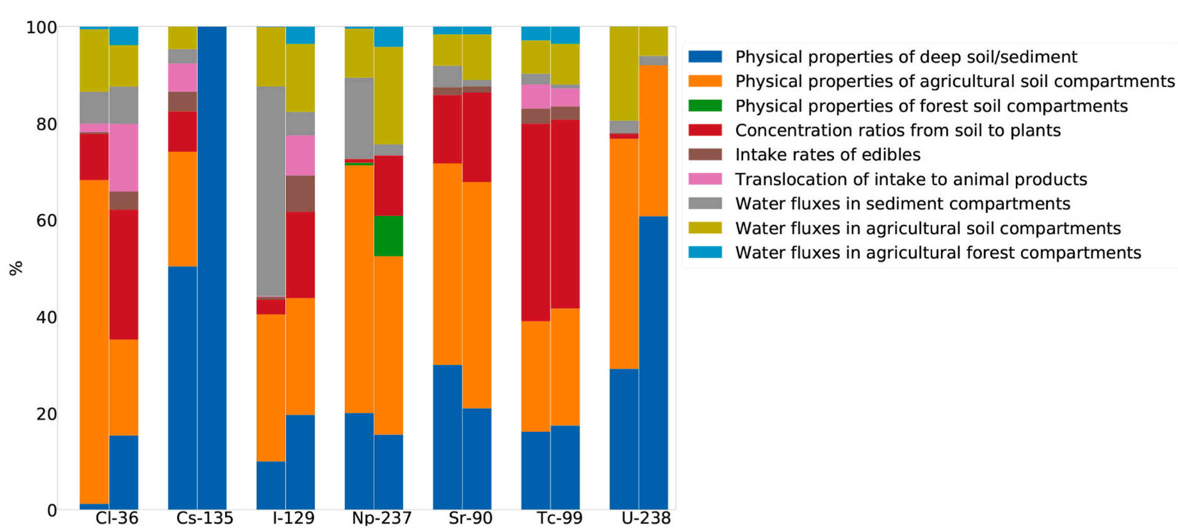

Fig. 4. Sensitivity analysis results for studied radionuclides. The left-hand columns indicate the results from EFAST and the right-hand columns the results for Sobol'.

Table 4
EFAST sensitivity analysis results (%). The five most contributing parameters for each radionuclide are in bold.

Parameter	^{36}Cl	^{135}Cs	^{129}I	^{237}Np	^{90}Sr	^{99}Tc	^{238}U
Bulk density of deep sediment/soil	0.3	**12.1**	0.6	0.8	**10.4**	3.1	1.0
Bulk density of intermediate agricultural soil		0.4		0.3	3.7		0.3
Bulk density of top agricultural soil						2.6	
Bulk density of top forest soil				0.3			
Concentration ratio from soil to cereal	**4.5**	0.6		0.2	1.5		0.2
Concentration ratio from soil to leguminous vegetables			0.1		0.8	**5.3**	0.2
Concentration ratio from soil to other vegetables	0.7	**3.6**	0.3	0.1	**9.1**	**28.4**	0.3
Concentration ratio from soil to pasture	**3.2**	3.6	0.0			3.0	
Concentration ratio from soil to potato	0.9	0.6	0.5	0.5	0.5		0.3
Concentration ratio from soil to root vegetables	0.3		2.2		2.2	4.3	0.0
Intake rate of leguminous vegetables			0.4				
Intake rate of other vegetables	0.4	2.6	0.2		1.6	3.1	0.2
Intake rate of pork		1.4					
Porosity of deep soil/ sediment	0.3		0.1	0.2	3.6	4.2	0.1
Porosity of intermediate agricultural soil						2.9	
Solid-liquid distribution coefficient of agricultural intermediate soil		**6.5**	0.1	0.2	7.5	4.0	0.2
Solid-liquid distribution coefficient of agricultural top soil	**67.0**	**13.8**	**29.6**	**49.8**	**15.0**	2.7	**46.4**
Solid-liquid distribution coefficient of deep soil/sediment	0.2	**37.0**	**9.2**	**13.4**	**9.3**	**5.2**	**22.1**
Thickness of deep soil/sediment	0.4	1.2	0.1	**5.5**	6.7	3.6	**6.0**
Thickness of forest soil rooting layer				0.2			
Thickness of intermediate agricultural soil	0.0	3.0	0.7	0.3	**15.0**	**5.4**	0.4
Thickness of top agricultural soil				0.7	0.4	**5.3**	0.4
Translocation factor to eggs	0.1						0.0
Translocation factor to milk	0.1	2.7	0.1			5.0	
Translocation factor to pork	0.2	0.6	0.0				
Translocation factor to poultry	1.3	1.2					
Translocation factor to sheep		1.3					
Water flux from agricultural soil to lake water	1.7	3.4	**4.6**	0.2	0.5	3.2	**4.6**
Water flux from deep sediment to intermediate sediment	**6.6**	2.9	**43.4**	**16.8**	4.5	2.2	2.6
Water flux from deep soil to intermediate agricultural soil	**11.3**	1.3	**7.7**	**9.4**	6.0	3.7	**14.8**
Water flux from deep soil to intermediate forest soil	0.5		0.1	0.4	1.6	2.8	0.1
Water flux from intermediate agricultural soil to top agricultural soil				0.3			0.1
Water flux from top agricultural soil to intermediate agricultural soil				0.2			

