Probabilistic assessment of the impact of bottom sediment on doses to humans from a groundwater-mediated radionuclide release in a farm-lake scenario

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Abstract. Radionuclide transport with groundwater flow and subsequent doses to people are an aspect to be studied when assessing the long-term safety of geological nuclear waste repositories. A scenario where the radionuclide release migrates through a three-layer sediment structure of a lake in a farming environment is presented in this paper. The sediment column consists of deep (till), intermediate (glacio-aquatic sediment) and top layers (clay). The radionuclide release is assumed to enter the deep sediment layer from a bedrock fracture system at a rate of 1 Bq/year. The main objectives of the paper are to investigate the most contributing parameters, especially linked to the sediment layers, to the overall dose estimates for humans. The sensitivity analysis was conducted in two phases where the Morris method was used for screening and the EFAST and Sobol's methods were used for estimating total-order indices. The studied radionuclides, ^{36}Cl, ^{135}Cs, ^{129}I, ^{94}Nb, ^{237}Np, ^{90}Sr, ^{99}Tc and ^{238}U, exhibit differences in how the sediment layers affect the concentration in the lake water used for drinking, irrigation and watering cattle and subsequently the dose conversion factors for humans through ingestion, inhalation and external radiation.

Keywords: Biosphere analysis, Dose conversion factor, Farm, Lake, Radionuclide, Sediment, Sensitivity analysis

1. Introduction

Groundwater is the physical medium carrying radionuclides from geological nuclear waste repositories, such as those in operation and planned at the Olkiluoto nuclear site in Finland, to surface waters in the case of releases from the waste packages. Groundwater also supplies water to plants, animals and humans (Codell & Duguid 2008). The Finnish Radiation and Nuclear Safety Authority (STUK) regulates the use of nuclear energy and management of nuclear waste in Finland. In the case of release from a repository, postulated where not expected, STUK prescribes in the Guide for Disposal of Nuclear Waste (STUK 2018) that for safety assessments assessing compliance with the dose constraints, the groundwater shall carry the radionuclides to the surface environment, in which there is also a lake, so that this release results in radiation exposure of people and biota.

Research concerning radionuclide transport through sediments has a relatively long history. For example, in (Sayre, Guy & Chamberlain 1963) the modelling of the transport of short-lived radioactive waste that is disposed in a river, was discussed. Their main concern was the pollution of drinking water and biota, and how the individual major modeling parameters and/or actual behavior of the waste disposal process (i.e., characteristics of the waste, stream flow, dilution, sediments, sorption etc.) of the water pathway will affect the doses. (Booth 1975) presented a four-compartment differential equilibrium model for surface sediment contamination and interaction with water bodies. In this model the contamination came via surface waters (i.e., drainage) and during the interaction, the sediment layer acted as an inhibitor of radionuclide flow. The time period in (Booth 1975) is not defined, but an example of a water-sediment radio-ion exchange equilibrium occurring within 40 days is mentioned. (Onishi & Trent 1982) presented the FLESCOT finite-element model for estimating radionuclide distribution in estuaries, considering the following factors: time variance, 3D-flow, temperature, salinity and sediments. Because radionuclide adsorption/desorption was strongly dependent on sediment characteristics, a total of three sediment types were included (silt, clay and sand). In this model the contamination was also added to water upstream and the timescale was in the order of days or months.

Radionuclide transport between lake water and sediments was modelled and evaluated in (Snodgrass 1984). In this steady-state model the sediment layer consisted basically of one layer and the radionuclides came from surface water and eventually sank and interacted with sediments. Adsorption, diffusion, decay and limnological consumption effects, such as epilimnetic biofilms and limnocorals, were also considered and reviewed in the analysis.

(Thiessen, Thorne, Maul, Pröhl & Wheater 1999) reviewed biosphere modelling practices and discussed extensively different modeling scenarios, influence of parameters and complex interactions between biota and water environments including sediments. Sediments were mainly radionuclide sinks in their context. In (Thiessen et al. 1999) the timescale in the radionuclide transportation simulation was 200 years. (Erichsen, Konovalenko, Møhlenberg, Closter, Bradshaw, Aquilonius & Kautsky 2013) compared two biosphere compartment modeling scenarios where radionuclides were point-released at the bottom of a sea bay. The models included sediment interactions and complex biota interactions. They discussed the benefits of fine and coarse resolution models and concluded that both resolutions were meaningful in different contexts. The timescales in the comparisons were up to 100 years.

All of the research mentioned previously assume that the contaminant is released into surface waters instead of groundwater. Also, the timescales in those studies are well below 1000 years. However, radionuclide releases in the context of lake and river sediments have been considered in (Sundblad, Puigdomenech & Mathiasson 1990) and (Klos, Müller-Lemans, van Dorp & Gribi 1996) using much longer timescales. In (BIOMOVS 1989) a lake-water biosphere simulation is presented using two radionuclides ^{226}Ra and ^{230}Th. There, the radionuclide release was assumed to enter the model via river water.

Radionuclide flow from bedrock to the biosphere was considered in (Kupiainen 2014) and (Kupiainen & Nummi 2016). They presented a simplified solute transport model, where biota, household water and radionuclide flow through sediments were considered. The radionuclide contamination in lake water was examined in terms of a two-layer sediment model. The parameters were varied using Monte Carlo simulation to obtain probabilistic results. The modelled water body in this study is a small eastern bay of the same lake modelled to appear in the future at the Olkiluoto repository site as the one used in our work. However, this study focused on the whole-system behaviour and results are not readily available on the specific role of the sediment layers to allow comparison with our results.

