PROJECT 'PURCHASE OF STUDIES FOR THE PREPARATION OF A DESIGNATED SPATIAL PLAN AND THE ASSESSMENT OF IMPACT'

PROJECT PART: STUDIES NECESSARY FOR THE ESTABLISHMENT OF THE RADIOACTIVE WASTE REPOSITORY

ACTIVITY 2. Studies of the three repository locations

Sub-activity 2.19. Possible impact of the repository on neighbouring countries

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01	Initial issue	20-07-2023
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Definitions and abbreviations

IAEA	International Atomic Energy Agency
IMMSP NASU	Institute of Mathematical Machines and Systems Problems of the National Academy of Science of Ukraine
NSDF	Near Surface Radioactive Waste Disposal Facility
IDDF	Intermediate Depth Disposal Facility
PAL	Paldiski site
PED	Pedase site
ALT	Altkula site

Introduction

This report contains the results of an assessment of the transboundary transport of radionuclides that may penetrate through the walls of the radioactive waste disposal facility, which will be installed in Estonia, to groundwater and then to the Gulf of Finland as a result of accidental degradation of this disposal facility due to extreme natural or artificial events.

An estimation of the transboundary impacts is in line with the requirements of International Conventions and Treaties.

Estonia and neighbouring countries are members of the **Joint Convention** on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management. It is the first legal instrument to address the issue of radioactive waste management safety on a global scale. It does so by establishing fundamental safety principles. The Convention applies to radioactive waste resulting from the operation of civilian nuclear reactors and other civilian applications. It also applies to radioactive waste from military or defence programmes if such materials are managed within exclusively civilian programmes. In addition, the Convention covers planned and controlled releases into the environment of liquid or gaseous radioactive materials from nuclear facilities.

Estonia as well as neighbours Finland, Latvia and Sweden have also ratified the Convention on Environmental Impact Assessment in a Transboundary Context (**ESPOO Convention**). The Russian Federation has signed this Convention as well, however, did not ratify it yet. Procedures of this Convention require that a state of potential impact to be informed about the planned activity and, when the Environment Impact Assessment is performed, a consulting with this state is to be carried out.

Pursuant to Article 37 of the **EURATOM Treaty**, signed in 1957, each EU Member State is obliged to provide the Commission with General Data relating to any plan for the disposal of radioactive waste in whatever form as will make it possible to determine whether the implementation of such plan is liable to result in the radioactive contamination of the water, soil or air of another Member State. The European Commission recommends that the general data be submitted to the Commission before any authorization for the disposal of radioactive waste is granted by competent national authorities. The Commission is to deliver its opinion after consulting the group of experts. Commission Recommendation of 11 October 2010 on the application of Article 37 of the Euratom Treaty (2010/635/Euratom) defines the content of the General Data and their submission procedure. The 'disposal of radioactive waste' within the meaning of Article 37 of the Treaty covers any planned or accidental release of radioactive substances associated with the operations listed below, in gaseous, liquid or solid form in or to the environment:

- the predisposal management, including storage, of radioactive waste
- the dismantling of nuclear reactors, except research reactors whose maximum power does

not exceed 50 MW continuous thermal load;

- the emplacement of radioactive waste above or under the ground without the intention of retrieval.

Affected Member States are to be selected by taking into account distance, and the route of water courses for liquid effluent releases as well as wind direction for gaseous effluent releases (if any).

Trivial operations having no or negligible radiological impact in other Member States should not be submitted to the Commission. In cases where the exposure of the population in the vicinity of the site of interest is very low, this information may be sufficient for the assessment of the impact on other Member States.

The purpose of this work is to assess potential transboundary impact of radioactive waste disposal. Possible impacts and suitability of previously proposed three candidate sites (PAL, ALT and PED) is to be compared. Two types of the radioactive waste disposal facilities are planned to be installed in one site that will be finally selected: Near Surface Radioactive Waste Disposal Facility (NSDF) and Intermediate Depth Disposal Facility (IDDF). These disposal facilities will be underground, so if radioactive materials ever penetrate their walls, they will enter the groundwater. The groundwater flow is directed to the Gulf of Finland in all three considered sites. Contaminants entered the Gulf of Finland can be transported by currents for a large distance over the whole Baltic Sea defining the transboundary impact. There are no atmospheric or other pathways for the transfer of radioactive material from the disposal facility to the environment. Therefore, only marine transboundary impact was assessed.

The marine dispersion model POSEIDON-R was used for the assessment. It adopted fluxes of radionuclides that transported by groundwater flows to the Gulf of Finland as a source term and simulated the transfer of radionuclides in the marine environment providing effective doses to humans from consumption of contaminated seafood. Two source terms corresponding to NSDF and IDDF for selected locations were used in the calculations. These source terms were previously estimated by RESRAD model in frame of Sub-activity 2.16 (Safety Assessment Report).

1. Methodology

In this report, the assessment of the impact of transboundary marine transport of a radioactive release from the repository unit is carried out for Finland as the state closest to the potential source of the release. The minimum distance from the Estonian coast near the location of repository unit to the territory of Finland is about 65–75 km through the Gulf of Finland. The settlements on the coastal area of Finland in front of repository unit was chosen as the object for conducting assessments of environmental pollution and the consequences for public health. Since Finland is the nearest country to the repository unit location, the application of the conservative scenario assures the maximum possible dose of human irradiation. The conservative scenario includes the application of the maximum concentration of radionuclides reaching the coast through the groundwater according to the RESRAD model results to the considered coastline. This scenario overestimates the possible flux of radionuclides to the sea providing the maximum possible effective dose of exposure to a reference person living in the considered settlement. Increasing the distance from the release point, while maintaining all other conditions, can only reduce the dose. Therefore, if dose obtained under such conservative scenario for Finland is below the allowable limits, this assures that it will be below these limits for all other countries, which are at a greater distance, in particular Sweden (the shortest distance from the repository unit location to Sweden territory is 280 km), Latvia (distance to the country border along the coast exceeds 200 km), and Russian Federation (the shortest distance to the country border is 230 km).

Such an approach agrees with the European Commission Recommendation of 11 October 2010 on the application of Article 37 of the Euratom Treaty (2010/635/Euratom) defining that (1) affected Member States are to be selected by taking into account distance, wind direction for gaseous and aerosol releases and the route of water courses for liquid effluent releases, and (2) in cases where the exposure of the population in the vicinity of the site of interest is very low, this information may be sufficient for the assessment of the impact on other Member States.

Criteria for evaluating radioactive exposure. The main criterion for limiting the exposure of the population in Europe due to man-made sources is the limit of the individual effective dose (all routes of exposure), set at the level of 1 $mSv \cdot year^{-1}$ [1, 2]. In this report, the annual individual effective doses are estimated, and they are compared with the above-mentioned limit which is the main criterion for the general public safety.

Marine model. In the event of a natural or artificial degradation of repository's walls, radionuclides may enter the groundwater and then to the Gulf of Finland that may lead to the contamination of seawater and seafood. Therefore, the report includes the modelling of the migration of radionuclides in the Gulf of Finland, and the assessment of the corresponding contribution to the doses of internal exposure of the population due to the consumption of radioactively contaminated seafood.

This part of the problem was solved using the compartment model POSEIDON-R [3], which is a part of the European Decision Support System for emergency response to nuclear accidents RODOS [4, 5]. The POSEIDON-R model was recently validated for the Baltic Sea [6-8]. Model results agree well with measured concentrations of ^{137}Cs in water, bottom sediments and marine biota contaminated due to the Chornobyl accident. In the POSEIDON-R model, the transfer of different radionuclides in water, bottom sediments and biota are considered. Transfer of radionuclides to marine organisms is described by means of a dynamical uptake model BURN [9], taking into account the trophic level of the organisms.

Population dose calculation. Assessment of individual doses for the population is an important part of the radiation protection system. Information about doses serves as a criterion for making decisions regarding the implementation of certain protective measures.

The following pathways of the population radiation exposure dose are considered in the POSEIDON-R model:

- external exposure from radionuclides in the water during swimming and boating activities;
- external exposure from radionuclides on seashore when human is on the coast;
- internal exposure caused by the inhalation of radionuclides with the sea spray when human is on the coast;
- internal exposure caused by the consumption of contaminated seafood.

Among considered pathways of radiation exposure internal exposure caused by the consumption of contaminated seafood is the dominant. Usually, dose from the consumption of contaminated seafood is several orders of magnitude higher than doses from other pathways of irradiation [10]. Therefore, the estimation of individual effective doses for population is based on the doses caused by the consumption of contaminated seafood. These doses are calculated using concentrations of radionuclides in different marine organisms obtained in the POSEIDON-R model and annual consumption rates.

2. POSEIDON-R model and its customization for the Gulf of Finland

POSEIDON-R is the compartment model for the simulation of radionuclide transport in the marine environment including water sediments and biota, and the estimation of doses to humans from marine pathways of irradiation [3]. In the model, the marine environment is represented as a system of compartments for the water column, bottom sediment, and biota. The compartment-averaged radionuclide concentration is governed by a set of ordinary differential equations describing the temporal variations of concentration, the exchange with adjacent compartments, and with the suspended and bottom sediment, radioactive sources, and decay. The exchange between the water column boxes is described by fluxes of radionuclides due to advection, sediment settling, and turbulent diffusion processes. A detailed description of the model is given in the Appendix A.

POSEIDON-R uses the dynamic food web model for the simulation uptake of radionuclides by marine organisms. The model includes pelagic and benthic food chains. Pelagic organisms are grouped into phytoplankton, zooplankton, non-piscivorous and piscivorous fishes. Benthic organisms include deposit-feeding invertebrates, demersal fish, and bottom predators. Coastal predators feed both pelagic and benthic organisms in shallow waters, whereas detritus feeders and filter feeders are presented by crustaceans and molluscs, respectively.

The generic parameters of the POSEIDON-R model with the dynamical food chain model were validated on the measurement data in different seas and oceans [3, 7, 8, 11]. The sensitivity analysis was carried out in [7] to estimate parameter sensitivity. Detailed comparison of simulation by POSEIDON-R with other compartment models and Eulerian models of radioactivity transport in the Baltic Sea [6] demonstrated the reliability and robustness of the model. Recently the model was used for the analysis of possible consequences of a severe accident at the first Polish nuclear power plant to be constructed on the Baltic Sea coast [12].

In the POSEIDON-R model, the area of interest is covered by a system of compartments, which can be of different sizes and shapes. The transfer of radionuclides between compartments is modelled based on average currents in the region. Correct values of water fluxes between boxes, which are calculated from available 3D velocity fields, are important for model customization. Here 3D-currents from the circulation model NEMO-Nordic (available online at http://marine.copernicus.eu/services-portfolio/access-to-products/) are used. To check the water velocities provided by the NEMO-Nordic model, they are compared with available measurement data in the locations of 4 buoys near the coast of Poland. Details of this comparison are given in the report for Sub-activity 4.11.

The resolution of the NEMO-Nordic model is 1/30 degree in latitude and 1/18 degree in longitude which is about 3.7 km in both directions. Based on this resolution, the optimal size of compartments (boxes) in the box model POSEIDON-R is 15x15 km (4x4 calculation nodes). Such compartments were created in the Gulf of Finland around potential locations of the repository unit between the Estonian and Finnish coasts (Fig. 2.1). Larger boxes were placed around them to prevent excessive mixing of contamination in the large volumes of seawater. The volume and average depth of each box were calculated based on the bathymetry data. Deep boxes were vertically subdivided on a surface layer (from surface to a depth of 25 m) and a bottom layer (from a depth of 25 m to the bottom) to describe the activity stratification in the water column.

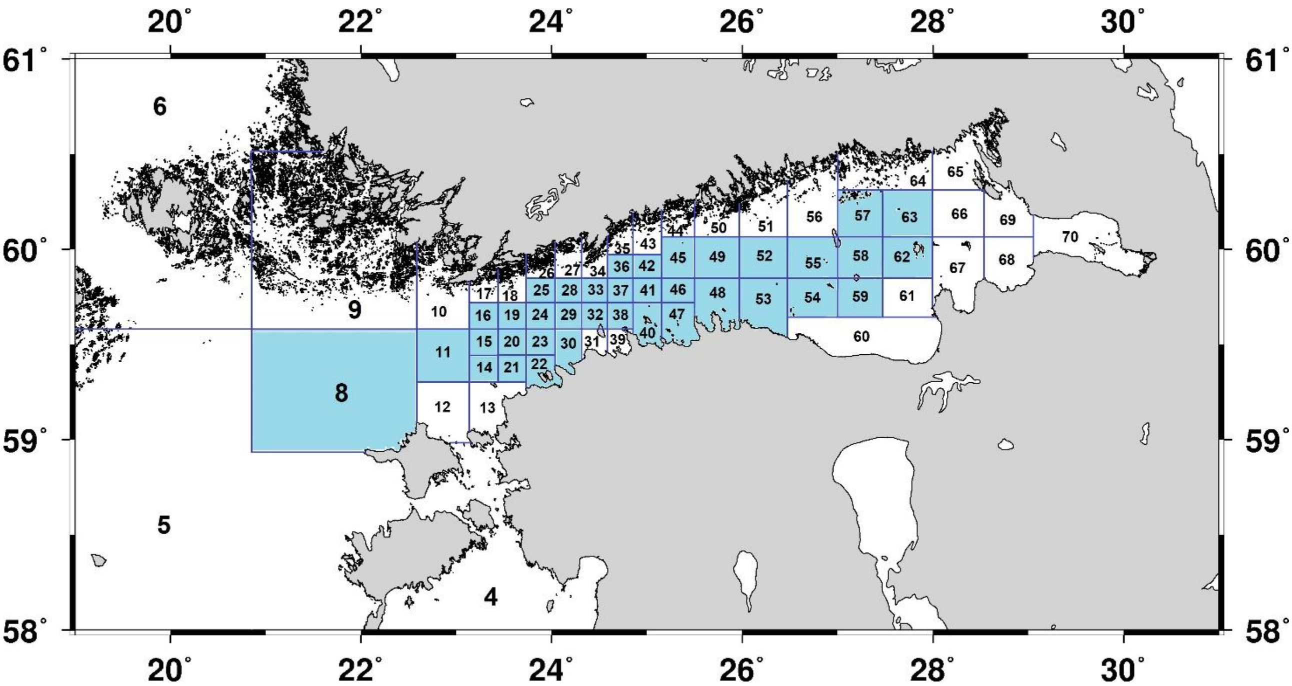


Fig. 2.1. Box system in the region of interest. Blue boxes are divided on two vertical layers.