comparison, milk alone will contribute 38% to the dose conversion factor of ^{135}Cs. Total food consumption and its standard deviation have been taken from Helldán et al. (2013) and Valsta et al. (2018), but the dose conversion factor of game meat, for example, renders the significance of this parameter so small that it has been omitted from the sensitivity analysis. Also, the inhalation of wood burning emissions is so small that its influence on the dose conversion factor is insignificant compared to other pathways.

The prominent role of ^{135}Cs and ^{129}I in the contribution of forest products may arise from the conservative concentration ratio data used, or alternatively from these data being less conservative respective to the data for the other nuclides. Data on concentration ratios for the understorey (used as a proxy for wild berries) have been taken from the literature compilation of Avila (2006), where these data have been assessed as of good quality for chlorine, caesium and strontium, medium quality for neptunium and poor for iodine. For mushrooms, the concentration data come from the literature compilation of Avila (2006) as well, ranked as good only for chlorine and medium for strontium, but poor for all other elements.

For crop plants, the data have been taken from the global compilation of IAEA (2009) with similarly varying quality, albeit not necessarily in the same order for the different elements and generally based on a wider data basis than those used here for forests. Concentration ratios are generally around the same orders of magnitude for wild berries and mushrooms as for crop plants, but much lower for wild berries and much higher for mushrooms than for crops in the case of ^{135}Cs. For ^{129}I, the concentration ratios are much higher for mushrooms but only slightly higher for wild berries. The concentration ratios are clearly higher for mushrooms also in the case of ^{238}U and somewhat higher for mushrooms with ^{237}Np. For ^{99}Tc, there is a high spread in the concentration ratios in general across the foodstuff categories. For ^{36}Cl, the concentration ratios are rather similar to one another across the foodstuff categories. Combined with the differences in the consumption rates (see Supplement 3, Table S3.11), the above explains the considerably growing role of the forest pathways, particularly for ^{135}Cs and ^{129}I, and the lack of difference for ^{99}Tc, in the case of no crop irrigation.

Based on the comparison of results for the two cases, with and without irrigation of crops with lake water, and the sensitivity analysis results, it can be observed that if the sensitivity analysis had been used alone on the default model without modulating the irrigation on and off, the dependence of the importance of the forest pathways on the scenario context may have not been identified. Indeed, it should be borne in mind that the results of a numerical sensitivity analysis are always dependent on the mathematical model and the input parameter distributions used. There is also a degree of imbalance in the overall model regarding the sensitivity analysis; there are several dietary items from the farm, each associated with an independent consumption rate, whereas the forest foods are represented by relatively few items and consumption rates. In

Table 5
Sobol' sensitivity analysis results (%). The five most contributing parameters for each radionuclide are in bold.

