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Using a chain of models to predict health and environmental impacts in Norway from a hypothetical nuclear accident at the Sellafield site

Keywords: nuclear accidents; radioactive fallout; multi-compartment modelling; decision support systems; model uncertainties

1 Introduction

When a nuclear accident occurs, decision makers in the affected country/countries would need to act promptly to protect people, the environment and societal interests from harmful impacts of radioactive fallout. The decisions are usually based on a combination of model prognoses, measurements, and expert judgements within in an emergency preparedness framework. Following the Chernobyl accident, several decision support systems (DSS) were developed in Europe to provide expert support to decision makers should a nuclear accident occur. The DSS have been expanded over the years to include an increasing number of modules with specific functions for various ecosystems. The early atmospheric dispersion and deposition models had subsequent food chain contamination modules and dose calculations for humans, while some DSS now include modules for urban, forest and aquatic environments as well as countermeasures in various ecosystems, e.g., ERMIN (European Model for Inhabited Areas) and AgriCP (Agricultural Countermeasure Programme) (Raskob et al., 2010). In the event of radioactive fallout, there would be a subsequent transfer of radionuclides between different environmental compartments due to e.g., erosion, run-off, forestry and agricultural practice, and from rivers via estuaries to the marine ecosystems. This environmental transfer of radionuclides and the radiation exposures of humans and biota should be assessed to determine whether protective actions and/or remediation would be necessary. Thus, predictive models would need to cover the atmospheric, terrestrial, freshwater, and marine ecosystems, the connections between these in terms of radionuclide fluxes and the various exposure pathways to both humans and biota. This could be achieved by coupling different ecosystem models either as stand-alone models used in a chain or as linked modules within a nuclear decision support system like ARGOS (Accident Reporting and Guiding Operational System) (Hoe et al., 2009) or RODOS (Real Time On-line Decision Support) (Landman et al., 2014). The results obtained could be further used in optimising countermeasure strategies and performing cost-benefit analyses for remediation.

As DSS are necessarily generic in character, regional or country specific predictions can be improved by adapting model configurations and through bespoke parametrization. However, this could be an elaborate and time-consuming process. Alternatively, already existing national or regional models may be applied instead, either alone or in combination with certain modules of DSS.

As pointed out by Salbu (2016) integrated models for impact and risk assessments are complex systems and the predictive model output is confounded by uncertainties that stem from various sources. She categorized the sources of uncertainties accordingly:

(1) *Input uncertainty* (experimental uncertainties);

(2) *Interpolation and extrapolation uncertainty* (due to insufficient or lacking input data);

(3) *Parameter uncertainty and variability* (inherent variability and parameter values that cannot be experimentally controlled);

(4) *Algorithmic uncertainty* (numerical errors and approximations in the mathematical model); and

(5) *Structural uncertainty*.

The latter, also referred to as conceptual or model uncertainty, is assigned to model bias or discrepancy from real life due to lack of knowledge or to deliberate omission of relevant mechanisms, processes or phenomena.

Reducing uncertainties in predictive models is one of the key aims of the Norwegian Centre of Excellence for Environmental Radioactivity (CERAD CoE)[1] to underpin the core objective of providing the scientific basis for impact and risk assessments in management of radiation risks, both for past events and for potential future events. CERAD initiated this case study to assess possible human and environmental impacts in Norway from a hypothetical accident at the Sellafield nuclear reprocessing plant using a range of models. The study had two distinct goals:

(a) To investigate if selected regional/local models and DSS could be linked in a meaningful way to predict impacts in relevant ecosystems from nuclear accidents and to test the linked models for a hypothetical accident with radioactive fallout in Norway.

(b) To identify key factors contributing to the uncertainties in predictive model outputs, and to decide, based on sensitivity analyses as well as expert judgement, which key factors to focus on in further research.

2 Methods and models

2.1 Scope

A combination of the ARGOS DSS with modules and other predictive models were used to assess the impacts for several ecosystems of a hypothetical accident at the Sellafield nuclear reprocessing plant. Table 1 presents the names of the DSS, modules and models used and their application. A generic description of each model is given in Appendix 1.

Table 1: Overview of the DSS, modules and models used to assess the impacts in Norway from a hypothetical accident scenario at the Sellafield site.

[1] https://www.nmbu.no/en/services/centers/cerad/node/24335

Short name	Full name	Application
SNAP	Severe Nuclear Accident Programme	Atmosperic dispersion and deposition of radionuclides
ARGOS	Accident Reporting and Guiding Operational System	Decision support system for nuclear and radiological accidents, external and inhalation doses to humans
FDMT	Food Chain and Dose Module - Terrestrial	Concentrations of radionuclides in food and feed, ingestion doses to humans
STRATOS	N/A	Concentrations of radionuclides in wild foodstuffs and rough grazing animals` products
INCA-RAD	INtegrated CAtchment model for RADionuclides	Transport and retention of radionuclides in freshwater systems
NRPA marine box model	N/A	Dispersion of radionuclides in marine waters, concentrations in and dose rates to marine biota
ROMS	Regional Ocean Model System	Dynamic dispersion of radionuclides in marine waters
ERICA Tool	N/A	Concentrations of radionuclides in biota, dose rates to biota, environmental risk assessment

Figure 1 shows how the models were linked. Based on an estimated inventory of ^{137}Cs, an accident source term and a real meteorological scenario, the SNAP (Severe Nuclear Accident Programme) model was used to simulate atmospheric dispersion and transport of radionuclides to Norway. Total deposition maps of ^{137}Cs were produced for marine, freshwater and terrestrial areas. ARGOS DSS estimated external and inhalation doses to humans while the contamination of agricultural produce and ingestion doses to humans were calculated using its module FDMT (Terrestrial Food Chain and Dose Module). For wild foodstuff and rough grazing animals, the STRATOS model was used to predict areas where food intervention levels might be exceeded. INCA-RAD (INtegrated CAtchment model for RADionuclides) modelled contamination in freshwater bodies from direct deposition and catchment run-off. Two models were used for the marine ecosystem: the NRPA marine box model and the dynamic Regional Ocean Model System (ROMS). The ERICA Tool was used to calculate concentrations in and dose rates to aquatic species.