The water fluxes between boxes were calculated by averaging currents on their faces over a 10-year period (2009-2018) from the NEMO-Nordic circulation model. The water inflow of main rivers (Neva, Narva, Kymijoki) was also taken into account to have the correct dominant flow of water from the Gulf of Finland to the Baltic Sea. Parameters describing the water-sediment interaction in each box such as suspended sediment concentration and sedimentation rate (see Table 2.1), which are typical for the Gulf of Finland, were adopted from [13] and [14]. Default values for other parameters (see Table 2.1) needed for the modelling were taken from the study [7] where the agreement between calculated and measured concentrations of ^{137}Cs in various components of the Baltic Sea was achieved. The salinity of the Baltic Sea is lower than ocean salinity due to large river runoff and low water exchange with the Atlantic Ocean. It increases the uptake of radionuclides (especially isotopes of Cs and Sr) by marine organisms [8, 15] due to decreasing competition ions concentration. In the model, salinity changes from 1.5 in the Neva Bay to 8 in the western part of the Gulf according to [16].

Table 2.1

Parameters of water-sediment exchange of radionuclides in the POSEIDON-R model.

Parameter	Value
Suspended sediment concentration in the water column, kg m^{-3}	0.006
Sedimentation rate, kg m^{-2} y^{-1}	1.5
Thickness of the top (active) sediment layer, m	0.1
Bioturbation coefficient, m^2 y^{-1}	$3.6 \cdot 10^{-5}$
Diffusion coefficient, m^2 y^{-1}	0.0315
Bottom sediments porosity	0.75
Density of sediment particles, kg m^{-3}	2600

Data for consumption rates of marine organisms are needed for the estimation of the doses to people from seafood consumption. According to FAO (Food and Agriculture Organization of the United Nations) data, the average annual human consumption of fish is 30.5 kg (https://www.fao.org/fishery/docs/DOCUMENT/fcp/en/FI_CP_FI.pdf). The statistical data from Faostat shows that changes in fish consumption rates in Finland was not significant during the last years (https://www.helgilibrary.com/indicators/fish-consumption-per-capita/finland/). However, this value includes the consumption of domestic marine and freshwater species and the consumption of imported fish. For conservative dose assessment, we assume that there is a reference person (group) that consumes all fish from the Gulf of Finland.

The new box system (see Fig. 2.1) was integrated into the JRODOS interface where all calculations are performed.

3. Scenarios for the release of radionuclides to the Gulf of Finland

Fig. 3.1 shows three locations for the NSDF and IDDF that were previously selected: Paldiski site (PAL), Pedase site (PED) and Altkula site (ALT). Here we can see that the PAL site is located closest to the seashore. In addition, this site is closest to Finland. Therefore, we consider it the worst location in terms of its impact on neighbouring countries. The groundwater flows of radionuclides were calculated by the RESRAD model in the frame of Sub-activity 2.16. Directions of these flows are schematically shown in the Fig. 3.2. Here we use the maximum concentration of radionuclides that reach the coast through the groundwater and apply it to the selected part of coastline, which is about 1 km (blue lines in the Fig. 3.1). The length of this coastline is in agreement with hydrogeological conditions, which are analyzed in Sub-activity 2.6. Such approach is the basis of the conservative scenario. It overestimates the possible flux of radionuclides to the sea providing the assessment from above for the effective dose of exposure to a reference person living in a settlement in a neighboring country.

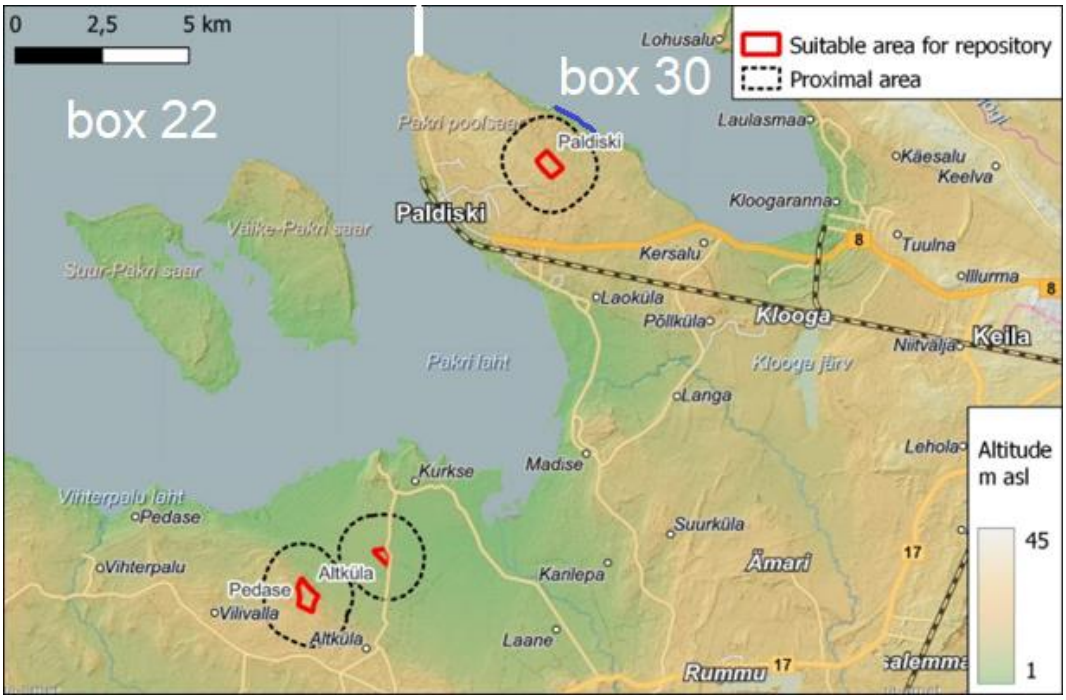


Fig. 3.1. Previously selected locations of the repository units with indication of box numbers, which are separated by white solid line, according to system of boxes described above. Blue lines along the coast shows the part of the coast (approximately 1 km) where source of radionuclides due to the groundwater flux is considered in the POSEIDON-R model.

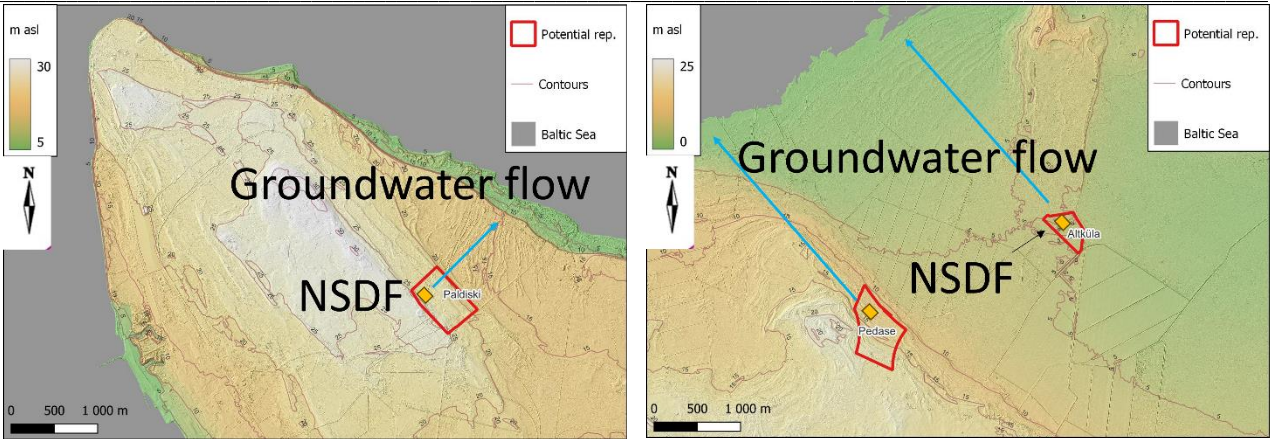


Fig. 3.2. Schematic directions of the groundwater flows (blue arrows) for different locations of the repository unit.

In general, there are no large differences between 3 considered sites in case of ordinary transfer of radionuclides by groundwater. The difference could be between NSDF and IDDF types of repository units due to different composition of stored radionuclides, their activities and groundwater flows at different depths. Therefore, both types are considered. The ALT site differs from other sites because it is located in the lowland and can be flooded due to rise of sea level caused by climate change in the coming centuries, as mentioned in Sub-activity 2.4. This means that there is the theoretical probability of destroying the NSDF by storms and flooding by seawater followed by dissolution of the waste in saline sea water. Hypothetical all waste inventory dissolution scenario could be considered as “absolutely worst” case in the assessment.

Thus, three scenarios for the release of radionuclides in marine environment are considered:

1) release of radionuclides from the NSDF to the sea by groundwater;
2) release of radionuclides from the IDDF to the sea by groundwater;
3) release of radionuclides from the NSDF directly to the sea resulted from its destroying by storms and flooding due to rise of sea level (relevant for ALT site only).

3.1. Release of radionuclides from the NSDF to the sea by groundwater

Fluxes of radionuclides from the NSDF located at the PAL site to the Gulf of Finland (Fig. 3.3) were assumed as a maximum radionuclide concentrations obtained by the RESRAD model at the coast multiplied by the volume of groundwater that flows to the sea through the part of the coast highlighted by blue line in the Fig. 3.1. According to the RESRAD model (Sub-activity 2.16), the annual groundwater flow through this part of the coast was estimated as 306,563 m^3. The calculated flux of radionuclides was set up as a source term for box 30 (see Fig. 2.1)

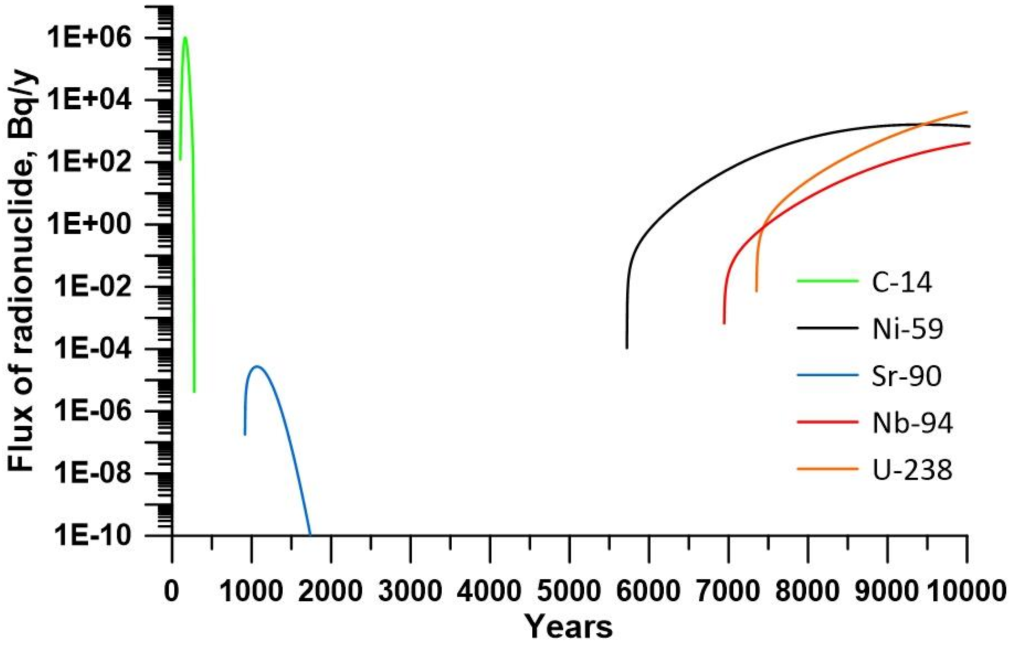


Fig. 3.3. Fluxes of radionuclides for the NSDF located at the PAL site used as a source term for the box 30 in the POSEIDON-R model.

Fig. 3.3 includes only radionuclides, which penetrate soil with groundwater and reach the sea. They are long-lived radionuclides with low (^{14}C) or moderate (^{59}Ni, ^{94}Nb, ^{238}U) ability to be adsorbed by soil particles and ^{90}Sr that is relatively short-lived radionuclide but with low ability to adsorption. Due to short half-life, ^{90}Sr will almost completely decay during 920 years needed for its transfer from the NSDF to the sea by groundwater even if its initial inventory in the NSDF is quite large (see Table 3.1). Therefore, the flux of ^{90}Sr to the Gulf of Finland will exist but it will be very small. Other radionuclides will decay or will be completely adsorbed by soil particles on the way from the NSDF to the sea with groundwater flows. For example, ^{137}Cs, which is the radionuclide with maximal inventory in the NSDF (see Table 3.3), has the moderate ability to adsorption, therefore it needs more time to reach the sea by groundwater, and it will completely decay for this time. Such radionuclides as isotopes of plutonium and americium have very high ability to adsorption, therefore they cannot reach the sea by groundwater. Inventories of considered radionuclides in the NSDF and their activity released to the sea during 10,000 years according to results of Sub-activity 2.6 are presented in Table 3.1.

As can be seen in Fig. 3.3 and in Table 3.1, each radionuclide is characterized by different travel time from the repository unit to the sea by groundwater. It depends on the property of radionuclide to be adsorbed by soil particles. For example, ^{14}C has low ability for the adsorption. Therefore, it reaches the sea much faster than other radionuclides.

Table 3.1

Parameters of radionuclides considered in the scenario of the release from the NSDF located at the PAL site.

Nuclide	Half-life (y)	Inventory in the NSDF (Bq)	Released activity to the sea (Bq)	Period of time needed to reach the sea (y)
^{14}C	5730	6.1E+08	4.9E+07	105
^{59}Ni	76,550	4.7E+07	2.9E+06	5720
^{90}Sr	28.8	1.3E+12	6.7E-03	920
^{94}Nb	20,316	3.8E+07	2.7E+05	6950
^{238}U	4.468E+09	5.8E+08	2.2E+06	7350

Radionuclide fluxes from the NSDF located at the ALT or PED sites to the Gulf of Finland would not differ significantly from the fluxes from the PAL site. Since the distance of the PAL site from the sea is the shortest (Fig. 3.2), while the other radionuclide transport conditions differ insignificantly, it was assumed that the PAL site represents a conservative case.

3.2. Release of radionuclides from the IDDF to the sea by groundwater

Fluxes of radionuclides from the IDDF to the Gulf of Finland by groundwater was assumed similarly to NSDF as a maximum radionuclide concentration obtained by the RESRAD model at the coast multiplied by the volume of groundwater that flows to the sea through the same part of the coast (highlighted by blue line in the Fig. 3.1). According to the RESRAD model, the annual groundwater flow through this part of the coast was estimated as 307,600 m^3 for PAL site (Sub-activity 2.16). The calculated flux of radionuclides was set up as a source term for box 30 (see Fig. 2.1)

List of radionuclides that reach the sea by groundwater and their activity released to the sea according to results of Sub-activity 2.6 are given in the Table 3.2. There is ^{210}Pb in the list, which is not among radionuclides to be disposed of. It will appear as a decay product of ^{226}Ra. Other radionuclides from the IDDF will completely decay or will have zero or very low concentration due to adsorption by soil particles on the way from the IDDF to the sea with groundwater flows. As can be seen in Fig. 3.3 and in Table 3.1, each radionuclide is characterized by different travel time from the repository unit to the sea by groundwater. It depends on the property of radionuclide to be adsorbed by soil particles. The maximum flux of radionuclides to the sea will take place approximately 25,000 years after the start of their release from the IDDF (Fig. 3.4). Therefore, simulations by the POSEIDON-R model will continue for 25,000 years.