In the model presented in this paper, our earlier lake scenario (Pohjola, Turunen, Lipping & Ikonen 2016) is altered so that that the radionuclides are transported from groundwater into the lake through the sediments instead of being directly released into the lake water. The agricultural and exposure parts of the model remain the same. Also, the effects of radionuclide contaminated dust and external irradiation are examined as a complementary view to the earlier work. Sensitivity analysis is improved by adding more simulations to get more reliable estimations of distributions than in (Pohjola et al. 2016) in which the modelling was based on the abovementioned scenario as well. Two variance-based methods are compared with each other in order to find differences in results and to discuss possible reasons for them. The presentation in this paper follows the BIOMASS-6 ((IAEA 2003)) assessment methodology, taking into account the recent interim-version upgrades (Lindborg 2018).

2. Methods

2.1. Assessment context

The current work has been made in research purposes to study the sensitivities of assumptions in biosphere assessments for a generic nuclear waste repository deep in the bedrock of the Olkiluoto site in western Finland. At that site there is a low- and intermediate-level nuclear waste ('VLJ') repository in operation and a spent nuclear fuel repository under construction (cf. e.g., (Posiva 2013a)). However, this piece of research has been made independently of the actual licensing assessments of those repositories, although assuming the general context for illustrative purposes in order to avoid arbitrary or artificial assumptions about the biosphere. Also, the respective regulatory regime summarised in (STUK 2018) is assumed for this methodological study. The releases from the repository are assumed to migrate to a future lake (Pohjola, Turunen, Lipping & Ikonen 2014, Pohjola et al. 2016) formed as a consequence of the post-glacial shoreline displacement prevailing in the region. Radionuclides released from the geosphere are assumed to be transported through the bottom sediments of the lake into the lake water with a release rate of 1 Bq/year of each nuclide (^{36}Cl, ^{135}Cs, ^{129}I, ^{94}Nb, ^{237}Np, ^{90}Sr, ^{99}Tc and ^{238}U) from the bedrock to the lowest sediment layer. This selection of radionuclides covers a range of bio-geochemical behaviors and includes nuclides considered in earlier assessments for the site in question (see, e.g., (Vieno & Suolanen 1991, Posiva 2013b)) as well as those of general interest (see, e.g., (Chen, Kowe, Mobbs, Pröhl, Olyslaegers, Zeevaert, Kanyar, Pinedo, Simón, Bergström, Hallberg, Jones, Oatway & Watson 2006)).

The lake, receiving the postulated releases, has been projected to form approximately 3 000 years after present (AP) as indicated, for example, by a probabilistic model of the changes in the geomorphic landscape in the Olkiluoto area (Pohjola 2014). For recognizability, it has been named Lake Liponjärvi in earlier assessments specific to the site (e.g., (Posiva 2013b)), and we have decided to continue the tradition. In Figure 1, the location of this lake in relation to the present Olkiluoto Island is illustrated alongside the topography modelled for 10 000 AP. The extent of shoreline displacement in 10 000 years (a time span characteristic of releases from deep geological repositories) is approximately 30 meters in the Olkiluoto area (Pohjola 2014). This uplift means that the coastline will move westwards about 10-15 kilometers and several new lakes will be formed. In this paper we consider one of these future lakes, Lake Liponjärvi, in our model because it is located near the nuclear waste repositories at Olkiluoto and because the volume and expected productivity of the lake are large enough to satisfy the needs of a small hypothetical farming 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 et al. 2014). Realistically oriented data distributions have been chosen in addition to deriving the lake properties from probabilistic geomorphological modelling aiming at best-estimate representation of a plausible future setting; for the radionuclide transport and dose calculations, cautious assumptions and data distributions have been employed.