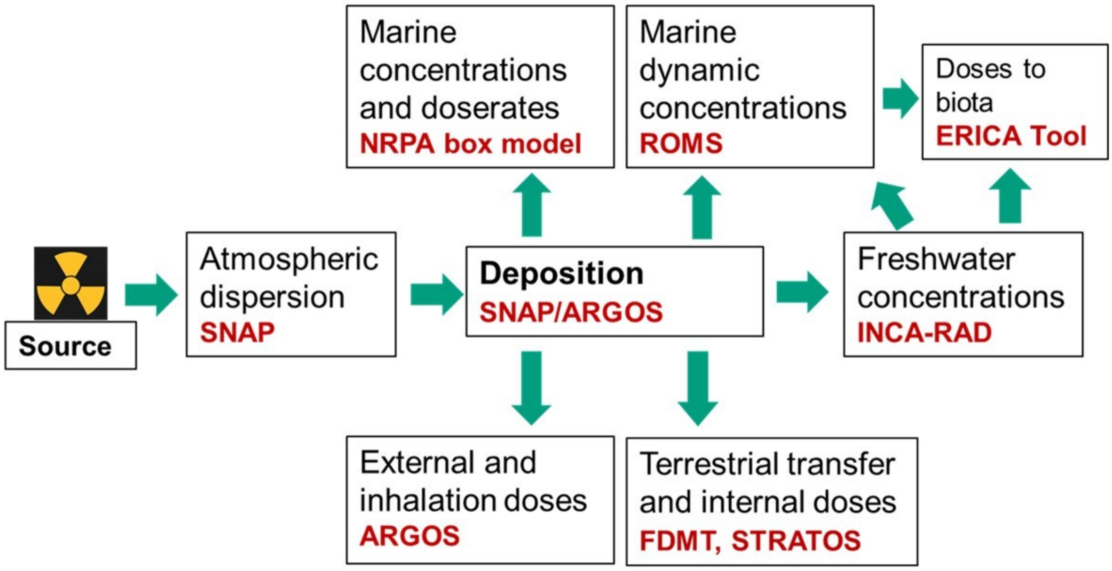

Figure 1: Linking the different DSS, modules and models in a chain to assess impacts.

2.2 Hypothetical accident scenario at the Sellafield nuclear reprocessing plant

2.2.1 Inventory and source term

The Sellafield nuclear site, situated on the coast of the Irish Sea in Cumbria, England, is currently owned by the Nuclear Decommissioning Authority (NDA) and operated by Sellafield Ltd. One of the main activities has been reprocessing of spent nuclear fuel during which uranium and plutonium were recovered for producing mixed oxide fuel. The reprocessing resulted in large volumes of highly active liquor (HAL) as a waste by-product. This HAL is temporarily stored in 21 specially designed tanks (Highly Active Storage Tanks - HASTs) until the waste can be vitrified, i.e., blended as part of a solid glass matrix that is easier to handle and safer for long-term storage. The HAL produces heat and needs continuous cooling to avoid the liquid to evaporate. The inventory in the HASTs was calculated to be approximately 1.9 - $3.0 \cdot 10^{18}$ Bq for ^{137}Cs as of April 2014 (see Appendix 2 for calculations).

The HASTs constitute a potential threat for contaminating the environment either due to failure of the cooling system, a natural disaster (e.g., earthquake) or a malevolent act (e.g., bomb, plane crash) that would destroy the integrity of the tanks. Indeed, in 1957 in Kysthym, Soviet Union, the cooling system of a similar waste storage tank from the Mayak Production Facility failed and evaporation of the cooling liquid resulted in a chemical explosion. It resulted in a loss of the tank`s integrity and vast amounts of radioactive material (in the order of PBq) were emitted to the atmosphere, contaminating a large area (Norwegian Radiation Protection Authority, 2007). Furthermore, historic discharges from Sellafield to the marine environment have contaminated the Norwegian coastline for many years. Combined with concerns over nuclear safety at the site, Sellafield Ltd has been of concern to Norwegian authorities, NGOs and the public for a long time (Liland et al., 2017).

We have deliberately not estimated the probability of a possible accident release scenario, be it the loss of cooling, a natural disaster or a malevolent act. We simply assume a scenario with a loss of the HAL tanks' integrity and that 1 % of the estimated ^{137}Cs in the tanks ($3.0 \cdot 10^{16}$ Bq) is released to the air as aerosols at a height where it can be mixed with the atmosphere and then transported to Norway by atmospheric dispersion. 1% is not an unreasonable estimate according to an earlier study by Ytre-Eide et al. (2009): "The UK Parliamentary Office of Science and Technology observe that earlier impact assessments … have used quite different source terms; from 0.01 % of one HAST inventory to over 10 % of all the HASTs contents".

The tanks also contain other long-lived radionuclides such as ^{90}Sr, ^{241}Am and ^{244}Cm, but this study is restricted to only include ^{137}Cs. There are no short-lived radionuclides present in the HAL.

2.2.2 Meteorological scenario

The selected meteorological scenario was based on real weather data recorded in October 2008 (Ytre-Eide et al., 2009) and the hypothetical accident was assumed to have occurred on 19 October 2008 at 13:00 UTC. The weather situation at that time was dominated by a low pressure system located southeast of Iceland. This caused southwest winds towards Scandinavia across the North Sea with extensive precipitation in the south-western part of Norway. The precipitation in Bergen, the largest city on the west coast, was 20, 15 and 30 mm of rainfall for 19, 20 and 21 October, respectively. The Norwegian Meteorological Institute (MET Norway) considered this weather to be representative for this season and geographical region. Under such circumstances a potential radioactive release could reach the Norwegian coast within a period of nine hours from the release time (Ytre-Eide et al., 2009).