Table 3.2

Parameters of radionuclides considered in the scenario of the release from the IDDF located at the PAL site.

Nuclide	Half-life (y)	Inventory in the NSDF (Bq)	Released activity to the sea (Bq)	Period of time needed to reach the sea (y)
^{59}Ni	76,050	1.3E+12	1.5E+11	9200
^{94}Nb	20,300	1.1E+11	2.9E+09	11,100
^{210}Pb	22.2	-	8.9E+09	3300
^{226}Ra	1600	2.6E+10	1.3E+10	3300
^{238}U	4.46E+09	4.6E+12	6.5E+11	13,000

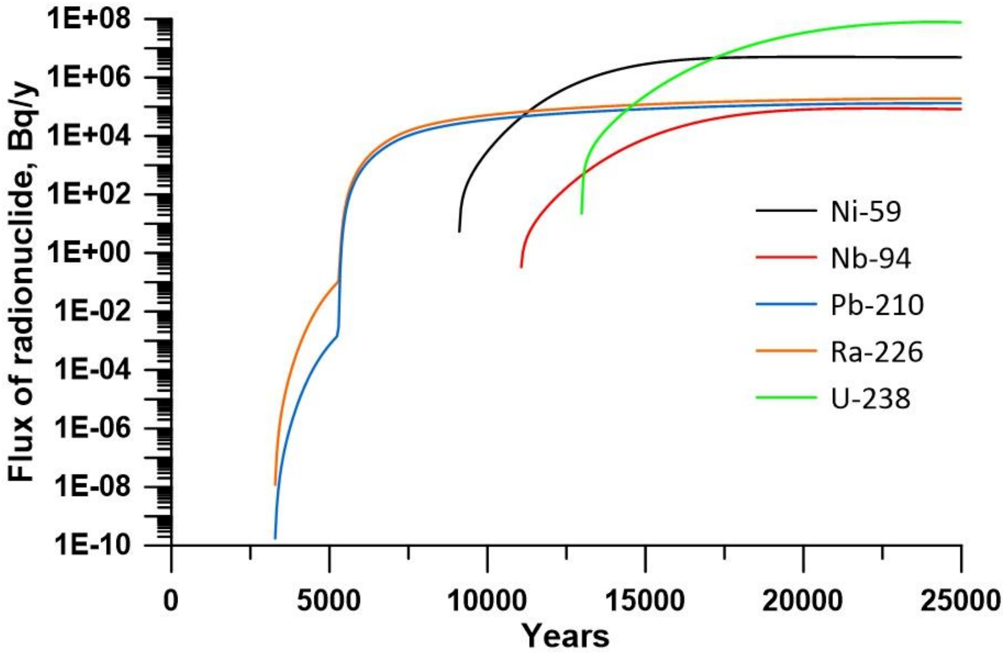


Fig. 3.4. Fluxes of radionuclides for the IDDF located at the PAL site used as a source term for the box 30 in the POSEIDON-R model.

Radionuclide fluxes from the IDDF located at the ALT or PED sites to the Gulf of Finland would not differ significantly from the fluxes from the PAL site due to similar geological structure. There are two main differences only: depth and distance. Other properties of the soil (Sub-activities 2.8 and 2.9) resulting radionuclide retention by the soil are very similar for all three sites. Since the distance of the PAL site from the sea is the shortest (Figure 3.2), and the expected disposal is the least deep, it can be stated that the PAL site represents the most conservative case. PAL site is followed by ALT and PED sites.

3.3. Release of radionuclides from the NSDF located at the ALT site directly to the sea

The topographical analysis carried out under Sub-activity 2.4 identified a negative aspect of the ALT site: due to climate warming, it is possible that the area will be flooded by the sea or affected by severe storms in the next century. Saline water would accelerate degradation of concrete structures, waste matrix and leaching of the radionuclides. For the potential impact assessment it was assumed that this hypothetical event happens immediately after end of active institutional control (during the period of active institutional control lasting the first 100 years after disposal, the integrity of the disposal facility should be maintained through the implementation of corrective measures). The accelerated degradation of engineered barriers could take several decades, instead of hundreds of years as foreseen in the disposal concept. However, it was conservatively assumed that the affected barriers degrade immediately (i.e. after single catastrophic event) and all inventory disposed of in the NSDF flows to the sea (sorption and retention of radionuclides in decomposed concrete and surrounding soil are neglected).

For the time of the hypothetical site flooding, the inventory stored in the NSDF will decrease due to radioactive decay. Remained inventory is considered as a source term for box 22 (see Figs. 2.1 and 3.1) in the POSEIDON-R model. Details are given in Table 3.3.

Table 3.3

Parameters of radionuclides considered in the scenario of the release from the NSDF located at the ALT site directly to the sea due to flooding.

Nuclide	Half-life (y)	Estimated inventory in the NSDF (Bq)	Decay corrected activity released to the sea after 100 years of storage (Bq)
H-3	12.3	6.2E+10	2.2E+08
C-14	5730	6.1E+08	6.0E+08
Co-60	5.275	9.3E+09	1.8E+04
Ni-59	76,050	4.7E+07	4.7E+07
Ni-63	100.2	2.2E+09	1.1E+09
Sr-90	28.9	1.3E+12	1.2E+11
Nb-94	20,300	3.8E+07	3.8E+07
Ba-133	10.551	5.1E+05	7.2E+02
Cs-137	30.05	2.4E+13	2.4E+12
Eu-152	13.55	5.9E+08	3.5E+06
Eu-154	8.599	9.4E+07	3.0E+04
Ra-226	1600	3.9E+09	3.7E+09

Ra-228	5.75	9.0E+03	5.2E-02
Th-232	1.4E+10	1.4E+05	1.4E+05
U-234	2.449E+05	2.2E+03	2.2E+03
U-238	4.46E+09	5.8E+08	5.8E+08
Pu-238	87.7	2.3E+05	1.0E+05
Pu-239	24,400	5.0E+05	5.0E+05
Pu-240	6570	1.4E+05	1.4E+05
Am-241	432.2	5.9E+09	5.0E+09

4. Results of simulation for the radioactive contamination of the marine environment

The flow of radionuclides by groundwater is very slow process with time scale of thousand years. Input data used in POSEIDON-R model for the groundwater flow of radionuclides from NSDF and IDDF at the PAL site covers 10,000 years and 25,000 years respectively with maximum release rates at the end of simulation period (see Figs. 3.3-3.4). Therefore, for these scenarios the POSEIDON-R model is applied for the same period, and results of simulations are provided for the end of simulations. For the scenario with flooding of NSDF at the ALT site, all radionuclides will be dissolved in seawater during relatively short period (days – weeks – months), and the maximum impact on the marine environment and human will take place during first years after such event. In the case of gradual degradation of the NSDF and dissolution of the radionuclides after the flooding, release rates of radionuclides to the sea will be lower, providing much lower concentrations in all components of marine environment and lower doses to human.

4.1. Scenario for the release of radionuclides from the NSDF located at the PAL site by groundwater

The highest concentration of all radionuclides will be near the Estonian coast in the box 30 where the contaminated groundwater flows. According to Fig. 3.3, ^{14}C will reach marine environment faster than other radionuclides, approximately 100 years after the start of radionuclide release from the NSDF. Simulation results show that the maximal concentration of ^{14}C in water will be about 4×10^{-6} and 3×10^{-5} Bq/m^3 near Finnish and Estonian coast, respectively (Fig. 4.1), 180 years after the start of radionuclide release from the NSDF. Water currents in the region of interest are directed mostly from the Gulf of Finland to the Baltic Sea defining the dominant direction for the transport of radionuclides. In addition, this leads to their dilution by large amount of seawater. Therefore, radionuclide concentrations in the main part of the Sea will be much lower. Concentrations of ^{14}C in bottom sediments and fish also will be very low. Thus, maximal concentration of ^{14}C in bottom sediments will be 1.8×10^{-5} Bq/kg, while in pelagic non-predatory fish it will be 1.7×10^{-5} Bq/kg (Fig. 4.1).

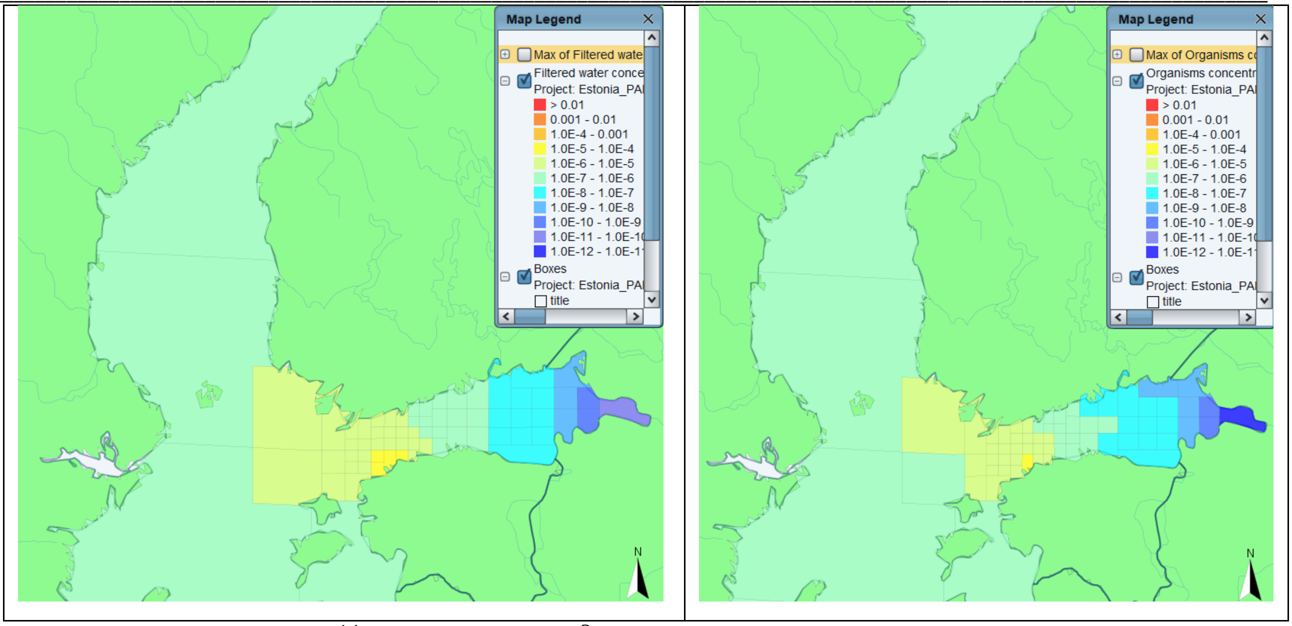


Fig. 4.1. Concentrations of ^{14}C in water [Bq/m^3] (left) and pelagic non-predatory fish [Bq/kg] (right).

^{59}Ni and ^{238}U will be the dominant radionuclides much later (see Fig. 3.3 and Table 3.1). This means that their concentrations in all components of marine environment will exceed concentrations of other radionuclides at that time. However, these concentrations will be very low – the maximum in water will not exceed 10^{-7} Bq/m^3 for ^{59}Ni and 10^{-6} Bq/m^3 for ^{238}U as shown on the Fig. 4.2. Water currents in the region of interest are directed mostly from the Gulf of Finland to the Baltic Sea defining the dominant direction for the transport of radionuclides. In addition, this leads to their dilution by large amount of seawater. Therefore, radionuclide concentrations in the main part of the Sea will be much lower.

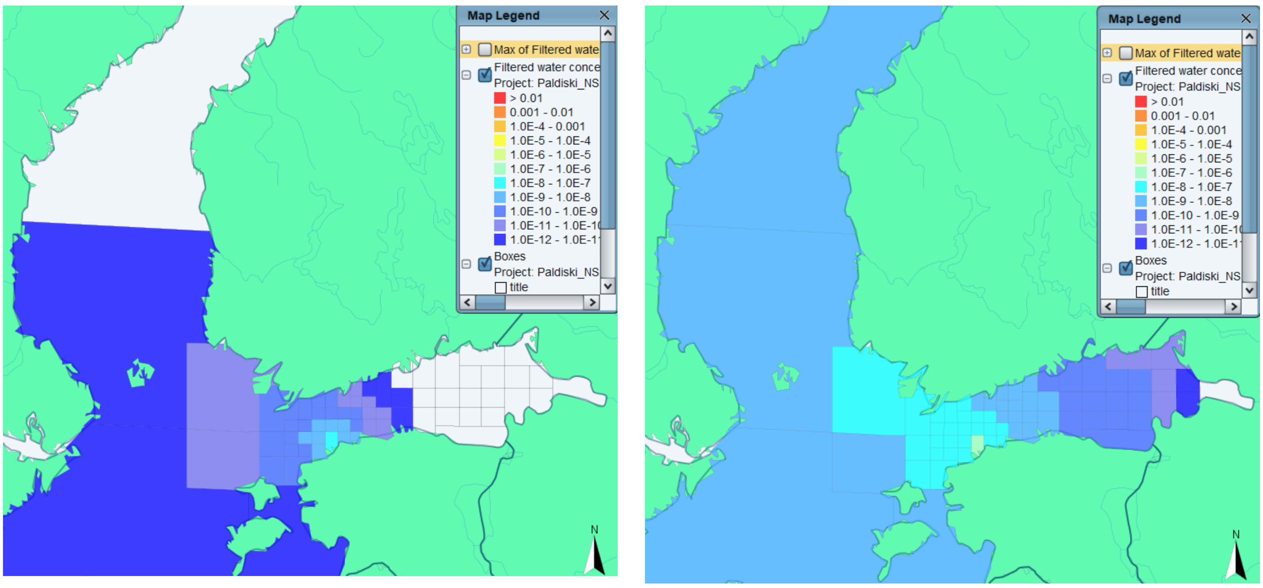


Fig. 4.2. Concentrations of ^{59}Ni (left) and ^{238}U (right) in water [Bq/m^3] at the end of simulations.