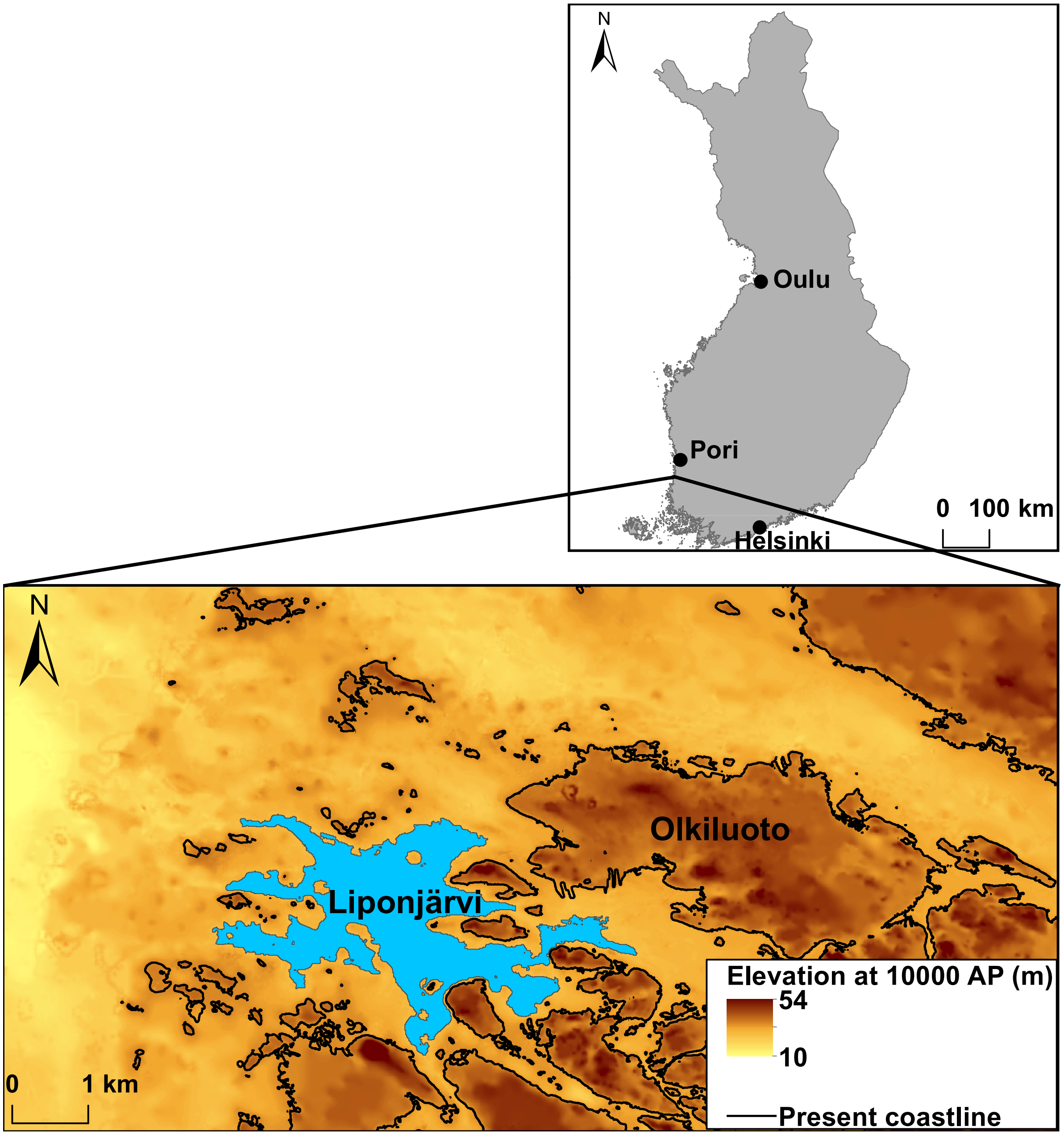

Figure 1. The projected future general topography and the location of the potential future lake, named Lake Liponjärvi, postulated for the assessment to form in the vicinity of the current Olkiluoto Island in Finland.

The purpose is to study the variation of different parameter values that will affect the doses to humans. Following from the regulatory guide, the time horizon to compute the doses is set, for simplicity, to a round 10 000 years (similarly to, e.g., (Posiva 2013b)) to cover *"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"* (STUK 2018) and to allow for sufficient time for the radionuclides to migrate and accumulate in the modelled system. For simplicity and comparability with the present-day context, we follow the regulatory guidance (STUK 2018) also in respect of assuming the climate, ecosystems, human habits, diet and metabolism similar to those at present; our focus is, after all, to study the impact of the sediment layers in the lake-farm system.

2.2. Biosphere system identification and justification

When considering the Finnish lakes formed due to the shoreline displacement and isolation from the Baltic Sea, it is found that they usually have a moraine layer above the bedrock overlaid by glacial, transition and post-glacial clays and recent mud (Winterhalter 1992). Groundwater is supposed to find the easiest way (as determined by the hydraulic properties of the sediment layers) through the sediments into the lake. The Geological Survey of Finland has conducted acoustic-seismic studies in the vicinity of Olkiluoto Island (Rantataro 2001). One of the purposes of these studies has been to identify the different sediment layers and their thicknesses as well as the elevation of the bedrock at the bottom of the sea. Based on these studies, deposits of till, glacioaquatic mixed sediment, post-glacial clays and recent clay are modeled as sediment compartments in this work.

The agricultural scenario presented in this study is based on a typical farming scenario in Finland, similar to the one presented in (Hjerpe & Broed 2010). In addition to the cultivation of field crops and garden products, livestock are kept for milk, meat and eggs and fish are caught from the lake. The differences compared with the model presented in (Hjerpe & Broed 2010) are that the household, drinking and irrigation water are taken directly from the lake instead of a well. The diet of humans is based on the nutrition survey conducted by (Helldán, Raulio, Kosola, Tapanainen, Ovaskainen & Virtanen 2013). Fish, included in the diet, is assumed to be taken from the lake, and the vegetables are irrigated with the contaminated water. Also, the livestock consumes the contaminated water for drinking. The radionuclide migration paths and human exposure pathways in the modelled biosphere system are illustrated in Figure 2.