2.3 Linking the models

2.3.1 Atmospheric dispersion and deposition, doses to humans

Atmospheric dispersion from the site of accident was modelled by SNAP based on the inventory and source term as described in chapter 2.2 and Appendix 2 and using meteorological data from the HIRLAM model (see Appendix A.1.1 for details). The emissions were considered to have a duration of 5 minutes, consistent with e.g., an explosion. The emission height given as model input to the SNAP model refers to the height where model particles enter the atmosphere without being further disturbed by the source. This is not actually the stack-height of the source, but rather a combination of stack-height and plume rise. Our scenario used an emission height between 0 and 800 m. The model particles carrying ^{137}Cs were assumed to be aerosols with density 2.3 g cm^{-3}.

The output from SNAP (Bq·h m^{-3}, Bq m^{-3}, Bq m^{-2}) was converted from a rotated earth projection used by the meteorological model to two projections: WGS84 and UTM zone 33N, the former extending further to the north, both in approx. 10 km grid resolution. This was used as input in ARGOS DSS for wet, dry and total deposition (Bq m^{-2}) and precipitation (kg m^{-2}). Then, ARGOS DSS could directly calculate external and inhalation doses to humans (mSv over a specified time period) from this input.

2.3.2 Food chain modelling

The ARGOS data on terrestrial deposition (Bq m^{-2}) and air concentration (Bq m^{-3}) was used as input in FDMT and STRATOS to predict concentrations (Bq kg^{-1}) in agricultural produce, and wild foodstuffs and rough grazing animals, respectively. FDMT combined the modelled concentrations in agricultural produce with dietary statistics to calculate internal doses to humans of various age groups (mSv over a specified time period). STRATOS combined the deposition data (Bq m^{-2}) with aggregated transfer factors (kg m^{-2}) to predict concentrations (Bq kg^{-1}) in rough grazing animals and wild foodstuffs. This was further linked to grazing and hunting statistics and extent of wild foodstuffs to predict how much of the annual yield or production (%) would exceed the food intervention levels for each foodstuff.

2.3.3 Freshwater modelling

The SNAP raster files for terrestrial deposition (Bq m^{-2}) was used as input in the INCA-RAD model. The deposition on lakes, streams and catchment areas were included and the transport of ^{137}Cs from soils and run-off to river, riverine transport and sediment dynamics was modelled for the investigated site in western Norway based on hydrology and land use. The outputs from INCA-RAD were flow ($m^3 s^{-1}$), concentration in water (Bq m^{-3}) and concentration in sediments (Bq kg^{-1}) in lakes, rivers and tributaries in the Vikedal area, in daily time steps.

The data on concentrations in water (Bq m^{-3}) and sediments (Bq kg^{-1}) were used as input in the ERICA Tool version 1.0 to calculate concentrations (Bq kg^{-1}) and dose rates (μGy h^{-1}) to freshwater species.

The daily outflow of ^{137}Cs from the river Vikedal to the estuary in Vindafjorden was also calculated as well as the monthly outflow of ^{137}Cs from 21 other rivers along the Norwegian coast. These were used as additional point sources of ^{137}Cs for marine modelling with ROMS (see below).

2.3.2 Marine modelling

The SNAP raster files for deposition onto sea surface (Bq m^{-2}) were converted into total surface deposition for each affected box (Bq per box) in the NRPA marine box model. This input was then used to calculate concentrations in seawater (Bq m^{-3}) at different depths, concentrations in sediments and marine organisms (Bq kg^{-1}), and corresponding dose rates to biota (μGy h^{-1}) over time.

In the dynamic ocean model ROMS, the SNAP raster files for sea surface deposition (Bq m^{-2}) were imported directly and combined with the daily river discharge data from INCA-RAD (Bq d^{-1}) as additional point sources to predict concentrations in water (Bq m^{-3}) at different depths in daily time steps. The water concentrations were used as input in the ERICA Tool to calculate concentrations (Bq kg^{-1}) and dose rates (μGy h^{-1}) to marine species.

3 Modelling results

Our study showed that all the models/modules/DSS in Figure 1 could be linked in a meaningful way to predict impacts in various ecosystems in Norway from radioactive fallout. The normal assumptions and default values in the models described in Appendix 1 were used in the predictive modelling of impacts from the hypothetical accident at Sellafield Ltd. unless otherwise specified in the text below. It should be noted that all the modelling results below are given as additional ^{137}Cs due to this hypothetical accident; ^{137}Cs already present in the environment due to global fallout from atmospheric testing of nuclear weapons and fallout from the Chernobyl accident is not included.

3.1 Simulated atmospheric dispersion and fallout in Norway - SNAP

The atmospheric dispersion from the accident site and the subsequent fallout in Norway was simulated with the model SNAP (Figure 2, left). The model particles carrying ^{137}Cs were assumed to be solely aerosols of density 2.3 g cm^{-3}. We know, however, that releases from nuclear accidents could entail particles of various sizes and densities. This will influence on the simulated transport and deposition. The selection of an aerosol size and 2.3 g cm^{-3} density is considered to be a conservative approach, appropriate for a worst case scenario, since the lifetime for such particles in the atmosphere is relatively long, compared to other forms e.g., particles of a larger size (Klein et al., 2016).

The largest fallout for this hypothetical accident at Sellafield will occur on the western coast of Norway, with large areas predicted to be contaminated with ^{137}Cs levels above 100 kBq m^{-2}. The simulated fallout levels are comparable to the most contaminated areas in Norway after the Chernobyl accident (Figure 3).

Of the most heavily contaminated areas in the hypothetical accident, the county of Rogaland (see Figure 2, right) was selected for further impact assessments for several reasons. It is the fourth most populated county in Norway with significant agricultural production, fisheries, aquaculture and environmentally important areas. A radioactive fallout in Rogaland would have substantial negative impacts on inhabitants, environment, agriculture, aquaculture, fisheries, recreation and tourism. In addition, there is a general lack of radiological data and impact assessments for this region since it was hardly affected by the Chernobyl accident. The River Vikedal area in Vindafjord municipality, Rogaland was selected to specifically study effects on the freshwater system since generic environmental monitoring data covering several decades are available (Sandlund et al., 2010).