After entering water, radionuclides start to interact with suspended and bottom sediments. All radionuclides have different ability to be adsorbed by sediments. For example, ^{59}Ni is adsorbed better than ^{238}U. The main part of ^{59}Ni will be deposited near the point of release, while ^{238}U will be transported by water over longer distances. The concentration of ^{59}Ni in bottom sediments will be approximately 1 order of magnitude higher than concentration of ^{238}U (Fig. 4.3), although the opposite relation occurs in water (Fig. 4.2). But, as in the case of water, the concentration of radionuclides in bottom sediments will be very low and not exceed 10^{-6} Bq/kg.

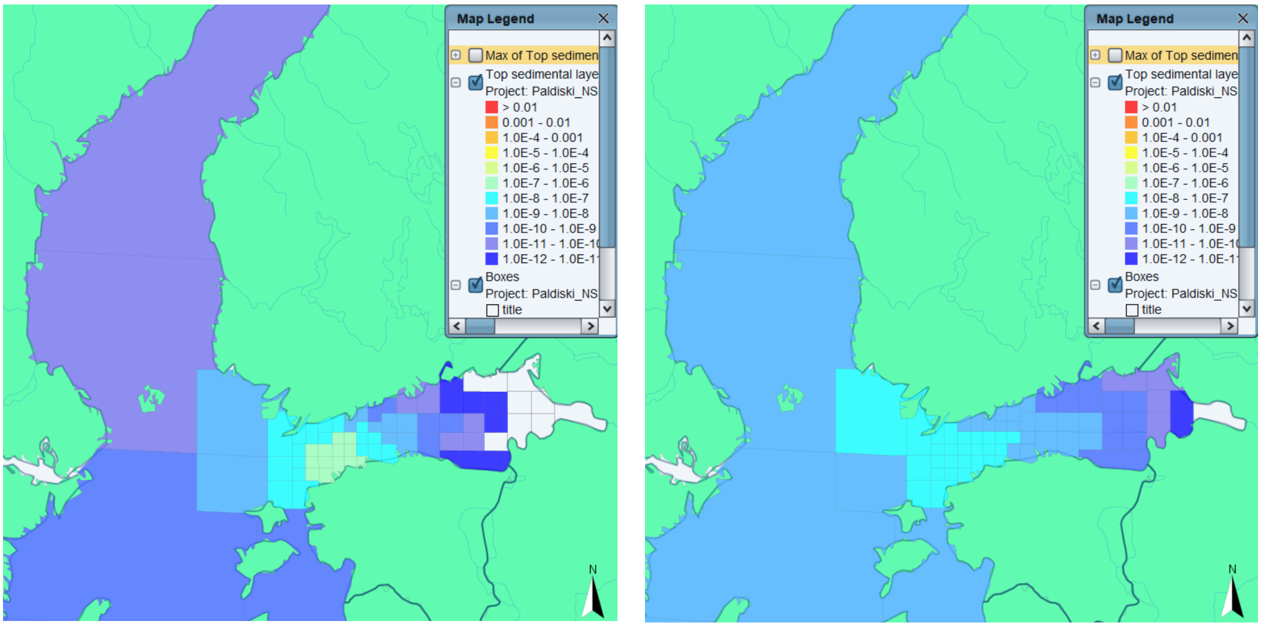


Fig. 4.3. Concentrations of ^{59}Ni (left) and ^{238}U (right) in bottom sediments [Bq/kg] at the end of simulations.

For most radionuclides and heavy metals, there is an inverse relationship between trophic levels and the concentration of radionuclide in aquatic organisms [17]. This means that the higher concentration of radionuclides will be in organisms from lower trophic levels. Therefore, we show the result of simulations (Figs. 4.4-4.5) for pelagic and demersal non-predatory fish. The concentration of radionuclides in them will be higher than in predatory types of fish, which are considered in the POSEIDON-R model. As we can see, the concentration of both radionuclides in demersal fish (Fig. 4.5) will be higher than in pelagic (Fig. 4.4), and concentration of ^{238}U in fish will be higher than concentration of ^{59}Ni. But again, all obtained concentrations are very low and only reach 10^{-8} Bq/kg.

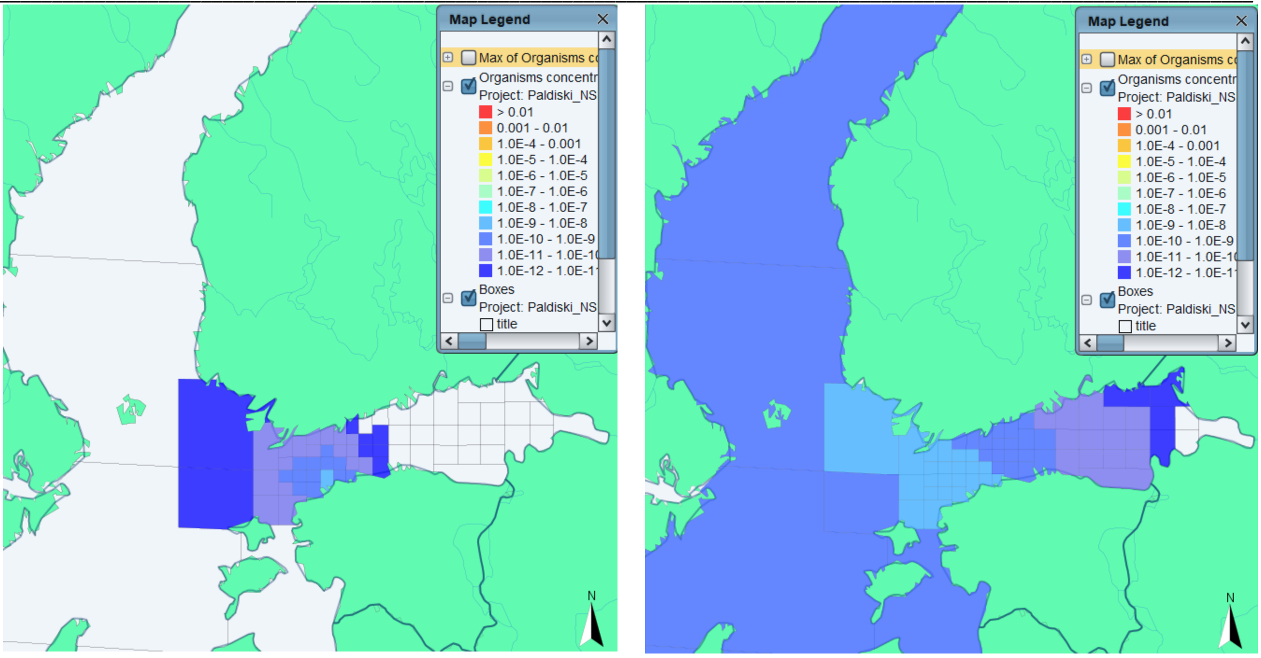


Fig. 4.4. Concentrations of ^{59}Ni (left) and ^{238}U (right) in pelagic fish [Bq/kg] at the end of simulations.

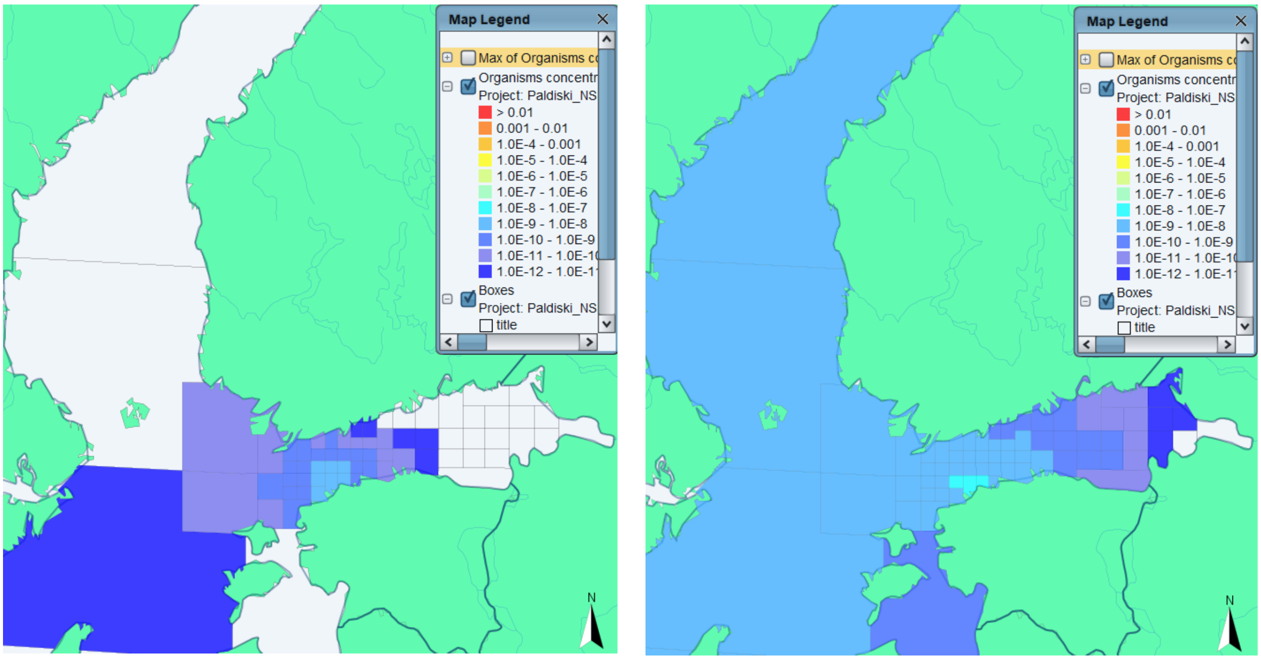


Fig. 4.5. Concentrations of ^{59}Ni (left) and ^{238}U (right) in demersal fish [Bq/kg] at the end of simulations.

4.2. Scenario for the release of radionuclides from the IDDF located at the PAL site by groundwater

Similar to the scenario with the release of radionuclides from the NSDF, the highest concentration of all radionuclides resulted from their release from the IDDF by groundwater will be near the Estonian coast in the box 30. The same is that ^{59}Ni and ^{238}U will be the dominant radionuclides with concentrations higher than concentrations of other radionuclides in all components of marine environment. Obtained in the modelling concentrations are somewhat higher than in NSDF scenario but they are still quite low – the maximum in water will be 5×10^{-5} Bq/m^3 for ^{59}Ni and 2×10^{-3} Bq/m^3 for ^{238}U near Estonian coast and 1.2×10^{-6} Bq/m^3 for ^{59}Ni and 4×10^{-4} Bq/m^3 for ^{238}U near Finnish coast (Fig. 4.6).

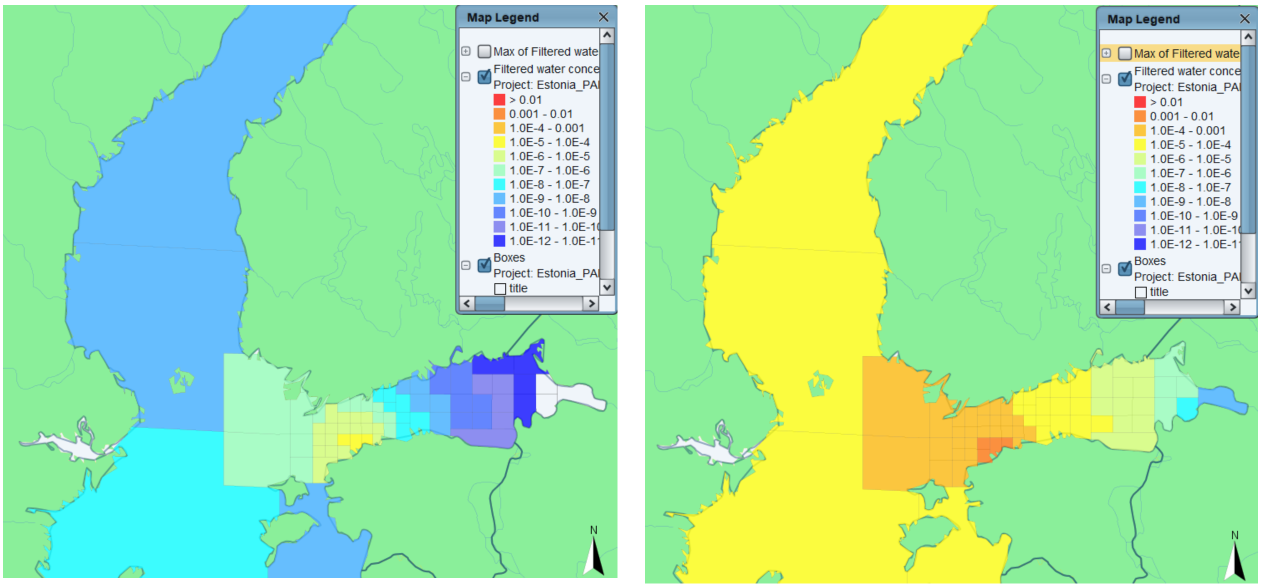


Fig. 4.6. Concentrations of ^{59}Ni (left) and ^{238}U (right) in water [Bq/m^3] at the end of simulations.

Similar to the scenario with the release of radionuclides from the NSDF, the main part of ^{59}Ni will be deposited near the point of release, while ^{238}U will be transported by water over longer distances. The concentration of ^{59}Ni in bottom sediments will be approximately the same as concentration of ^{238}U (Fig. 4.3), although in water concentration of ^{238}U was obtained 40 times higher (Fig. 4.2). But, as in the case of water, the concentration of radionuclides in bottom sediments will be quite low, around 10^{-3} Bq/kg.