Parameter	^{36}Cl	^{135}Cs	^{129}I	^{237}Np	^{90}Sr	^{99}Tc	^{238}U
Bulk density of deep sediment/soil	3.9		2.9	3.7	**7.8**	**3.9**	
Bulk density of intermediate agricultural soil				3.9	1.7		
Bulk density of top agricultural soil						3.7	
Bulk density of top forest soil				4.2			
Concentration ratio from soil to cereal	**10.3**			4.2	1.7		
Concentration ratio from soil to leguminous vegetables			3.1		1.6	3.7	
Concentration ratio from soil to other vegetables	**5.8**		**4.6**	4.2	**13.4**	**28.1**	
Concentration ratio from soil to pasture	3.3		3.0			3.7	
Concentration ratio from soil to potato	3.7		3.5	4.2	1.6		
Concentration ratio from soil to root vegetables	3.8		3.7		0.4	3.7	
Intake rate of leguminous vegetables			3.7				
Intake rate of other vegetables	3.7		3.8		1.3	2.7	
Intake rate of pork							
Porosity of deep soil/sediment	3.8		3.7	4.2	1.4	2.7	
Porosity of intermediate agricultural soil						3.7	
Solid-liquid distribution coefficient of agricultural intermediate soil			3.6	4.0	1.3	3.7	
Solid-liquid distribution coefficient of agricultural top soil	**16.0**		**16.8**	**20.8**	**32.1**	**5.8**	**31.3**
Solid-liquid distribution coefficient of deep soil/sediment	3.8	**100.0**	**9.0**	**5.7**	4.4	3.7	**60.7**
Thickness of deep soil/sediment	3.9		4.1	2.0	**7.4**	**7.2**	
Thickness of forest soil rooting layer				4.2			
Thickness of intermediate agricultural soil	3.8		3.7	4.1	**10.2**	3.7	
Thickness of top agricultural soil				4.2	1.6	3.7	
Translocation factor to eggs	3.5					0.0	
Translocation factor to milk	3.7		4.6			3.7	
Translocation factor to pork	3.7		3.7				
Translocation factor to poultry	3.0						
Translocation factor to sheep							
Water flux from agricultural soil to lake water	4.2		3.3	**4.7**	3.1	**4.7**	
Water flux from deep sediment to intermediate sediment	**7.7**		**4.8**	2.3	1.3	0.8	**2.0**
Water flux from deep soil to intermediate agricultural soil	**4.4**		**10.8**	**7.1**	6.3	3.7	**6.1**
Water flux from deep soil to intermediate forest soil	3.8		3.6	**4.2**	1.6	3.6	
Water flux from intermediate agricultural soil to top agricultural soil				4.2			
Water flux from top agricultural soil to intermediate agricultural soil				4.2			

addition to affecting the number of parameters under analysis, these consumption rates are heavily intertwined and should be represented by an enhanced aggregation of food items and correlation between the consumption rates of the food groups resulting from the aggregation. However, no self-consistent overall dietary statistics for subgroups representing various dietary regimes have been identified that would provide consumption rates of all the food items in the model, e.g., for those consuming a lot of wild berries, mushrooms and game. Nevertheless, such aggregation would help in constructing the parametrized replacement of some consumption rates or using a priori correlations between the consumption rates based, for example, on a reasonable total calorific consumption constraint. However, this would require a significant number of further computational trials outside the current project framework.

Overall, the study demonstrates that, at least under some assumptions, the exposure pathways related to forest products may considerably contribute to the dose in a lake–farm–forest system, but this does not necessarily become apparent by using a straightforward sensitivity analysis alone. As readily exemplified in a few other studies (e.g., Reid and Corbett, 1993; Pröhl and Müller, 1996; Ekström and Broed, 2006; Broed, 2007; Olyslaegers et al., 2011; Avila et al., 2013; Kupiainen, 2014; Kupiainen and Nummi, 2017; Broed et al., 2021; Nicoulaud-Gouin et al., 2022), it was also confirmed that an advanced sensitivity analysis for a radionuclide transport and dose assessment model in such a landscape scale is feasible even with moderate computational efforts, as it took about one month to run the simulations with a standard office computer simultaneously used also for other tasks.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

The work presented in this paper has been supported by the EURAD DONUT project, the Robocoast R&D Center - Robotics Living Lab for Companies project and the Finnish national research program for nuclear waste management (KYT2018).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvrad.2022.107019.

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