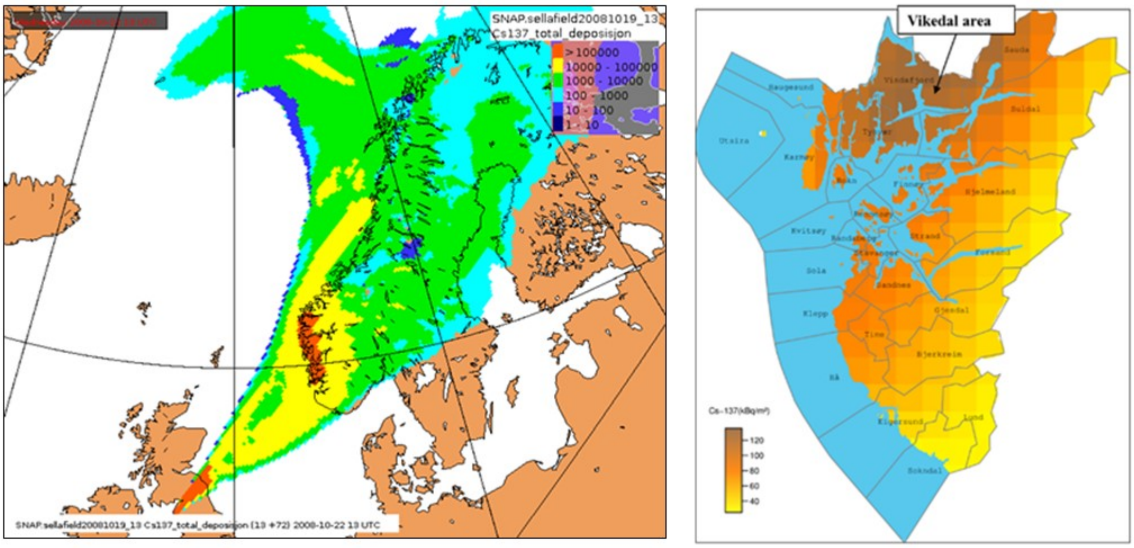

Figure 2: Atmospheric dispersion and deposition of ^{137}Cs in Bq m^{-2} during 24 h over Norwegian territories after a hypothetical accident at the Sellafield HASTs using SNAP and presented in the visualisation programme DIANA (left). A detailed map over the deposition (kBq m^{-2}) in Rogaland County produced in GRASS GIS is shown on the right together with the location of the Vikedal area (Liland et al., 2017).

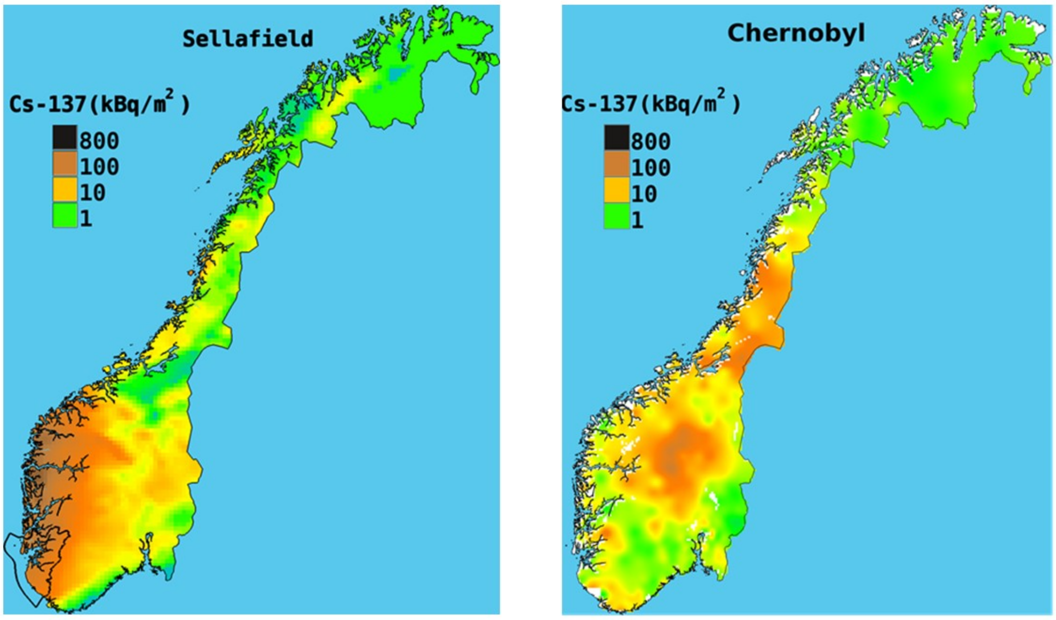

Figure 3: Compared fallout from the hypothetical accident at Sellafield (left) (Liland et al., 2017) and the Chernobyl accident in 1986 (right) (Backe et al., 1986). Rogaland County is outlined in the south-western part of Norway on the left figure.

3.2 External and inhalation doses to humans - ARGOS DSS

The total effective dose to adult humans from inhalation and external exposure to ^{137}Cs after one week, one month and one year was calculated in ARGOS and is presented in Figure 4. The highest average outdoor dose modelled in Norway was 2.7 mSv over the first year. This is a conservative estimate calculated on the basis that people stay outdoors all the time. In real life, people would be less exposed as buildings will provide substantial shielding to external doses (Komperød et al., 2015). According to the Nordic guidelines and recommendations (2014) on protective measures in a nuclear or radiological accident, early protective actions such as evacuation, sheltering and partial sheltering are only justified if the projected dose is above 20 mSv the first week, above 10 mSv over two days and 1-10 mSv in two days, respectively. Countermeasures to reduce external and inhalation doses are thus not justified in this scenario and this exposure pathway was not investigated further.