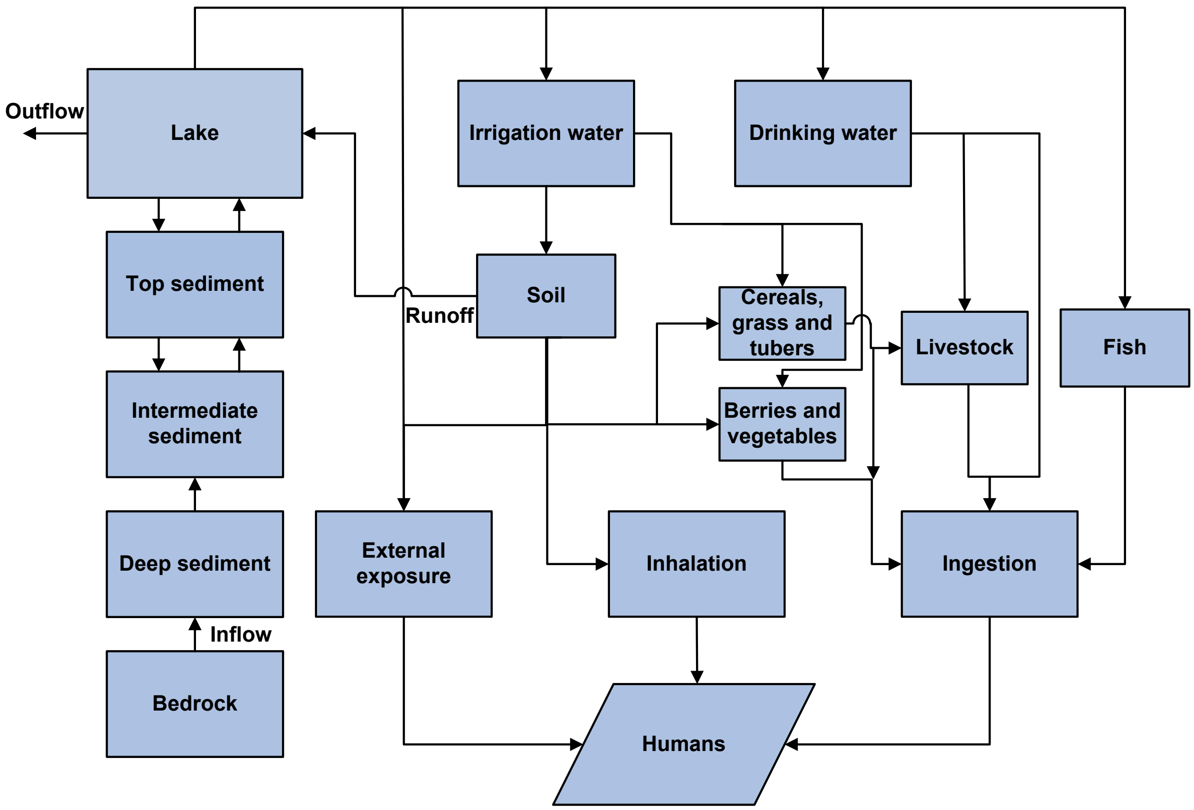

Figure 2. Flowchart of the transport paths of radionuclides in the model.

The consideration of features, events and processes (FEPs) is based on the characteristics of the site chosen for the study and an assumption that the climate and environment will remain approximately the same during the next 10 000 years. A screening list, included in the BIOMASS-6 assessment methodology report (IAEA 2003) is used in this study to ensure that all generally important FEPs have been duly considered (Supplement 1).

2.3. Potentially exposed group definition

The deep geological disposal facilities are assumed to release radionuclides in the distant future in this type of scenario. These releases result in migration to the geosphere and finally into the biosphere. Far future release scenarios contain several sources of uncertainties when predicting doses or risks to humans. For reducing the computation, the ICRP (International Commission on Radiological Protection) suggests that it would be adequate to estimate the lifetime dose or risk to an adult person and this estimation can be considered to represent the whole population (ICRP 2013). Therefore, only adult humans are considered in our model and the male dietary profile is selected when exploring the potential sensitivity of the dose to dietary assumptions.

A dose constraint of 0.1 mSv per year has been set for the most exposed individuals in the surroundings of nuclear waste disposal facilities in Finland (STUK 2018). This can be simplified into average individual dose in a small community of self-sustaining families near a nuclear waste disposal site. The representative person of the potentially exposed group is defined in this study as an individual who satisfies all his needs of food and drinking water solely from the contaminated sources. Such a situation can occur in self-sustaining communities, where the production and consumption of contaminated food is equal, and the community life habits are relatively homogenous.

2.4. Biosphere model development and calculation

2.4.1. Model formulation The biosphere model consists of several compartments as presented in Figure 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 different clay layers were handled as a single layer for simplicity and also due to the lack of data available in the literature for the different clay types. The thicknesses of the sediment layers in the broader area of interest were interpolated from the point-wise acoustic-seismic data (Rantataro 2001) using the thin plate spline method that has earlier been shown appropriate for the topography at the Olkiluoto site (Pohjola 2014). From these data, the average value and range of the thickness of each sediment layer within the projected Lake Liponjärvi were extracted for the calculations; average values were used for the nominal dose conversion factors and the ranges in the sensitivity analysis.

Lake volume, outflow and depth histograms are obtained from the UNTAMO simulation tool described in detail in (Pohjola 2014). These histograms will simulate the annual variations, water depth and volume of Lake Liponjärvi. Sedimentation from catchment area is reducing the size of the lake according to the method described in (BIOMOVS 1989). For simplicity reasons the method assumes that the depth profile of the lake is triangular.