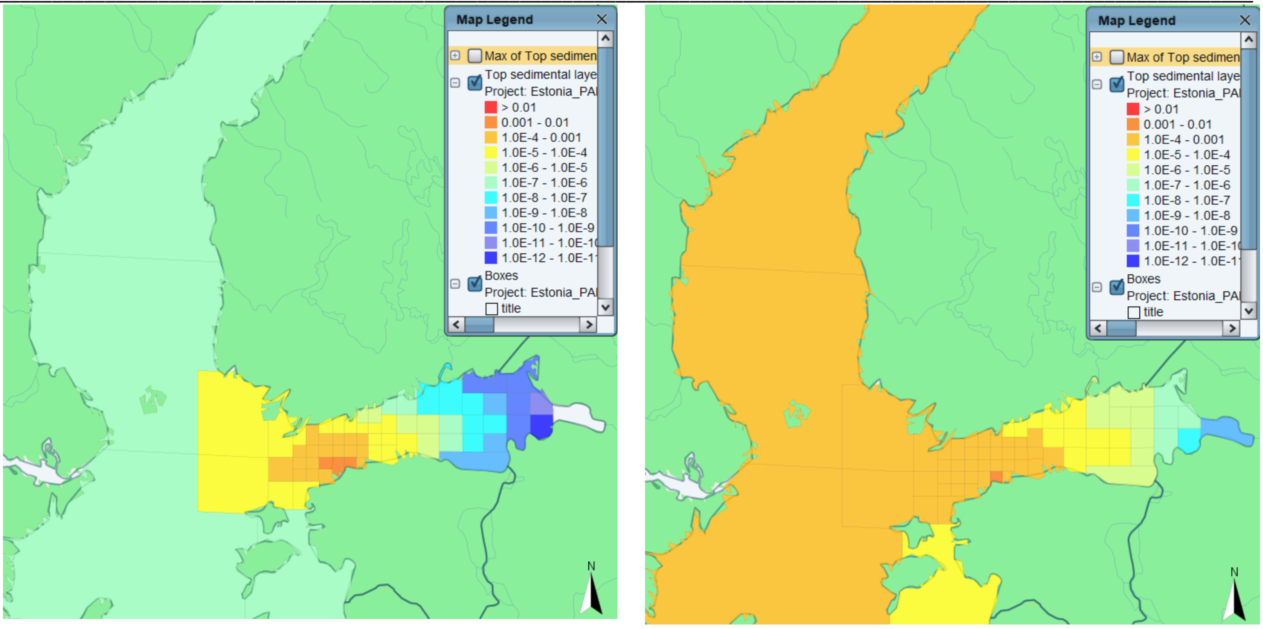


Fig. 4.7. Concentrations of ^{59}Ni (left) and ^{238}U (right) in bottom sediments [Bq/kg] at the end of simulations.

Results of simulations indicate that the concentration of both radionuclides in demersal fish (Fig. 4.5) will be higher than in pelagic (Fig. 4.4), and concentration of ^{238}U in fish will be higher than concentration of ^{59}Ni. But again, all obtained concentrations are very low, in the range 10^{-5} to 10^{-4} Bq/kg.

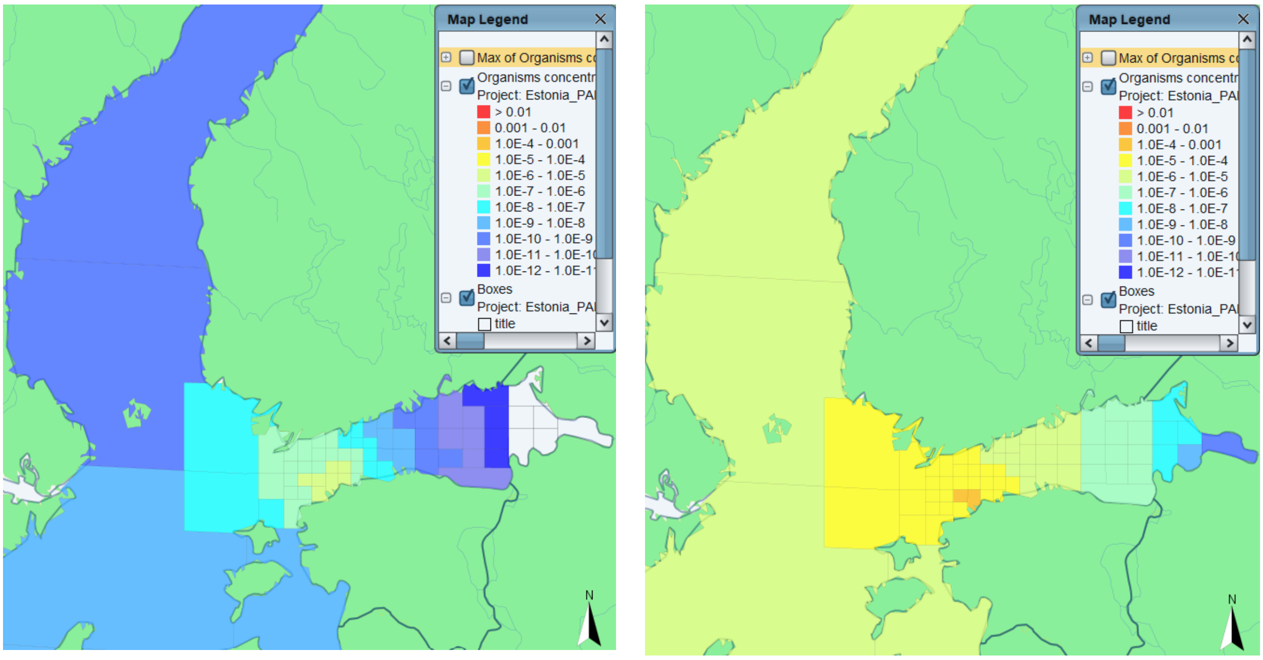


Fig. 4.8. Concentrations of ^{59}Ni (left) and ^{238}U (right) in pelagic fish [Bq/kg] at the end of simulations.

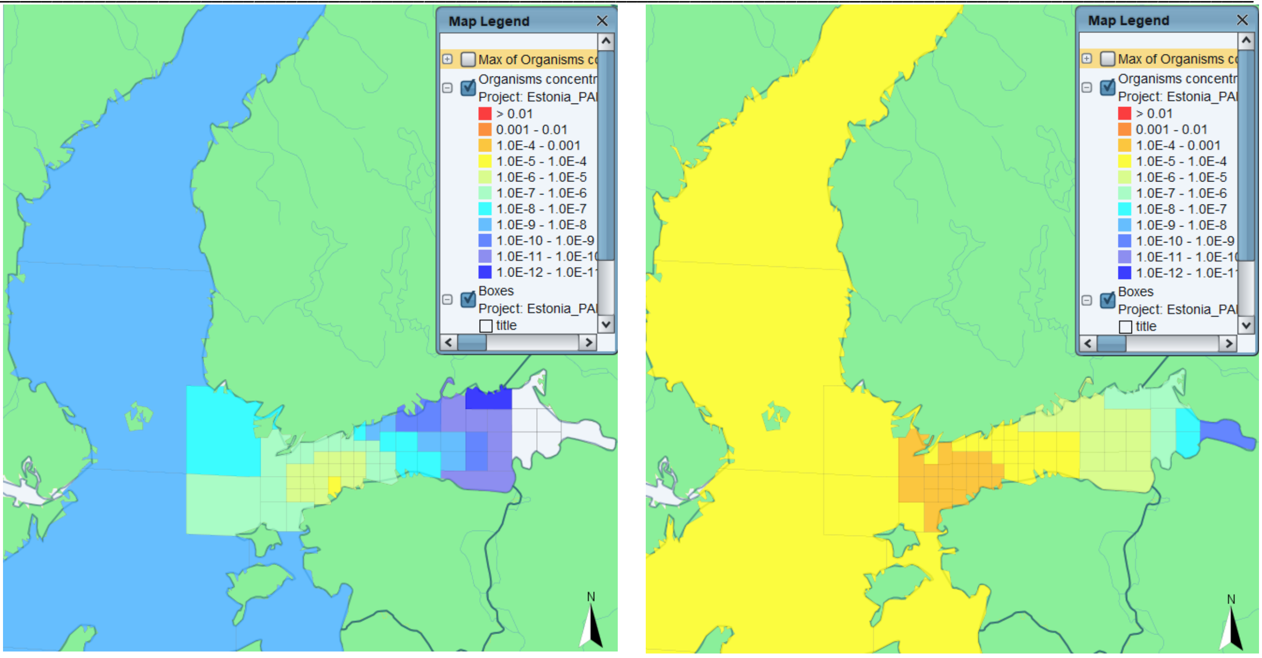


Fig. 4.9. Concentrations of ^{59}Ni (left) and ^{238}U (right) in demersal fish [Bq/kg] at the end of simulations.

4.3. Scenario for the release of radionuclides from the NSDF located at the ALT site due to flooding

According to Table 3.3, ^{137}Cs will be the radionuclide with maximum release rate. Therefore, detailed results of modelling are given for ^{137}Cs, while results for other nuclides are summarized in the Table 4.1. As we can see on the Fig. 4.10 a, the maximum concentration of Cs-137 in water will be near the coast of Estonia with the value of 434 Bq/m^3 that is much larger than background concentration of ^{137}Cs in the Baltic Sea about 15 Bq/m^3. The maximum concentration near the Finnish coast will be 4.25 Bq/ m^3 approximately one year after flooding of the ALT site (Fig. 4.10 b). Latter concentration will gradually decrease due to dilution of contamination in the Baltic Sea (Fig. 4.10 c-d).

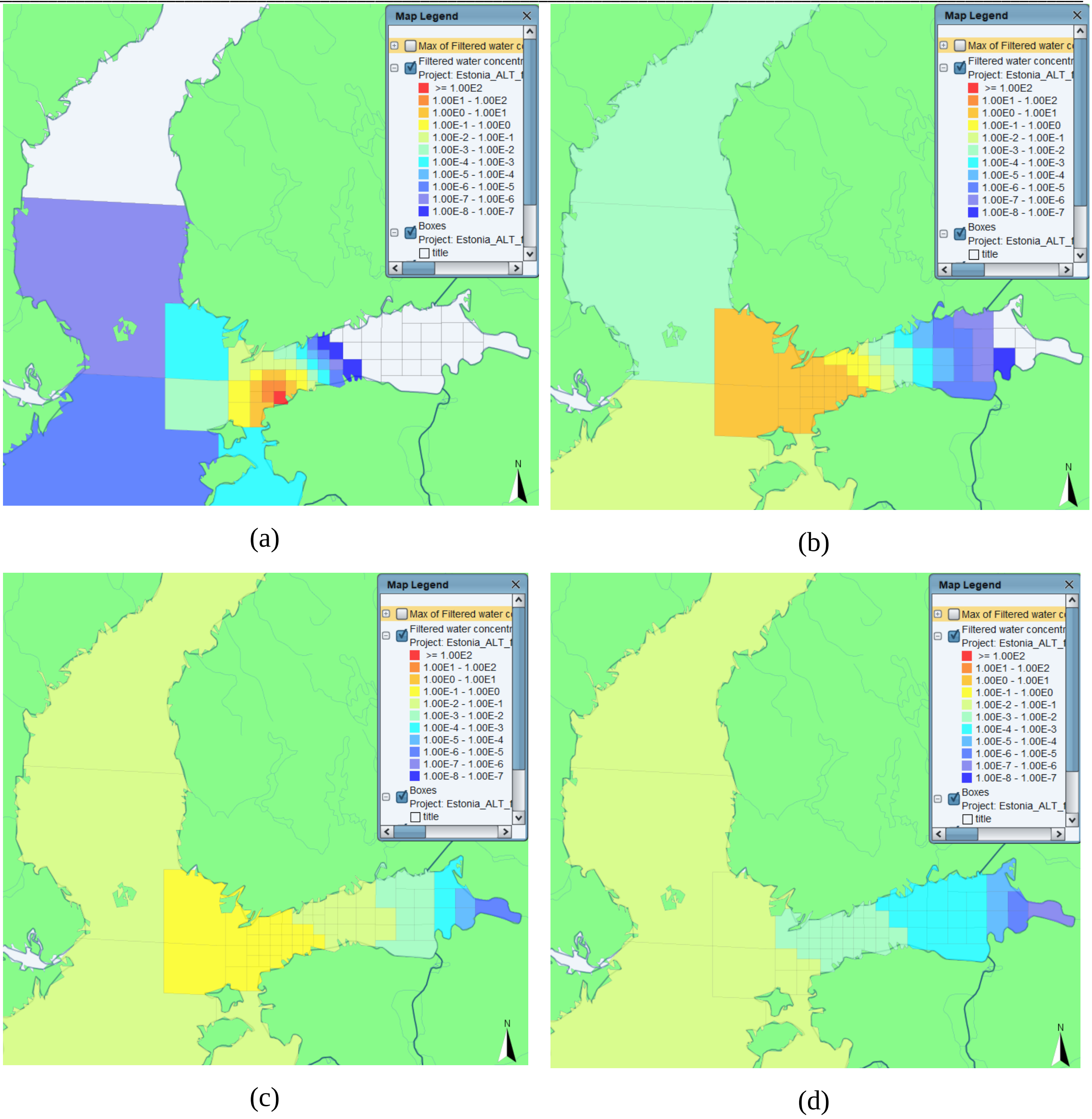
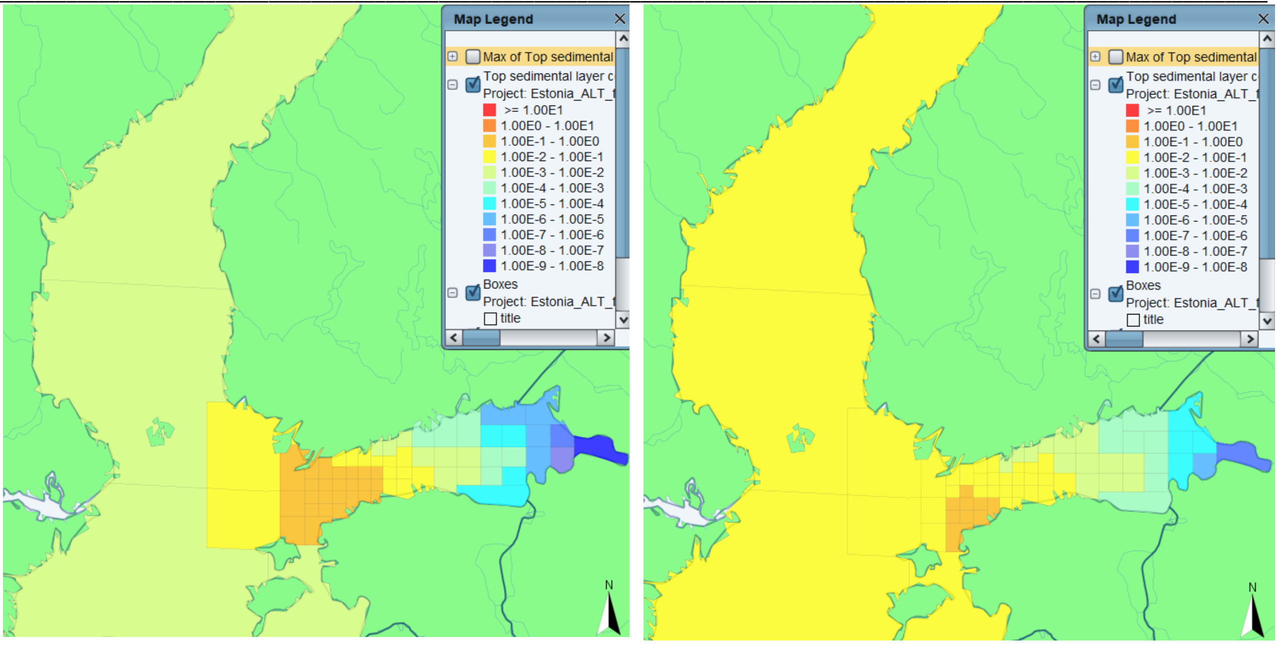


Fig. 4.10. Concentrations of ^{137}Cs in water [Bq/m^3] 1 month (a), 1 year (b), 5 years (c), 20 years (d) after flooding of the ALT site.