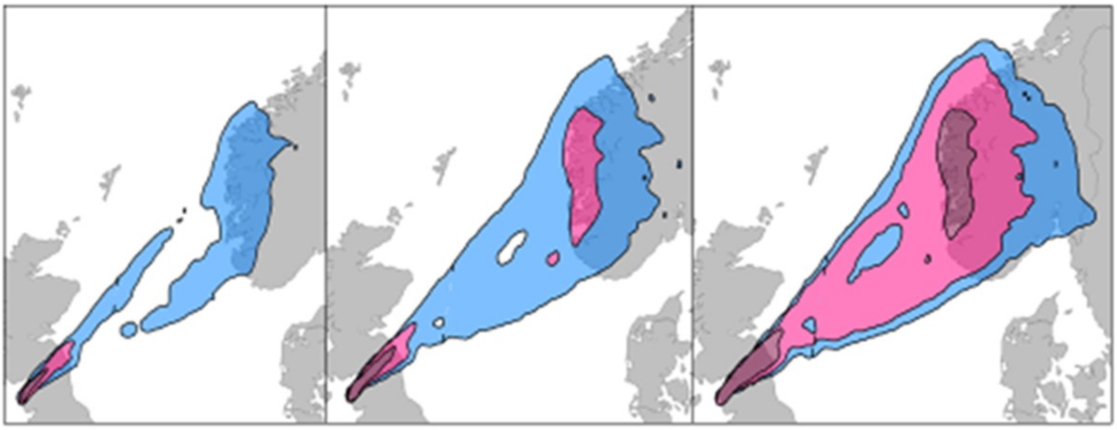

Figure 4: Total effective dose from inhalation and external exposures of ^{137}Cs after one week (left), one month (middle) and one year (right) calculated by ARGOS. Blue: < 0.01 mSv, pink: 0.01 - 0.1 mSv, Purple: >0.1 mSv.

3.3 Food chain modelling

3.3.1 Agricultural produce - FDMT

The model was run for Vindafjord municipality where the predicted deposition was 130 kBq m^{-2} of ^{137}Cs for our hypothetical scenario. The default values in FDMT were used (Müller et al, 2004) (see Appendix A.1.3 for details). FDMT generally assumes all fodder to be locally produced, including feed concentrates. Pigs, poultry and cattle (bulls) are not presumed to be on pasture, so contamination of products from such animals is mainly due to ^{137}Cs levels in the concentrates. Note that concentrates in Norway are usually not locally produced, so it can be debated whether it is justified to include model results for such products. We have, however, kept them based on an argument of being "conservative".

Since agricultural impacts vary considerably with deposition time of the year, we chose to run the scenario with three deposition dates: 20 May, 1 August and 20 October. Figure 5 shows the highest estimated concentrations with FDMT for the three deposition dates. With a deposition on 20 May, all foodstuffs except potatoes, beef (cattle) and eggs are predicted to exceed the Norwegian food intervention levels of 370 Bq kg^{-1} for milk and infant foods and 600 Bq kg^{-1} for all other basic foodstuffs. For a deposition on 1 August, all foodstuffs except pork and eggs are predicted to exceed the limits. For a deposition on 20 October, on the other hand, only root vegetables, cow`s milk, goat`s milk, beef (cows) and lamb meat would exceed the limits.

The time development of concentrations in cow's milk and lamb meat is shown in Figure 6. The predicted concentrations are significantly higher in lamb meat than in cow`s milk and the concentrations remain well above the food intervention level in meat even for the second year. The second peak observed for the red and blue lines is due to feeding with hay contaminated by the event, then cut and stored for later use. Without any mitigating actions the concentration in lamb meat from Vindafjord is predicted to be > 600 Bq kg^{-1} for 10–20 years due to extensive grazing.

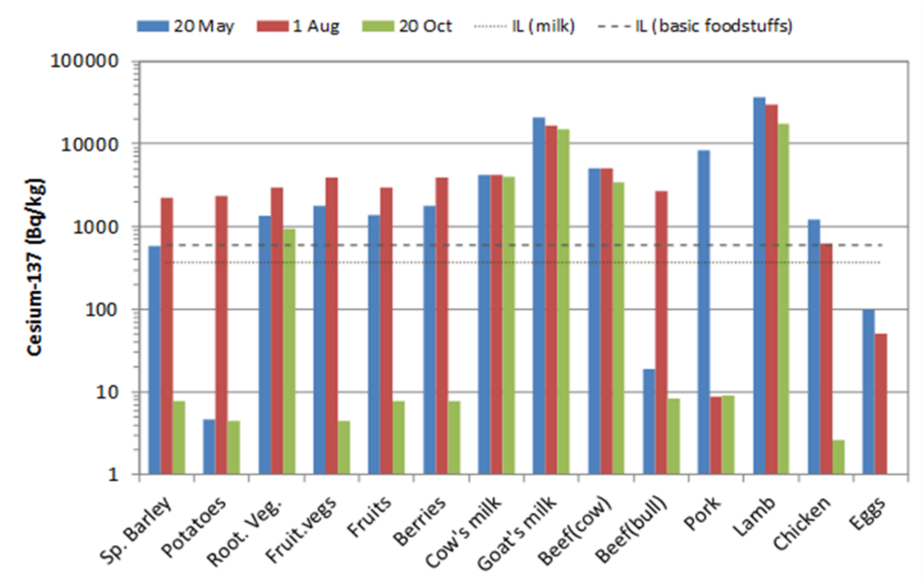

Figure 5: Highest estimated concentrations of ^{137}Cs (Bq kg^{-1}) in various foodstuffs predicted for the five year period using FDMT for three deposition dates: 20 May (blue), 1 August (red) and 20 October (green). The food intervention level for milk, IL (milk), and for other basic foodstuffs, IL (basic foodstuffs), are shown with dotted lines.

If we assume that all consumed food is locally produced, the internal doses from a contaminated diet in Vindafjord municipality would be up to 9.1 mSv for adults and 3.8 -7.0

mSv for children in Rogaland over 5 years if no agricultural countermeasures were implemented (Table 2). The largest part of the dose is received during the first year. It thus exceeds the general Norwegian recommendation to keep total radiation doses to the public from radioactive contamination below 1 mSv/y and also the Nordic guidelines and recommendations (2014) which state that the aim is to keep ingestion doses below 1 mSv/y the first year after an accident.