Water fluxes between the sediment compartments, from the bedrock to the deep sediment and from the field soil to lake water are adopted from (Posiva 2014), as detailed in Table 20 in Supplement 3. These fluxes have been derived from a surface hydrological and near-surface geohydrological model (Posiva 2012) that has used the same sediment stratigraphic description of (Rantataro 2001) as employed in this model. The lake water volume is represented by a single compartment so that the radionuclide release will mix instantly and homogeneously throughout the whole lake that has a water turnover time of 0.8 years. The postulated constant release rate of 1 Bq/year of each radionuclide studied is assumed to enter the deep sediment compartment (till) from the bedrock. Annual precipitation, evaporation and catchment area water fluxes have been taken into account when estimating the outflow flux (cf. Figure 2). Runoff and interflow from the surrounding terrestrial catchment into the sediment layers are assumed to be zero.

Interactions between the sediment layers (advection, bioturbation and diffusion), sedimentation, resuspension from the top sediment, and the outflow from the lake affect the lake water radionuclide concentrations as presented in Figure 2. The transfer and interaction functions describing these geosphere-biosphere processes are taken from

(Posiva 2014), (Karlsson & Bergström 2000) and (Vieno & Suolanen 1991). The transfer of radionuclides to the crops caused by root uptake and retention of irrigation water, the transfer to the livestock from contaminated fodder and drinking water and the transfer of radionuclides due to intakes of drinking water by humans are modelled using transfer functions described in (Hjerpe & Broed 2010). The equation for bioaccumulation from water to fish is taken from (Vieno & Suolanen 1991). With the exception of the sediment compartments, the mathematical model is the same as in our earlier paper (Pohjola et al. 2016).

2.4.2. Data selection The data sources used in the calculations are presented in Table 1. The parameter values are presented in more detail in Electronic supplement 3.

Table 1. The references for the parameter values in dose calculations.

Food intake rates	(Helldán et al. 2013)
Dose coefficients for ingestion	(ICRP 2012)
Dose coefficients for inhalation	(ICRP 2012)
Dose coefficients for external exposure	(Eckerman & Ryman 1993)
Soil to plant transfer factors	(IAEA 2009)
Water to fish bioaccumulation factor	(Karlsson & Bergström 2000)
Translocation factors (animal intake)	(Karlsson & Bergström 2000)
Translocation factors (root crops)	(Karlsson & Bergström 2000)
Solid-liquid distribution coefficients	(Karlsson & Bergström 2000)
	(Sheppard, Long, Sanipelli & Sohlenius 2009)
Water fluxes	(Posiva 2014)
Sediment layer thicknesses	(Rantataro 2001)
Sedimentation and resuspension rates	(Vieno & Suolanen 1991)
Lake properties	(Pohjola 2014)
Other parameters	(Hjerpe & Broed 2010)

The average Finnish daily food intake rates presented in (Helldán et al. 2013) were converted to annual values, and they were assumed to follow a normal distribution. For the soil to plant transfer factors, log-normal distributions were assumed, and geometric means were used together with geometric standard deviations. In the cases when these were not available, 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, a geometric standard deviation of 3.2 based on (Sheppard 2005) was assumed for the sensitivity analysis. Applicable water fluxes between the sediment compartments have been reported in figure 4.17 of (Posiva 2014) using the mean, maximum and minimum values. A triangular distribution determined by these mean, maximum and minimum values was used to specify the water fluxes in the sensitivity analysis.

2.4.3. Modelling platform The modelling platform used for the dose conversion factor calculations and the sensitivity analysis was the Ecolego software product (Facilia AB, version 6.5.23) (Avila, Broed & Pereira 2003). Ecolego can be used for creating dynamic models and running simulations deterministically or probabilistically. Radiological risk assessment was considered in the software, for example, by using a built-in database of radionuclides so their half-lives are automatically taken care of in the dose assessment calculations. The assessments in this study were computed using 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 Numerical Differentiation Formulas, as described in (Shampine & Reichelt 1997). The Mathworks Matlab (version R2017A) simulation platform was used for data processing and visualization.

2.4.4. Sensitivity analysis The sensitivity analysis was implemented using the Sensitivity analysis toolbox within Ecolego (Ekström 2005). The sensitivity analysis was done in two phases using three sensitivity analysis methods: the Morris method and the variance-based EFAST and Sobol's methods. In order to find out the parameters having the largest influence on the total dose, the method presented in (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 400 realizations to ensure the validity of the results. 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 two variance-based methods described below. The sample size was set to 1 000 for both methods in order to ensure convergence.

FAST (Fourier Amplitude Sensitivity Testing) is a method presented in (Cukier, Fortuin, Shuler, Petschek & Schaibly 1973, Schaibly & Schuler 1973). FAST uses Fourier Transforms for estimating the variance of the outputs of the model. The resulting sensitivity analysis results are in the form of first-order indices. EFAST (Extended Fourier Amplitude Sensitivity Testing), a modification of FAST, was presented in (Saltelli, Tarantola & Chan 1999). The EFAST method differs from FAST in terms of sampling and producing total-order indices as a result. First-order sensitivity indices describe the main effects of single parameters. Total order sensitivity indices take into account all the first and higher-order effects, including the interactions between the parameters, enabling the evaluation over the whole parameter space.

The Sobol method is a variance-based method quantifying the contribution of each input factor to total unconditional variance of the model output (Sobol 1990). Whereas the FAST method uses sinusoidal components of the Fourier method for pattern search, Sobol uses Monte Carlo simulation for that purpose. The Sobol algorithm is able to produce both first-order and total-order sensitivity indices.