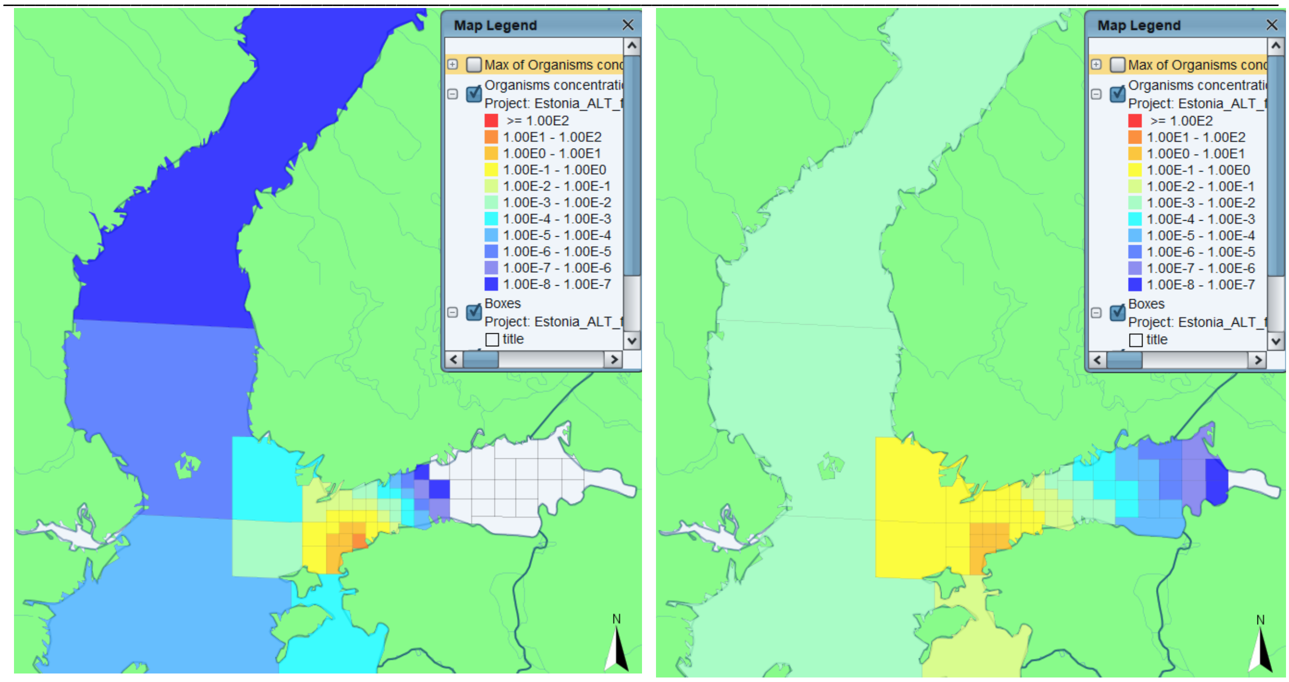
The contamination of bottom sediments is different for each radionuclide and depends on their ability to be adsorbed by sediments. The common conception is that radionuclides will remain much longer in bottom sediments than in water. For ^{137}Cs, the maximum contamination of bottom sediments will occur approximately 3 years after flooding of the ALT site (Fig. 4.11 a). It will decrease only 6 times in 40 years (Fig. 4.11 b).



(a) (b)

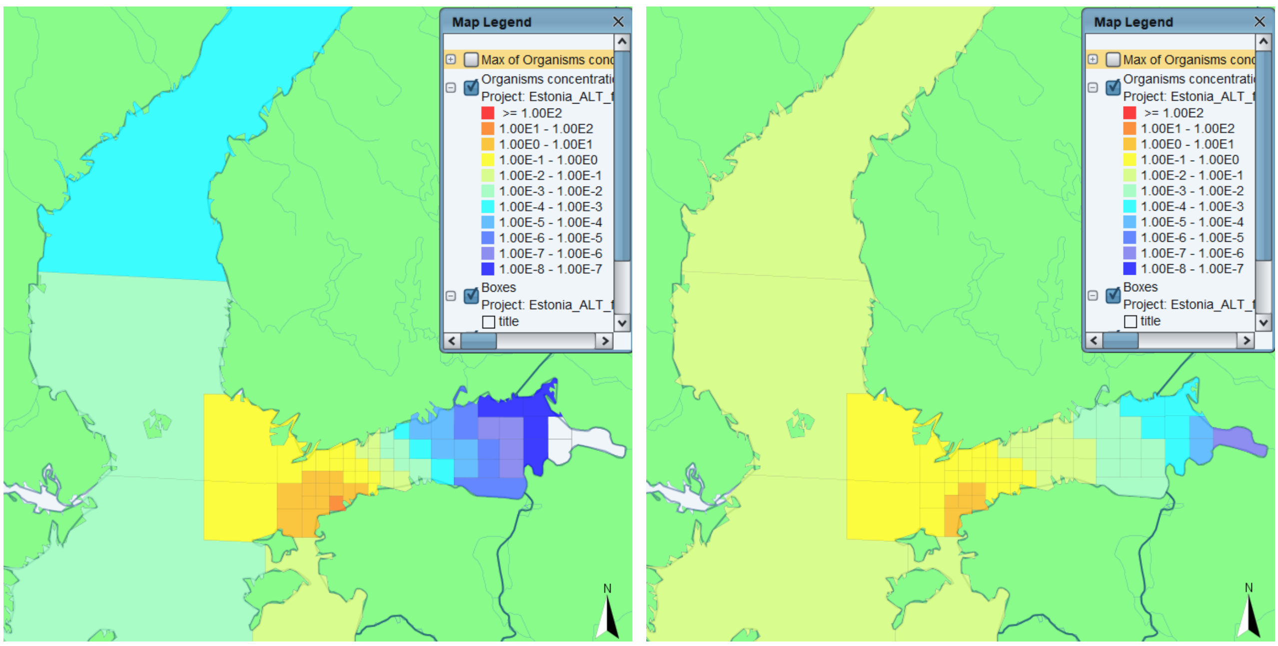
Fig. 4.11. Concentrations of ^{137}Cs in bottom sediments [Bq/kg] 3 years (a) and 40 years (b) after flooding of the ALT site.

Different types of fish also will be contaminated. Initially pelagic non-predatory fish will have the highest concentration of ^{137}Cs that can reach 21.5 Bq/kg near Estonian coast approximately 4 months after flooding of the ALT site (Fig. 4.12 a). But it will decrease 10 times 4 years later (Fig. 4.12 b). Later concentration of ^{137}Cs in predatory types of fish will become higher than in nonpredatory. It will reach 22.8 Bq/kg near Estonian coast 1.5 years after flooding of the ALT site (Fig. 4.13 a) and decrease 10 times after 5 years (Fig. 4.13 b). But such relation is valid only for ^{137}Cs. For other nuclides, concentration in predatory species will be always lower than in non-predatory species due to an inverse relationship between trophic levels and the concentration of radionuclide in aquatic organisms for most radionuclides and heavy metals [17]. Near Finnish coast the maximum concentration of ^{137}Cs in fish will be around 1.3 Bq/kg.



(a) (b)

Fig. 4.12. Concentrations of ^{137}Cs in pelagic non-predatory fish [Bq/kg] 4 months (a) and 4 years (b) after flooding of the ALT site.



(a) (b)

Fig. 4.13. Concentrations of ^{137}Cs in predatory fish [Bq/kg] 1.5 years (a) and 5 years (b) after flooding of the ALT site.

Maximum concentrations of radionuclides in different components of marine environment based on results of modelling are given in Table 4.1. Concentrations in other radionuclides from Table 3.3 will be much lower.

Table 4.1

Obtained in simulations maximum concentrations of radionuclides. Values in brackets show time needed to reach this concentration after flooding of the ALT site.

Nuclide	Water, Bq/m^3	Bed sediments, Bq/kg	Fish, Bq/kg
^{3}H	0.04 (1 month)	2.1E-06 (9 months)	5.5E-04 (4 months)
^{14}C	0.11 (1 month)	2.7E-04 (4 years)	0.038 (4 months)
^{63}Ni	0.11 (1 month)	0.014 (1.5 years)	0.012 (4 months)
^{90}Sr	22 (1 month)	0.016 (3 years)	0.33 (6 months)
^{95}Nb	1.3E-03 (1 month)	9.3E-4 (1 year)	4.3E-05 (4 months)
^{137}Cs	434 (1 month)	0.99 (3 years)	22.8 (1.5 years)
^{152}Eu	1.2E-04 (1 month)	8.3E-05 (9 months)	3.9E-05 (4 months)
^{226}Ra	0.65 (1 month)	3.2E-03 (4 years)	0.26 (6 months)
^{238}U	0.11 (1 month)	1.6E-04 (5 years)	6.8E-04 (6 months)
^{239}Pu	7.4E-05 (1 month)	9.3E-06 (1.5 years)	2.6E-04 (4 months)
^{241}Am	0.05 (1 month)	0.15 (1 year)	2.0E-05 (2 months)

5. Calculated maximum exposure doses for the population of Finland

Exposure doses to humans from seafood consumption were calculated based on simulated by POSEIDON-R model concentrations of radionuclides in marine organisms described in Chapter 4. Doses from other pathways of irradiation such as doses from swimming and boating activities, activity on seashore and inhalation of sea spray are usually several orders of magnitude less than doses from seafood consumption [10]. Very low doses were obtained for the scenarios of groundwater flow of radionuclides from the NSDF and IDDF located in the PAL site. For the release of radionuclides from the NSDF, a maximum annual dose will be in the range 1.4×10^{-9} to 4×10^{-8} microSv for Finnish people living in settlements in front of the PAL site. For Estonian people the corresponding maximal annual dose will be in the range 6.2×10^{-9} to 2.6×10^{-7} microSv. For the release of radionuclides from the IDDF, a maximum annual dose will be around 8.4×10^{-5} microSv for Finnish people and 4.4×10^{-4} microSv for Estonian people. Note that all doses were obtained for the reference person consuming all fish in their diet from the Gulf of Finland as it was described in Chapter 2. Obtained in simulations doses to human from seafood consumption, which can be caused by groundwater flow of radionuclides from the NSDF and IDDF to the Gulf of Finland, are absolutely negligible and are far below all allowable limits.

In the case of hypothetical flooding of the NSDF located at the ALT site due to rise of sea level caused by climate change in coming centuries, the maximal annual dose to human from seafood consumption will be 6.7 microSv for the first year after flooding (Fig. 5.2). Such a dose will be received by Estonian people living near the ALT site. Maximal annual dose to Finnish people will be 0.37 microSv for second year after flooding. So, even dissolving by seawater of all radionuclides placed in NSDF after 100 years of storage will not provide doses to human that exceed allowable limits in Estonia and in neighboring countries. However, this will lead to the release of a significant activity of radionuclides into the marine environment.

Doses for all considered scenarios are summarized in Table 5.1. In the first scenario, two periods are separated: initial phase (approximately 180 years after the start of release) when ^{14}C flows to the sea by groundwater, and late phase (approximately 10,000 years after the start of release) when the highest concentration of other radionuclides in marine environment takes place. In the last scenario, the flooding happened after 100 years of radioactive materials storage, with the highest dose obtained during the first years thereafter.

Based on the simulation results, it can be said that none of the selected locations for NSDF and IDDF will have negative impact on neighboring countries. But there is a probability of flooding of NSDF at the ALT site in future that can lead to release of a significant activity of radionuclides into the marine environment. Therefore, it is better to avoid placing NSDF at the ALT site.

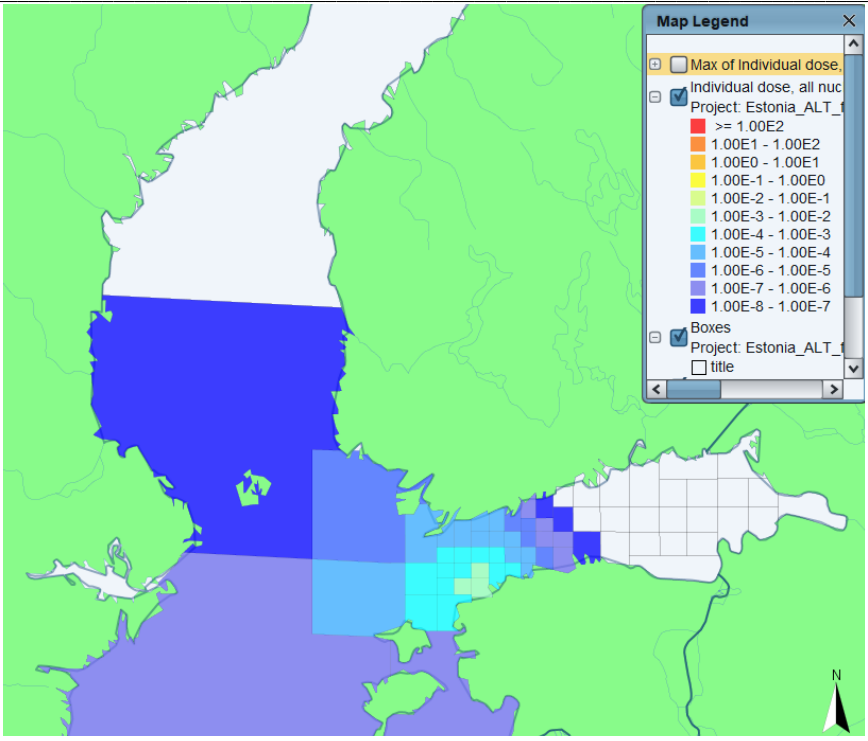


Fig. 5.1. Effective annual doses to human due to seafood consumption from all considered radionuclides [mSv/y] for the first year after hypothetical flooding of the NSDF located in the ALT site.

The water of the Gulf of Finland flows into the main part of the Baltic Sea (in the direction of Sweden and Latvia). Despite this, the concentrations of radionuclides in these parts as well as the calculated doses are a couple of orders of magnitude lower (Fig.5.1).

Table 5.1

Maximum doses to human from seafood consumption. Values in brackets show time from the installing the corresponding repository unit.