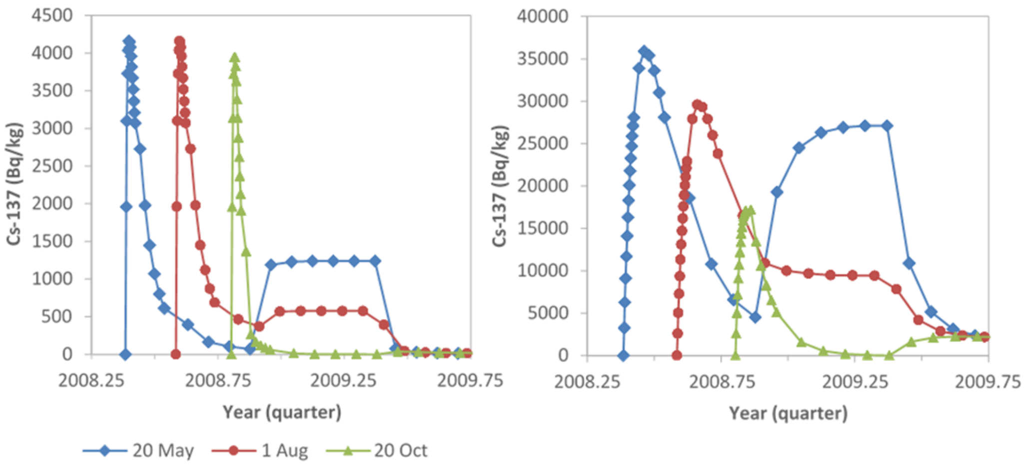

Figure 6: Time development of concentrations (Bq kg^{-1}) in cow`s milk (left) and lamb meat (right) for Vindafjord municipality with a deposition of 130 kBq/m^2 of ^{137}Cs on three different dates – 20 May (blue), 1 August (red) and 20 October (green). Please note the different scales on the y-axes for the two figures.

Table 2: Ingestion doses from ^{137}Cs (mSv) for different age groups calculated by FDMT for the Vindafjord municipality where the modelled deposition was 130 kBq m^{-2}.

Individual ingestion doses (mSv) from Cs-137		
Age (y)	First year	After 5 years
1	4,7	4,8
5	3,3	3,6
10	4,2	4,6
15	6,4	7,0
Adults	8,3	9,1

3.3.2 Rough grazing animal products and wild foodstuffs - STRATOS

The deposition map was combined with three aggregated transfer factors (T_{ag}'s) (expected, min, max) (see Appendix A.1.4 for details) for rough grazing animal products and wild foodstuffs. The modelled concentrations were compared to the food intervention levels for ^{137}Cs for trade of 600 Bq kg^{-1} for mushrooms, berries, and cheese; 370 Bq kg^{-1} for milk; and 3000 Bq kg^{-1} for game. As examples, the resulting maps presented in Figure 7 show which areas are expected to show activity concentrations above these limits for roedeer, wild berries and brown whey goat cheese.

The modelled concentrations can be combined with the statistics for production and hunting to show how large percentage of the annual production/yield will be above the food intervention level. For instance, 56 % of the produced goat's milk in Rogaland was estimated to be above the intervention level for milk.

This model can be used for any wild foodstuff or rough grazing animal as long as three T_{ag}'s (expected, min, max) can be properly estimated and the geographical distribution of production/hunting/gathering is known from e.g., regional or national statistics. The predicted concentrations can be further used in the ERICA Tool to calculate dose rates to terrestrial animals and plants.

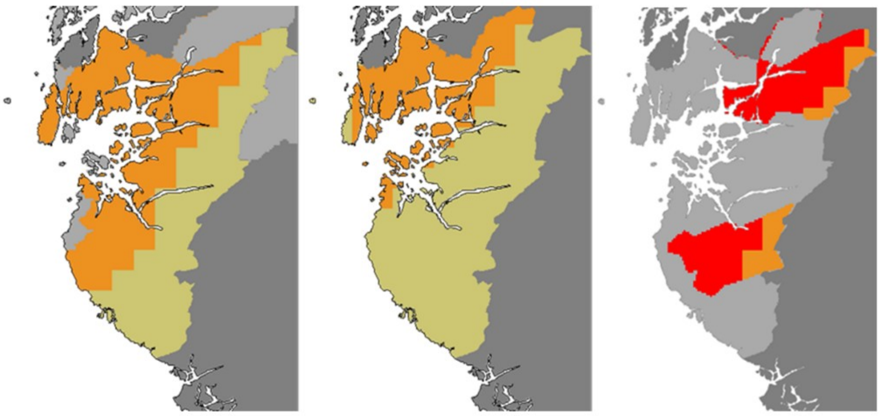

Figure 7: Maps indicating where food intervention levels might be exceeded for roedeer (left), wild berries (middle) and brown whey goat cheese (right). Colour coding: Red (lowest T_{ag}) – product will clearly be above in the long term perspective – years to decades; orange (expected T_{ag}) - product will most probably be above in the long term perspective (years); kaki (highest T_{ag}) – product might be above in the first year, but probably not in the following years; light grey – no production or hunting in this area; dark grey – outside Rogaland County.

3.4 Freshwater modelling - INCA-RAD

The Vikedal area consists of various lakes, streams and catchment areas with a main river that flows into the fjord Vindafjorden, an arm of the fjord Boknafjorden. The river Vikedal is popular for salmon fishing. The area has a coastal climate and the river may experience high flows particularly in the autumn. The map in Figure 8 shows the five different sub-catchments of the Vikedal catchment that are used in the modelling calculations. As an acid rain countermeasure, the lower part of Vikedal River has been extensively limed since 1987 (Sandlund et al., 2010). Due to a physical barrier (the liming station), the salmon and sea trout cannot migrate beyond the main river.