2.4.5. Definition of calculation cases A calculation case including two variants for dose assessment was implemented and compared to the reference case of (Pohjola et al. 2016). The landscape configuration projected for 10 000AP when Lake Liponjärvi is fully formed (Pohjola et al. 2016) was used. The purpose of the two variants of the calculation case 2 is to find out if external radiation from soil and lake water and radiation from dust inhalation have a major impact on the doses to humans. In the reference case study in (Pohjola et al. 2016) external radiation and dust inhalation were not taken into consideration as it was shown that their influence was negligible on the total doses. However, in this study, as ^{94}Nb was added to the list of radionuclides to enhance the variability across bio-geochemical properties, the external radiation and radiation caused by inhalation were considered more significant contributors to the total dose. The calculation cases can be summarized as follows:

- Case 1: Scenario based on (Pohjola et al. 2016) where the radionuclide release is mixed directly with the lake water
- Case 2:
  - a: Scenario where the radionuclide release has migrated through the three-layer sediment structure into the lake water omitting external radiation from soil and water and dose from inhalation
  - b: Scenario where the radionuclide release has migrated through the three-layer sediment structure into the lake water including external radiation from soil and water and dose from inhalation.

3. Results

3.1. Dose assessment results

The dose assessment results are presented in Figure 3 displaying the results of the probabilistic runs for the full range of parameter values. There seems to be two types of behaviour regarding the doses: for ^{36}Cl, ^{129}I, ^{237}Np and ^{99}Tc the difference between cases 1 and 2 is rather small, on the other hand for ^{135}Cs, ^{94}Nb, ^{90}Sr and ^{238}U the difference is much more noticeable. Another observation is that the standard deviation of the dose conversion factors has increased significantly with the addition of uncertainties from the sediment layer structure. The impact of the addition of doses for inhalation and external radiation is most clearly seen with ^{94}Nb as the dose is higher in calculation case 2b compared with case 2a. The impact of sedimentation, causing the lake volume to decrease and the radionuclide concentration to increase, is presented in Figure 4. Again, there two types of behaviour regarding the results: the dose conversion does not increase for ^{36}Cl and ^{99}Tc. For the other radionuclides the dose conversion factor increases slightly over the years due to sedimentation. The results in Figure 4 were calculated using the most probable parameter values for case 2b.

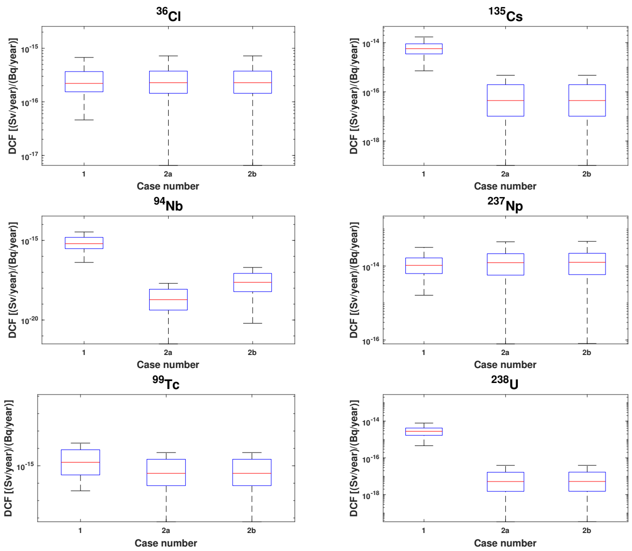

Figure 3. Dose conversion factor calculation results. The red line marks the median values, the bottom and top edges of the box correspond to the 25th and 75th percentiles, respectively and the whiskers extend to the most extreme data points not considered outliers. Case 1 in the figure is from (Pohjola et al. 2016) and cases 2a and 2b have been described in section 2.4.5.

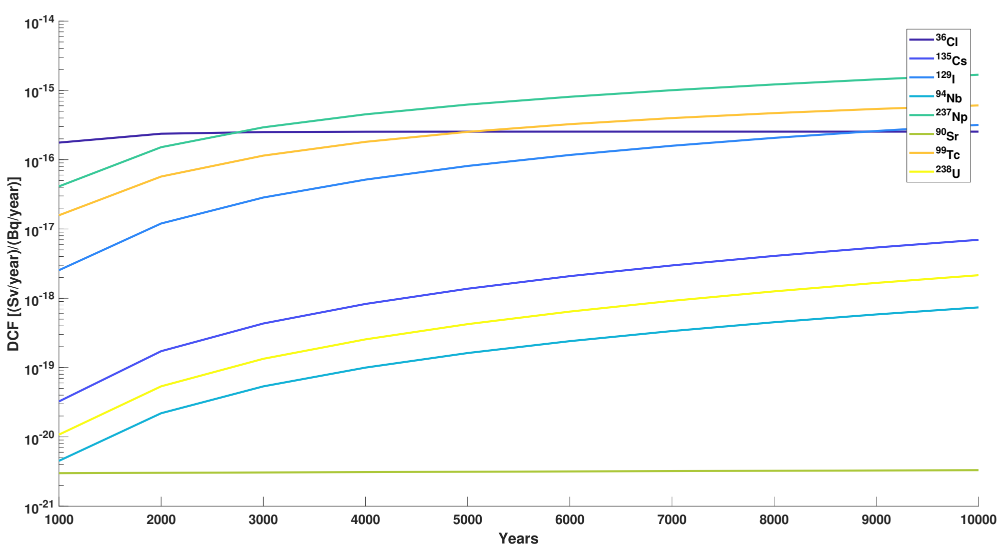

Figure 4. Dose conversion factor calculation results for case 2b.