Scenario	Maximum annual dose to human from seafood consumption, microSv	
	Resident of Finland	Resident of Estonia
Release of radionuclides from NSDF located at the PAL site by groundwater	4×10^{-8} from ^{14}C (180 yrs) 1.4×10^{-9} (10,000 yrs)	2.6×10^{-7} from ^{14}C (180 yrs) 6.2×10^{-9} (10,000 yrs)
Release of radionuclides from IDDF located at the PAL site by groundwater	8.4×10^{-5} (25,000 yrs)	4.4×10^{-4} (25,000 yrs)
Release of radionuclides from NSDF located at the ALT site directly to the sea due to flooding	0.37 (100 + 2 yr)	6.7 (100 + 1 yr)

Conclusions

1. Simulation of the transboundary transport of radionuclides released from the radioactive waste disposal facility, which will be installed in Estonia, to the Gulf of Finland by groundwater as a result of degradation of this disposal facility due to natural or artificial events was carried out using the POSEIDON-R model that allows to estimate the levels of radioactive contamination of the marine environment, as well as the associated population exposure doses from the seafood consumption.

2. Obtained in simulations doses to human from seafood consumption, which can be caused by groundwater flow of radionuclides from the Near Surface Radioactive Waste Disposal Facility (NSDF) and Intermediate Depth Disposal Facility (IDDF) to the Gulf of Finland, are very low and are far below all allowable limits. For the release of radionuclides from the NSDF, a maximum annual dose was obtained in the range 1.4×10^{-9} to 4×10^{-8} microSv for Finnish people living in settlements in front of the potential disposal sites. For Estonian people the corresponding maximal annual dose will be in the range 6.2×10^{-9} to 2.6×10^{-7} microSv. For the release of radionuclides from the IDDF, a maximum annual dose will be around 8.4×10^{-5} microSv for Finnish people and 4.4×10^{-4} microSv for Estonian people. Note that all doses were conservatively obtained for the reference person consuming all fish in their diet from the Gulf of Finland.

3. In the scenario of hypothetical flooding of the NSDF located at the ALT site due to rise of sea level resulted from climate change in coming centuries, the maximal annual dose to human from seafood consumption will be 6.7 microSv for the first year after flooding. Such a dose will be received by Estonian people living near the ALT site. Maximal annual dose to Finnish people will be 0.37 microSv for second year after flooding followed degradation of the facility.

4. Calculations of radioactive contamination of water, bottom sediments and fish in the Gulf of Finland and the corresponding doses to the population as a result of the transboundary transport of radioactive releases from the planned NSDF and IDDF by groundwater showed very low, almost negligible, effects on the environment and public health in Finland. Even dissolving by seawater of all radionuclides placed in NSDF due to flooding of the ALT site caused by the sea level rise resulted from climate chance after 100 years of storage will not provide doses to human that exceed allowable limits in Estonia and in neighboring countries. However, this will lead to the release of a significant activity of radionuclides into the marine environment. The calculations were carried out for Finnish people living in settlements located at a distance of about 65–75 km from the potential release source through the Gulf of Finland. The distance from the source to the borders with other neighbouring states by the marine pathways is about 200 km to Latvia, 280 km to Sweden, and 230 km to Russian Federation. With an increase in the distance, the concentrations of radionuclides in all components of marine environment and the associated exposure doses to the population will decrease. So, it can be said with certainty that the conclusions about the fulfilment of the established safety criteria for the

population, obtained as a result of calculations for Finland, will be all the more true for other countries.

5. Based on the simulation results, it can be said that none of the selected locations for the disposal facility will have significant negative impact on neighboring countries. All three sites are acceptable for construction of the disposal facility, because the radiation protection limits would not be violated. However, ALT site is not recommended seeking to implement the radiation protection optimization principle. Also, the decision to dispose of the wastes at the ALT site can be interpreted as a violation of the London Convention (Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter), which bans the dumping into sea of radioactive wastes.

6. PAL and PED sites are evaluated nearly equally suitable: the associated exposure doses in the neighboring countries would be significantly below the exemption level.

References

1. Council Directive 96/29/Euratom of 13 May 1996 laying down basic safety standards for the protection of the health of workers and the general public against the dangers arising from ionizing radiation. http://data.europa.eu/eli/dir/1996/29/oj.
2. 2010/635/Euratom: Commission Recommendation of 11 October 2010 on the application of Article 37 of the Euratom Treaty (OJ L 279 23.10.2010, p. 36, ELI: http://data.europa.eu/eli/reco/2010/635/oj).
3. Maderich V., Bezhenar R., Tateda Y., Aoyama M., Tsumune D., Jung K.T., de With G. The POSEIDON-R compartment model for the prediction of transport and fate of radionuclides in the marine environment. MethodsX 5, 2018, 1251–1266.
4. Lepicard S., Heling R., Maderich V. POSEIDON-R/RODOS models for radiological assessment of marine environment after accidental releases: application to coastal areas of the Baltic, Black and North Seas. Journal of Environmental Radioactivity 72, 2004, 153–161.
5. Bezhenar R., Heling R., Ievdin I., Iosjpe M., Maderich V., Willemsen S., de With G., and Dvorzhak A. Integration of marine food chain model POSEIDON in JRODOS and testing versus Fukushima data. Radioprotection 51(HS2), 2016, S137–S139.
6. Periañez R., Bezhenar R., Iosjpe M., Maderich V., Nies H., Osvath I., Outola I., de With G. A comparison of marine radionuclide dispersion models for the Baltic Sea in the frame of IAEA MODARIA program. Journal of Environmental Radioactivity 139, 2015, 66–77.
7. Bezhenar R., Jung K.T., Maderich V, Willemsen S., de With G., Qiao F. Transfer of radiocaesium from contaminated bottom sediments to marine organisms through benthic food chain in post-Fukushima and post-Chernobyl periods. Biogeosciences 13, 2016, 3021–3034.
8. Maderich V., Bezhenar R., Tateda Y., Aoyama M., Tsumune D. Similarities and differences of ^{137}Cs distributions in the marine environments of the Baltic and Black seas and off the Fukushima Dai-ichi nuclear power plant in model assessments. Marine Pollution Bulletin 135, 2018, 895–906.
9. Heling R., Koziy L., Bulgakov V. A dynamical approach for the uptake of radionuclides in marine organisms for the POSEIDON model system. Radioprotection 37(C1), 2002, 833–838.
10. Maderich V., Bezhenar R., Kovalets I., Khalchenkov O., Brovchenko I. Long-term contamination of the Arabian Gulf as a result of hypothetical Nuclear Power Plant accidents. Journal of Marine Science and Engineering 11(2), 2023, 331.
11. Bezhenar R., Maderich V., Schirone A., Conte F., Martazinova V. Transport and fate of ^{137}Cs in the Mediterranean and Black Seas system during 1945-2020 period: A modelling study. Journal of Environmental Radioactivity 208-209, 2019, 106023.
12. Pylypenko O., Bezhenar R., Kivva S., Kopka P., Potempski S., Wojciechovicz H., Zheleznyak M. Application of the hydrological model chain of the RODOS decision support system for nuclear emergencies to the analysis of possible consequences of severe accident. Annals of Nuclear Energy 188, 2023, 109823.
13. Kankaanpaa H., Vallius H., Sandman O., Niemisto L. (1997). Determination of recent sedimentation in the Gulf of Finland using ^{137}Cs. Oceanologica Acta 20(6), 823–836.
14. Alari V., Erm A., Vali G., Lips I., Lips U. (2008) Optical properties of North-Eastern Baltic Sea in spring and summer 2007. Proceedings of IEEE/OES US/EU Baltic International Symposium, Tallinn, Estonia, 27-29 May 2008. pp. 1–7.
15. Heling R., Bezhenar R. Modification of the dynamic radionuclide uptake model BURN by salinity driven transfer parameters for the marine foodweb and its integration in POSEIDON-R. Radioprotection 44(5), 2009, 741–746.
16. Myrberg K. Sensitivity tests of a two-layer hydrodynamic model of the Gulf of Finland with different atmospheric forcings. Geophysica 33(2), 1997, 69–98.
17. Bezhenar R., Kim K.O., Maderich V., de With G., Jung K.T. Multi-compartment kinetic-allometric (MCKA) model of radionuclide bioaccumulation in marine fish. Biogeosciences 18, 2021, 2591-2607.

Appendix A. Marine dispersion model POSEIDON-R

The box model POSEIDON-PC was initially developed for the simulation of routine discharges from nuclear installations located on the coast [1]. To describe the accumulation of radionuclides in marine organisms as the result of accidental releases, the BURN extinction of the POSEIDON model was developed [2]. In the model, the dynamical equations for organisms occupying different trophic levels are solved allowing the model to reproduce the time delay between the uptake of activity by biota and variations of radionuclide concentration in the seawater. In this model, the "target-tissue" approach was also introduced based on the assumption that each radionuclide is accumulating in a single specific organ. Finally, the benthic food web was added to the model [3] to describe the transfer of activity from contaminated bottom sediments to marine organisms through the food chain. Currently, there is a POSEIDON-R version of the model, which is integrated into the RODOS decision support system [4,5].

Dispersion of activity in water and sediments

In all modifications of the POSEIDON model, the water environment is considered as a system of boxes of different sizes, which can be subdivided into several vertical layers in the water column (Figure C.1). Each box contains a constant concentration of suspended sediments, which are continuously settling down to the lower water layers and then to the bottom. The model assumes that the fraction of radionuclide dissolved in the water (liquid phase) is in instantaneous equilibrium with the adsorbed on suspended or bottom sediments fraction (solid phase) with the constant ratio between their concentrations equal to the distribution coefficient K_d. The temporal variations of radionuclide concentration in each compartment are calculated taking into account the exchange of activity with adjacent compartments, suspended and bottom sediment, as well as radioactive sources and decay. The transfer of activity from one compartment to another depends on the concentration of radionuclide in the initial compartment and is governed by fluxes of water due to advection and diffusion processes on the faces between boxes. A simple three-layer model is considered for describing the migration of radioactivity in bottom sediments. The exchange of activity between the top bottom layer and near bottom water layer is described by diffusion and bioturbation processes. Only the diffusion process is considered between the top and middle sediment layers. In addition, the settling of sediments to the lower layers determines the continuous flux of activity in the bottom sediments directed downward. The equations for water column layers (Eqn. C.1), and upper (Eqn. C.2) and middle (Eqn. C.3) sediment layers read as follows:

$$\frac{\partial C_{0ik}}{\partial t} = \sum_m \sum_j \left[\frac{F_{jimk}}{V_{0ik}} C_{0jm} - \frac{F_{ijkm}}{V_{0jm}} C_{0ik} \right] + \gamma_{0ik} C_{0(i+1)k} - (\gamma_{1ik} + \lambda) C_{0ik} + \frac{L_{t,k}}{h_{1k}} \gamma_{2k} C_{1k} + Q_{sik}, \qquad \text{(A.1)}$$

$$\frac{\partial C_{1k}}{\partial t} = -(\gamma_{2k} + \gamma_{3k} + \lambda)C_{1k} + \frac{h_{1k}}{L_{t,k}}\gamma_{1k}C_{01k} + \frac{L_{m,k}}{L_{t,k}}\gamma_{4k}C_{2k}, \quad \text{(A.2)}$$

$$\frac{\partial C_{2k}}{\partial t} = -(\gamma_{4k} + \gamma_{5k} + \lambda)C_{2k} + \frac{L_{t,k}}{L_{m,k}}\gamma_{3k}C_{1k}. \quad \text{(A.3)}$$

where C_{0ik} is the spatially averaged concentration of radionuclide in the water column layer i of box k; i=1 corresponds to the near bottom water column layer; C_{1k} and C_{2k} are the averaged concentration of radionuclide in the upper and middle sediment layers of box k, respectively; λ is the radionuclide decay constant; F_{ijkm} is the water flux from layer i of box k to layer j of box m; V_{0ik} is the volume of layer i of box k; h_{ik} is the thickness of the water column layer i of box k; $L_{t,k}$ and $L_{m,k}$ are the thicknesses of the top and middle bottom sediment layers of box k, respectively; Q_{sik} is the source of the activity in layer i of box k; $\gamma_{0ik}\ldots\gamma_{5k}$ are the radionuclide transfer rates in the system water – suspended sediments – bottom sediments, t is the time.

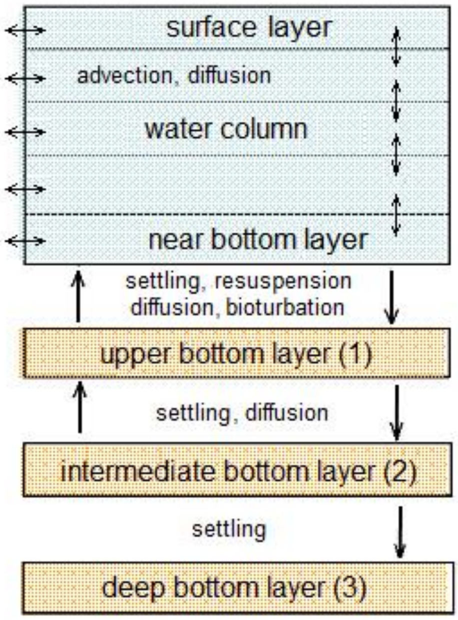


Fig. A.1. Vertical structure and radionuclide transfer processes in the compartment model POSEIDON-R.

Dynamic food web model

The recent biota model intercomparison [6] shows that dynamic biota models, which handle situations out from equilibrium, perform better than equilibrium models for the radioecological dose assessment after nuclear accidents. The biota model in POSEIDON-R is a dynamic food web model, where marine organisms are grouped into classes according to trophic level and species type (Fig. C.2). Radionuclides are also grouped into classes according to the fish tissue type in which they are preferentially accumulated (e.g., ^{137}Cs tends to accumulate in muscle). These simplifications allow for a limited number of standard input parameters. The scheme of transfer of radionuclides through the marine food web is shown in Fig. C.2. The different food chains exist in the pelagic zone and in the benthic zone. Pelagic organisms are grouped into a primary producer (phytoplankton) and consumers: zooplankton, forage (non-piscivorous) fish and piscivorous fish. The benthic food web includes three primary pathways for radionuclides: (i) transfer from water to macroalgae, then to grazing invertebrates; (ii) transfer through the vertical flux of detritus and zooplankton faces to detritus-feeding invertebrates; and (iii) transfer through contaminated bottom sediments to deposit-feeding invertebrates. External boxes in Fig. C.2 show the concentrations of radionuclides in the water and in the organic deposit, which is in instantaneous equilibrium with the upper layer of the bottom sediment, calculated by the above described POSEIDON-R model. In the benthic food chain, the radioactivity is transferred from the deposit feeding invertebrates to the demersal fish, and to the bottom predators. The components of this system are crustaceans (e.g detritus-feeders), molluscs (filter-feeders) and coastal predators feeding in the whole water column of shallow coastal waters. Along with the food web, all organisms take radionuclides directly from water.