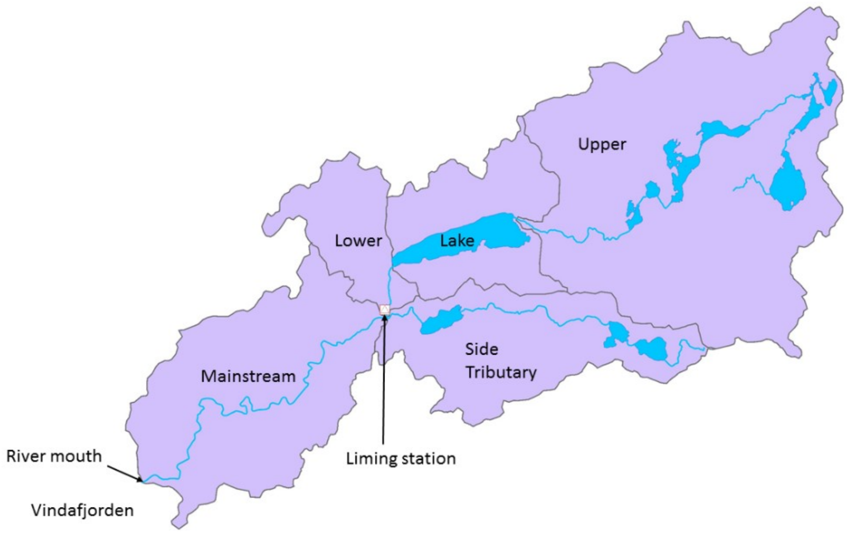

Figure 8: Map over the Vikedal catchment with a division into five sub-catchments used in the calculations: Upper, Lake, Side Tributary, Lower and Main stream. The liming station in the Vikedal River and the river mouth is indicated with arrows.

A mean geometric value for K_d of $2.9 \cdot 10^4$ L/kg (IAEA, 2010), was selected to run the INCA-RAD model for ^{137}Cs for the time period 19.10.2008 to 14.01.2014 using monitoring data for water flow from the Norwegian Water Resources and Energy Directorate and the SNAP raster files for deposition of ^{137}Cs (Bq m^{-2}). The results were activity concentrations in both water (Bq m^{-3}) and sediments (Bq kg^{-1}) in the five different sub-catchments in daily time steps. The predicted concentrations in water were fluctuating according to the daily water flow and generally decreasing steadily over time. The predicted concentrations in water were <10 Bq l^{-1} for ^{137}Cs for all areas, decreasing over time to a few Bq l^{-1} or less. For sediments, however, substantial concentrations were predicted, in particular in the Lake and Lower areas reaching levels of 223 000 and 574 000 Bq kg^{-1}, respectively, while the sediment concentrations peaked around 150 000 Bq kg^{-1} in the Main stream. In all sub-catchments but the Lake, the sediment concentrations are decreasing with time over the five years (see Figure 9).

INCA-RAD also predicted the outflow of ^{137}Cs to the estuary in Vindafjord in Bq d^{-1} for the same time period. The size of the drainage area and the deposition of radioactivity within the Vikedal area was used to estimate monthly outflow to the sea from 21 main rivers in Norway. They were used as additional point sources with a time-dependent flux of ^{137}Cs into the marine ocean model ROMS (see section 3.6.1).

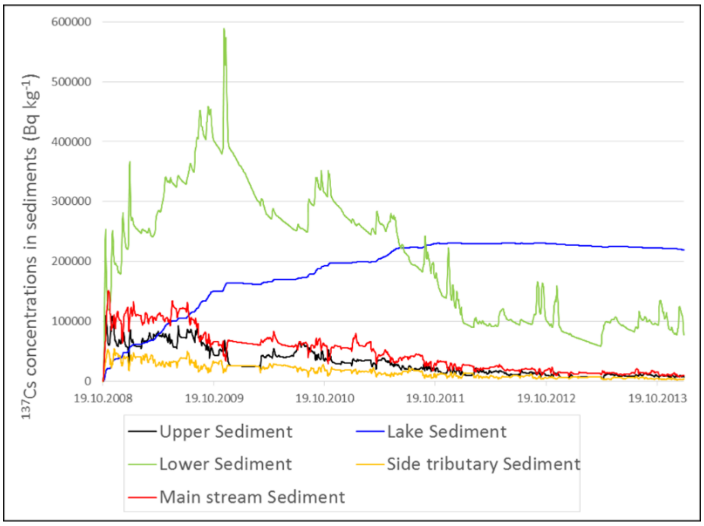

Figure 9: Time development of ^{137}Cs concentrations in sediments (Bq kg^{-1}) for all five areas.

3.5 Dose rates to aquatic organisms: from INCA-RAD to the ERICA Tool

The data on ^{137}Cs in water and sediments from INCA-RAD were used as input to the ERICA Tool version 1.0 to calculate dose rates to native biota. The report from Sandlund et al. (2010) gives an overview of the aquatic species present in the Vikedal area. They are presented in Table 3 together with the corresponding ERICA reference organisms and the concentration ratios and occupancy times used. Occupancy times are important in calculating dose rates since the exposures are either coming only from the contaminated water (for pelagic fish, salmon, zooplankton), only from the contaminated sediments (for insect larvae that live within the sediments) or from both water and sediments for gastropods and crustaceans that live on the sediment surface.

Table 3: Reference organisms and parameter values used in the ERICA Tool calculations

ERICA reference organism	**Representative species present in Vikedal**	**Concentration ratios** *Derived from*	**Occupancy times**
Pelagic fish	brown trout (*Salmo trutta*) and Arctic char (*Salvelinus alpinus*)	7100 L/kg *ERICA default value*	100 % in water
Salmon	Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta trutta)*	5600 L/kg *Hosseini et al, 2008*	100 % in water
Crustacean	various copepod species e.g. *Eudiaptomus gracilis*	$1.04 \cdot 10^4$ L/kg *ERICA default value*	100 % on sediment surface
Gastropod	freshwater snails e.g. *Lymnaea peregra, Radix balthica*	2800 L/kg *ERICA default value*	100 % on sediment surface
Insect larvae	larvae of mayfly such as *Baetis rhodani*	$1.04 \cdot 10^4$ L/kg *ERICA similar reference organism extrapolation*	100 % in sediments
Zoo-plankton	various species	1560 L/kg *ERICA default value*	100 % in water

The ERICA Tool has not yet incorporated a kinetic approach to uptake and depuration in organisms. Thus, daily fluctuations in dose rates according to daily variations in the concentrations in water and sediment, would not be representative of real life. A moving average over 15 days has been applied to the data on dose rates to account for this limitation.

The results from the combination of INCA-RAD and the ERICA Tool for six reference organisms in the Main Stream are shown in Figure 10 (left). The dose rates for pelagic species reflects the flow pattern and the development in water concentrations over time. Time periods of low flow, for instance, would be reflected in a dip in predicted dose rates to pelagic species (Figure 10, left). The dose rates for sediment dwelling organisms are related to the slower sedimentation processes with a large variation in predicted sediment concentrations between the five different areas (Figure 9).