In Figure 5 an inventory of the total radionuclide activities accumulated into different compartments within 10 000 years, taking also radioactive decay into account, is presented. In case 1 the compartment for deep sediment was not modelled and the sediment layers were only taken into consideration in terms of sedimentation and re-suspension. Due to its relatively short half-life, the majority of the release of ^{90}Sr has decayed. Otherwise two types of behavior can be noticed for cases 2a and 2b: either the release is sorbed in the sediment layers (^{135}Cs, ^{129}I, ^{94}Nb, ^{237}Np, ^{99}Tc and ^{238}U) or the majority of the release is outside the model after 10 000 years due to discharge from the lake (^{36}Cl).

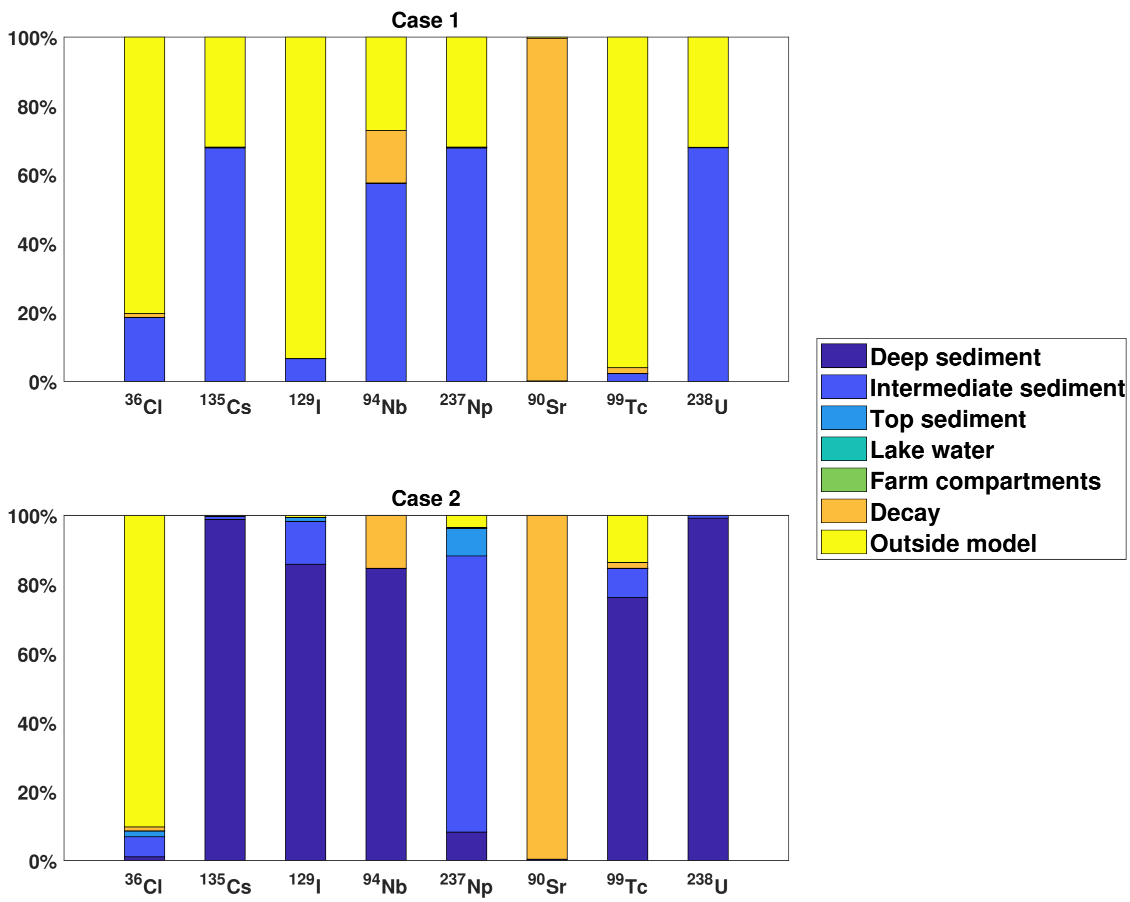

Figure 5. Inventory showing how the radionuclide release has accumulated into different compartments within 10 000 years (1 Bq/year).

3.2. Sensitivity analysis results

The results of the sensitivity analysis for ^{135}Cs and ^{94}Nb with the EFAST and Sobol methods are presented in Tables 2 and 3, respectively. The sensitivity analysis results for the other radionuclides are presented in Electronic supplement 4 in tables 1-6. The number in parentheses denotes the rank order of the significance of the parameter (1=the parameter having the greatest influence) and the following number denotes the influence (%).

Table 2. Sensitivity analysis results for ^{135}Cs. DS stands for the deep sediment, IMS the intermediate sediment layer, TS the top sediment, CR the soil-to-plant concentration ratio and K_d the solid-liquid distribution coefficient. The number in parentheses denotes the order of the parameter (1=the parameter having the greatest influence) and the following number denotes the influence (%).