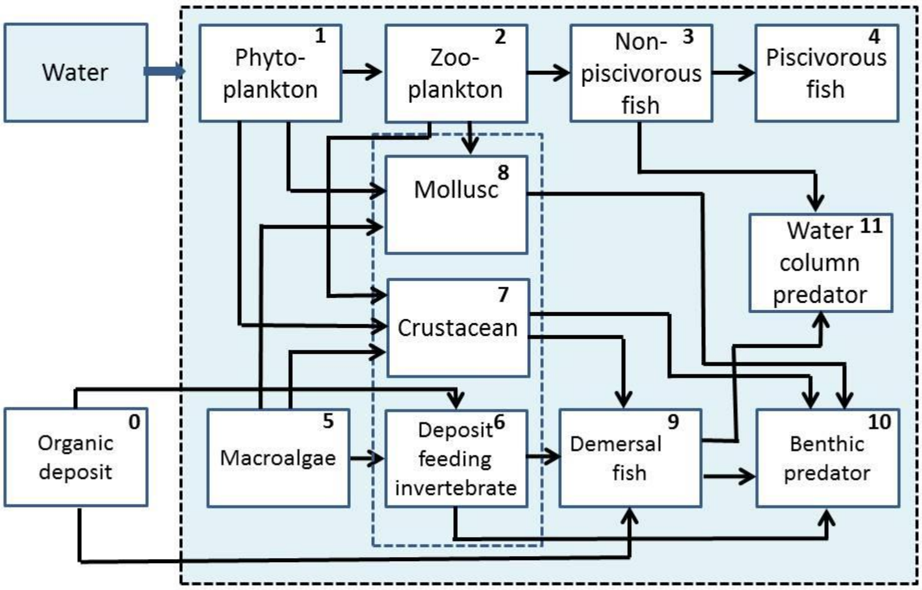


Fig. A.2. Radionuclide transfer from the water and bottom sediment boxes to marine organisms [7]. The radionuclide transfers among marine food web compartments are given for 11 types of marine organisms.

Due to the rapid uptake from water and the short retention time of radioactivity, the concentration of radionuclides in phytoplankton is calculated using the Biological Concentration Factor (*BCF*) approach [7]. For the macroalgae, a dynamic model is used to describe radionuclide concentrations due to the longer retention times

$$\frac{dC_{ma}}{dt} = \left(CF_{ma}C_w - C_{ma}\right)\frac{\ln 2}{T_{0.5,ma}}, \tag{A.4}$$

where C_w and C_{ma} are the radionuclide concentration in the water and macroalgae, respectively, CF_{ma} is the corresponding BCF, $T_{0.5,ma}$ is the biological half-life of the radionuclide in the macroalgae, t is the time. The concentration of a given radionuclide in other considered marine organisms is described by the following differential equation:

$$\frac{dC_i}{dt} = a_i K_{f,i} C_{f,i} + b_i K_{w,i} C_w - \frac{\ln 2}{T_{0.5,i}} C_i , \tag{A.5}$$

where C_i and $C_{f,i}$ are the radionuclide concentration in the i-th marine organisms and their food, respectively, a_i is the food extraction coefficient (assimilation rate), b_i is the water extraction coefficient, $K_{f,i}$ is the food uptake rate, $K_{w,i}$ is the water uptake rate and $T_{0.5,i}$ is the biological half-life of the radionuclide in the organism.

The activity concentration in the food of a predator C_f is expressed by the following equation, summing for a total of n prey types,

$$C_f = \sum_{i=0}^{n} C_{prey,i} \, P_{prey,i} \, \frac{drw_{pred}}{drw_{prey,i}}, \tag{A.6}$$

where $C_{prey,i}$ is the activity concentration in prey of type i, $P_{prey,i}$ is preference for prey of type i, drw_{pred} is the dry weight fraction of predator, and drw_{prey} is the dry weight fraction of prey of type i. The index "0" corresponds to the organic deposit in bottom sediments. Values of the model parameters are discussed in [3] and are given in Tables C.1-C.3. The generic parameters of the model were calibrated for different aquatic environments [3,8,9]. The sensitivity of parameters was investigated in [3].

It is well known that the uptake of caesium and strontium decreases with increasing salinity due to the increase in concentration of competing ions of potassium and calcium, respectively. For caesium it was taken into account when introducing the salinity-dependent correction factor F_K for phytoplankton and macroalgae because caesium enters the food web primarily through the lowest trophic level whereas the contribution of direct uptake from water is minor [10]. Instead of using fixed concentration factors, the BCF for ^{137}Cs is related to potassium concentration via the electrochemical competition for which the parameters are based on laboratory experiments with marine plants. For strontium, the direct gill uptake is more important due to the lack of bioaccumulation through the food web. The gill extraction coefficient b_i is based on empirical correlations derived from measured equilibrium levels in the seas.

Table A.1. Parameters of dynamical food chain model.

Parameters

i	Organism	drw	K_1 (d^{-1})	a	K_w ($m^3(kg\ d)^{-1}$)	b	$T_{0.5}$ (d)
1	Phytoplankton	0.1	-	-	-	-	-
2	Zooplankton	0.1	1.0	0.2	1.5	0.001	5
3	Non-piscivorous fish	0.25	0.03	0.5	0.1	0.001	Table 3
4	Piscivorous fish	0.3	0.007	0.7	0.075	0.001	Table 3
5	Macroalgae	0.1	-	-	0.6	0.001	60
6	Deposit feeding invertebrates	0.1	0.02	0.3	0.1	0.001	15
7	Molluscs	0.1	0.06	0.5	0.15	0.001	50
8	Crustaceans	0.1	0.015	0.5	0.1	0.001	100
9	Demersal fish	0.25	0.007	0.5	0.05	0.001	Table 3
10	Bottom predator	0.3	0.007	0.7	0.05	0.001	Table 3
11	Coastal predator	0.3	0.007	0.7	0.075	0.001	Table 3

Table A.2. Food preference for prey of type *i*, for prey of type *j*.

Predator Prey	2	3	4	6	7	8	9	10	11
0				0.5			0.1		
1	1.0				0.6	0.1			
2		1.0			0.2	0.8			
3			1.0						0.2
5				0.5	0.2	0.1			
6							0.7	0.3	0.25
7							0.1	0.2	0.1
8							0.1	0.2	0.2
9								0.3	0.25

According to the review of radiological data [11, 12], every radionuclide is mainly accumulated in a specific tissue (target tissue). It can be assumed that the target tissue controls the overall elimination rate of the nuclide ($T_{0.5}$) in the organism. The radioactivity in the food for the predator is then the activity concentration in the target tissue diluted by the remaining body mass of the prey fish, calculated by multiplying the predicted level in the target tissue by its weight fraction. To calculate the concentration in the edible part of fish (flesh) from the calculated levels in the target tissues, a target tissue modifier (TTM) is introduced. This is also based on tissue distribution information as reported by [11, 12]. Values of described parameters for the dynamic food-chain model are listed in Table C.3.

Table A.3. Parameters for the fish in dynamical food chain model.

Target tissue	Bone	Flesh	Organs	Stomach
Weight fraction *f*	0.12	0.80	0.05	0.03
Target tissue modifier (TTM)	0.5	1	0.5	0.5
Biological half-life of non-piscivorous fish (d)	500	75	20	3
Biological half-life of piscivorous fish (d)	1000	150	40	5
Biological half-life of demersal fish (d)	500	75	20	3
Biological half-life of bottom predator fish (d)	1000	150	40	5
Biological half-life of coastal predator fish (d)	1000	150	40	5

Sources of activity

The POSEIDON-R model can deal with four types of routine and accidental radioactive releases:

(i) atmospheric deposition directly on the sea surface;

(ii) runoff of land deposited radionuclide;

(iii) point sources associated with routine releases of nuclear facilities, located either directly at the coast or inland at river systems;

(iv) point sources associated with accidental releases located in any box of the model domain.

For coastal discharges, it is useful to provide a more detailed description in the area close to the release point. For this purpose, the additional "coastal" boxes are nested into the large ("regional") boxes in the box system of the considered region. There are some assumptions and restrictions to the approach. These are as follows: (i) a coastal box has one vertical layer for the water column; (ii) a coastal box interacts with the surface layer of the surrounding regional box only, the depth of a coastal box is therefore less or equal to that of the surface layer of regional box; (iii) the exchange fluxes with the adjacent regional box are equal in both directions, i.e. only lateral diffusion is taken in account; (iv) only one coastal box can be added per regional box; (v) a coastal box contains at most one source of radioactivity.

POSEIDON-R has also the possibility to deal with off-shore point releases (e.g. for evaluation of the impact of sunken vessels, nuclear submarines, and off-shore waste dumping). In that case, it is possible to use a so-called "local" box. The off-site local boxes have the following features: (i) a local box can be placed at any point in the surrounding regional box at any depth; (ii) the volume and thickness of the local box are calculated as proportional parts of the outer regional box; (iii) as in the case of the coastal box, the exchange flows between the local box and the surrounding regional box are assumed to be equal in both directions.

Numerical solution

The problem is described by a set of ordinary differential equations, which may be written in a vector-matrix notation as:

$$\frac{d\mathbf{C}}{dt} = A\mathbf{C} + \mathbf{Q}_{re}\,, \tag{A.7}$$

where $\mathbf{C}$ is the concentration vector; A is the coefficient matrix that includes water fluxes between boxes, parameters of the food-chain model, etc; $\mathbf{Q}_{re}$ is the vector for the release term. Step-like variations of the release in time are assumed, and the implicit Matrix Exponential Method [13] is used to solve a set of equations (26). Description of the method is given in [8].

Dose calculation

The POSEIDON-R model includes dose module to assess individual and collective doses to the population due to the regular and accidental releases of radionuclides. The exposure pathways that are considered in the model include: internal exposure through ingestion of seafood and inhalation via sea spray and external exposure through swimming, boating and beach occupancy. However, the dose from seafood consumption dominates, that was shown in [14], where calculated doses from other pathways of irradiation were two orders of magnitude less than dose from seafood consumption.

The annual dose from consumption of marine products $E_{marine,k}$ ($Sv \cdot yr^{-1}$) from the ingestion of 8 categories (*f*) of marine products (piscivorous and non-piscivorous fish, demersal, bottom predator, coastal predator, crustaceans, molluscs and macro-algae) for a given box *k* is described as follows:

$$E_{marine,k} = DC_{ing}\left(\sum\nolimits_{f=1}^{8} C_{f,k} CR_{f,k}\right), \tag{A.8}$$

where $C_{f,k}$ (Bq kg^{-1}) is the activity concentration of the radionuclide in the marine product of type *f*, $CR_{f,k}$ is the marine food intake rate (kg y^{-1}), DC_{ing} (Sv Bq^{-1}) is the dose coefficient due to ingestion from marine products given by [15].

References

1. Lepicard S., Raffestin D., Rancillac F. POSEIDON: a dispersion computer code for assessing radiological impacts in European seawater environment. Radiation Protection Dosimetry 75, 1998, 79–83.
2. Heling R., Koziy L., Bulgakov V. On the dynamical uptake model developed for the uptake of radionuclides in marine organisms for the POSEIDON-R model system. Radioprotection 37(C1), 2002, 833–838.
3. Bezhenar R., Jung K.T., Maderich V., Willemsen S., de With G., Qiao F. Transfer of radiocaesium from contaminated bottom sediments to marine organisms through benthic food chain in post-Fukushima and post-Chernobyl periods. Biogeosciences 13, 2016, 3021–3034.
4. Lepicard S., Heling R., Maderich V. POSEIDON-R/RODOS models for radiological assessment of marine environment after accidental releases: application to coastal areas of the Baltic, Black and North Seas. Journal of Environmental Radioactivity 72, 2004, 153–161.
5. Bezhenar R., Heling R., Ievdin I., Iosjpe M., Maderich V., Willemsen S., de With G., Dvorzhak A. Integration of marine food chain model POSEIDON in JRODOS and testing versus Fukushima data. Radioprotection 51(HS2), 2016, S137–S139.
6. Vives i Batlle J., Beresford N.A., Beaugelin-Seiller K., Bezhenar R., Brown J., Cheng J.-J., Cujic M., Dragovic S., Duffa C., Fievet B., Hosseini A., Jung K.T., Kamboj S., Keum D.-K., Kryshev A., LePoire D., Maderich V., Min B.-I., Perianez R., Sazykina T., Suh K.-S., Yu C., Wang C., Heling R. Inter-comparison of dynamic models for radionuclide transfer to marine biota in a Fukushima accident scenario. Journal of Environmental Radioactivity 153, 2016, 31-50.
7. IAEA, International Atomic Energy Agency, Sediment distribution coefficients and concentration factors for biota in the marine environment, Technical Report Series No 422, IAEA, Vienna, Austria, 2004.
8. Maderich V., Bezhenar R., Tateda Y., Aoyama M., Tsumune D., Jung K.T., de With G. The POSEIDON-R compartment model for the prediction of transport and fate of radionuclides in the marine environment. MethodsX 5, 2018, 1251-1266.
9. Bezhenar R., Maderich V., Schirone A., Conte F., Martazinova V. Transport and fate of ^{137}Cs in the Mediterranean and Black Seas system during 1945-2020 period: A modelling study. Journal of Environmental Radioactivity 208-209, 2019, 106023.
10. Heling R., Bezhenar R. Modification of the dynamic radionuclide uptake model BURN by salinity driven transfer parameters for the marine foodweb and its integration in POSEIDON-R. Radioprotection 44(5), 2009, 741-746.
11. Coughtrey P.J., Thorne M.C. Radionuclide distribution and transport in terrestrial and aquatic ecosystems: A critical review of data. Vol. 2. A.A. Balkema. Rotterdam, 1983.
12. Yankovich T., Beresford N., Wood M., Aono T., Anderson P., Barnett C.L., Bennett P., Brown J.E., Fesenko S., Fesenko J., Hosseini A., Howard B.J., Johansen P., Phaneuf M.M., Tagami K., Takata H., Twining J.R., Uchida S. Whole-body to tissue-specific concentration ratios for use in biota dose assessments for animals. Radiation Environmental Biophysics 49, 2010, 549-565.
13. Patten B.C. (Ed.) Systems analysis and simulation in ecology. Academic Press. Vol. 1, 1971, 81-82.
14. Maderich V., Bezhenar R., Kovalets I., Khalchenkov O., Brovchenko I. Long-term contamination of the Arabian Gulf as a result of hypothetical Nuclear Power Plant accidents. Journal of Marine Science and Engineering. 11(2), 2023, 331.
15. ICRP (International Commission on Radiological Protection). Compendium of Dose Coefficients based on ICRP Publication 60. Elsevier: Amsterdam, The Netherlands, 2012. Volume 41.