Parameter	EFAST	Sobol
Bioaccumulation factor (fish)	(17) 2.0	(10) 0.0
Bulk density (DS)	(4) 8.0	(5) 7.3
Bulk density (IMS)	(11) 4.4	(4) 9.3
Bulk density (TS)	(8) 6.6	(2) 26.1
CR (other vegetables)	(20) 0.2	(11) 0.0
CR (pasture)	(14) 2.7	(12) 0.0
Intake rate (fish)	(9) 5.8	(7) 2.0
K_d (DS)	(3) 8.9	(6) 4.1
K_d (TS)	(18) 1.7	(13) 0.0
K_d (lake)	(13) 3.1	(14) 0.0
K_d (soil)	(10) 4.9	(15) 0.0
Lake volume	(15) 2.6	(16) 0.0
Mean depth (lake)	(19) 0.8	(17) 0.0
TF (milk)	(5) 7.7	(9) 0.0
Thickness (DS)	(1) 9.9	(18) 0.0
Thickness (IMS)	(2) 9.8	(1) 28.1
Thickness (TS)	(6) 7.5	(19) 0.0
Water exchange rate	(16) 2.3	(20) 0.0
Water flux (DS to IMS)	(12) 3.8	(3) 21.8
Water flux (TS to lake)	(7) 7.3	(8) 1.4

Table 3. Sensitivity analysis results for ^{94}Nb. For the abbreviations and the meaning of the different numbers in the table, see the caption of Table 2.

Parameter	EFAST	Sobol
Bioaccumulation factor (fish)	(9) 5.7	(12) 0.0
Bulk density (DS)	(2) 8.4	(13) 0.0
Bulk density (IMS)	(18) 2.5	(14) 0.0
Bulk density (TS)	(14) 3.7	(8) 0.5
CR (leguminous vegetables)	(10) 5.5	(7) 0.6
Intake rate (berries)	(16) 3.0	(15) 0.0
Intake rate (fish)	(11) 5.2	(9) 0.3
Intake rate (leguminous vegetables)	(15) 3.0	(11) 0.0
K_d (DS)	(7) 6.2	(3) 16.3
K_d (TS)	(12) 4.3	(16) 0.0
K_d (lake)	(4) 6.9	(5) 12.9
K_d (soil)	(19) 1.9	(17) 0.0
Lake volume	(17) 2.8	(6) 4.8
Mean depth (lake)	(6) 6.3	(10) 0.2
Thickness (DS)	(1) 9.1	(4) 13.2
Thickness (IMS)	(5) 6.6	(1) 27.0
Thickness (TS)	(3) 7.3	(2) 24.2
Water exchange rate	(8) 6.1	(18) 0.0
Water flux (DS to IMS)	(20) 1.4	(19) 0.0
Water flux (TS to lake)	(13) 4.1	(20) 0.0

For the majority of the studied radionuclides, parameters related to the lake bottom sediment layer structure were among the most contributing ones for the total outcome. The two exceptions to this were ^{36}Cl and ^{99}Tc where parameters related to the solid-liquid distribution in the soil and the root uptake from the soil had the largest role, respectively. In general, the order of the parameters in terms of their influence remains almost the same between the EFAST and the Sobol methods. In some cases, e.g. ^{99}Tc, the Sobol method emphasizes the influence of the most contributing parameter compared with the EFAST method.

4. Discussion and conclusions

Assessment of the radionuclide transport in the biosphere and the corresponding radiation doses is an important part of safety assessments of nuclear waste disposal. The purpose of this study was to analyze the dose to humans in a case where the radionuclide release is transported along with the ground water through a three-layer sediment structure into lake water that is then used for drinking, irrigation and watering cattle. This scenario was compared with an earlier study where the release enters the lake water directly.

The results from the dose calculations and the sensitivity analysis show a similar pattern in the behavior of the studied radionuclides. The addition of doses from external irradiation and inhalation of dust was important for ^{94}Nb. The most contributing parameters for the majority of the radionuclides are related to the physical quantities of the sediment layer structure, e.g. bulk density, solid-liquid distribution coefficient and thickness. Exceptions to this are ^{36}Cl and ^{99}Tc whose most contributing parameters are solid-liquid distribution coefficients in the soil and the concentration ratio from soil to 'other vegetables', respectively. With the other radionuclides (^{135}Cs, ^{129}I, ^{237}Np, ^{90}Sr and ^{238}U) there were only small differences in the total doses between the calculation cases differing in the source term location (directly into the lake water or through sediments), and whether or not the external and inhalation exposures were accounted for. This implies that for the other radionuclides the sediment layers act as a retardation layer limiting the exposure, whereas for ^{36}Cl and ^{99}Tc the doses are regulated rather by the properties of the agricultural system and the lake (see also Figure 5). It is also worthwhile to notice that while the mean of the dose calculation results can be quite similar, the distribution is much wider in case 2 compared with case 1.

The two total-order sensitivity analysis methods, EFAST and Sobol, gave similar results in terms of the order of the parameters. The emphasis of the parameter having the most influence in the Sobol method could be related to the computation routine. In Sobol the different occurrences of parameters fed into the model are decomposed into summands using the Fourier-Haar transform (Sobol 1993). The summands are orthogonal to each other but the numerical integration of summands (with respect to the output) may increase the influence of large components. It is stated in (Sobol 1993) that some functions may have an effect on numerical integration, leading to unexpected results.

In this paper a framework of a constant system to deal with the variability in dose estimates to humans that arises from the uncertainties in probabilistically estimated landscape properties is presented. Extension of this work to address relevant ecosystems other than the lake and farm system is being planned.

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