The dose rates to zooplankton were very low, resting below 2 $\mu Gy\ h^{-1}$ for the whole period in all areas. The dose rates to salmon and pelagic fish were very similar and reached levels of around 10-12 $\mu Gy\ h^{-1}$ after a few days, then steadily decreased with time to levels below 2 $\mu Gy\ h^{-1}$ after a few years. The dose rates were clearly higher for sediment dwelling organisms than pelagic ones as shown for the Main Stream (Figure 10, left). This was predicted for all five areas (not shown). The dose rates are highest for insect larvae due to an anticipated 100 % occupancy time within the sediments. Due to sedimentation processes, the time development was very different in the Lake and in the Lower area compared to the other areas, as shown for insect larvae (Figure 10, right). For the Lake the dose rates increased for the first three years and then decreased very slowly. The dose rates after five years were around 35, 40 and 80 $\mu Gy\ h^{-1}$ for gastropods, crustaceans and insect larvae, respectively. In the Lower area the dose rates increased steadily for the first year to around 100 $\mu Gy\ h^{-1}$ for gastropods, 110 $\mu Gy\ h^{-1}$ for crustaceans and 220 $\mu Gy\ h^{-1}$ for insect larvae, then steadily decreased over the following years. For sediment dwelling organisms the exposure was significantly higher than the ERICA screening value of 10 $\mu Gy\ h^{-1}$ indicating that a more detailed site-specific study would be needed (in the event of an accident) to address the potential environmental risk. The modelled dose rates for sediment dwelling organisms are very sensitive to the modelled sediment concentrations, which in turn strongly depends on the sediment particulate matter value used in INCA-RAD.

Although the dose rates in pelagic fish remain low, the corresponding predicted concentrations in brown trout and Arctic char peaked at values around 58 500 Bq kg^{-1} in the Lake after 16 days. The values for salmon and sea trout in the Main Stream peaked at 51 000 Bq kg^{-1}. After 5 years the predicted concentrations had decreased to around 4500 Bq kg^{-1} for brown trout and Arctic char in the lake while for salmon and sea trout they had decreased to around 3500 Bq kg^{-1} in the Main Stream. Although the dose rates hardly exceeded the ERICA screening value of 10 µGy h^{-1} (Brown et al., 2008) indicating negligible environmental risk, the concentrations predicted in these fish species were clearly much higher than the food intervention level for wild freshwater fish in Norway which is currently 3000 Bq kg^{-1} for ^{137}Cs if traded for human consumption.

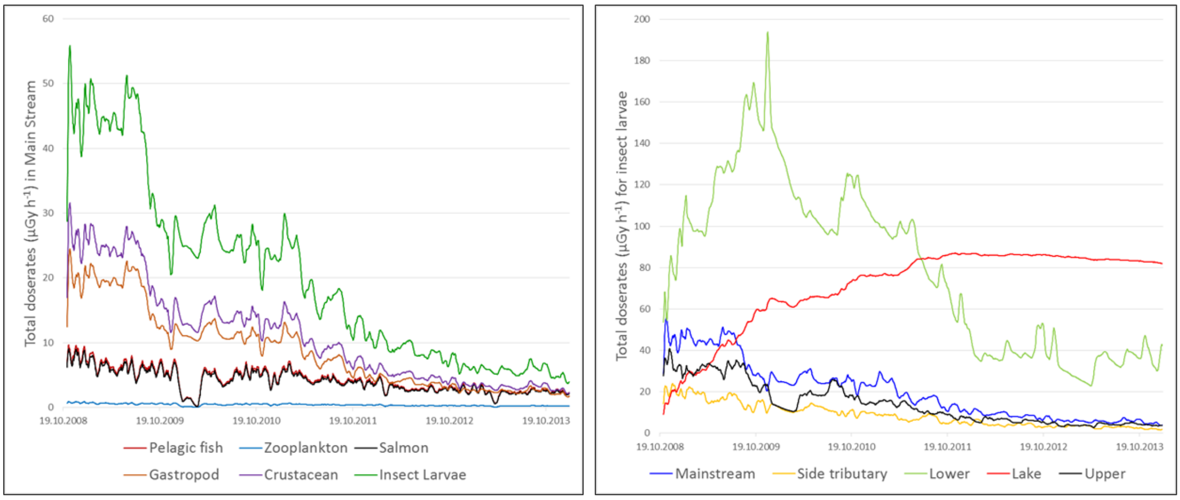

Figure 10: Dose rates (µGy h^{-1}) from ^{137}Cs for six reference organisms calculated with the ERICA Tool for the Main stream (left). Compared dose rates for insect larvae (right) from the five areas. All data transformed with a moving average of 15 days. Please note the different scale on the two y-axes.

3.6 Marine modelling

3.6.1 ROMS

The results from the dispersion modelling for three water depths at different dates after deposition are shown in Figure 11. For any chosen point in coastal areas or offshore, the variation in concentrations over time can be shown as a transect time series. This is shown in Figure 12 for two locations. The simulations were done both with and without riverine input as additional point sources. The magnitude of the riverine input was more pronounced closer to the Vikedal river mouth, i.e., in Boknafjorden, than in open waters off the coast. These data can be used to calculate concentrations in pelagic marine organisms and corresponding dose rates with the ERICA Tool as done for freshwater species. In Boknafjorden, when the outflow from the Vikedal River is taken into account, the predicted peak ^{137}Cs activity concentrations (using the ERICA Tool version 1.2) were up to 257 Bq kg^{-1} in pelagic fish, in macroalgae 294 Bq kg^{-1} and in molluscs 153 Bq kg^{-1} six days after deposition, decreasing rapidly to < 20 Bq kg^{-1} for all categories after one month. The corresponding peak dose rates were 4.7 (pelagic fish), 5.4 (macroalgae) and 5.2 (mollusc) μGy h^{-1}. Default values from the ERICA Tool version 1.2 were used for all calculations.

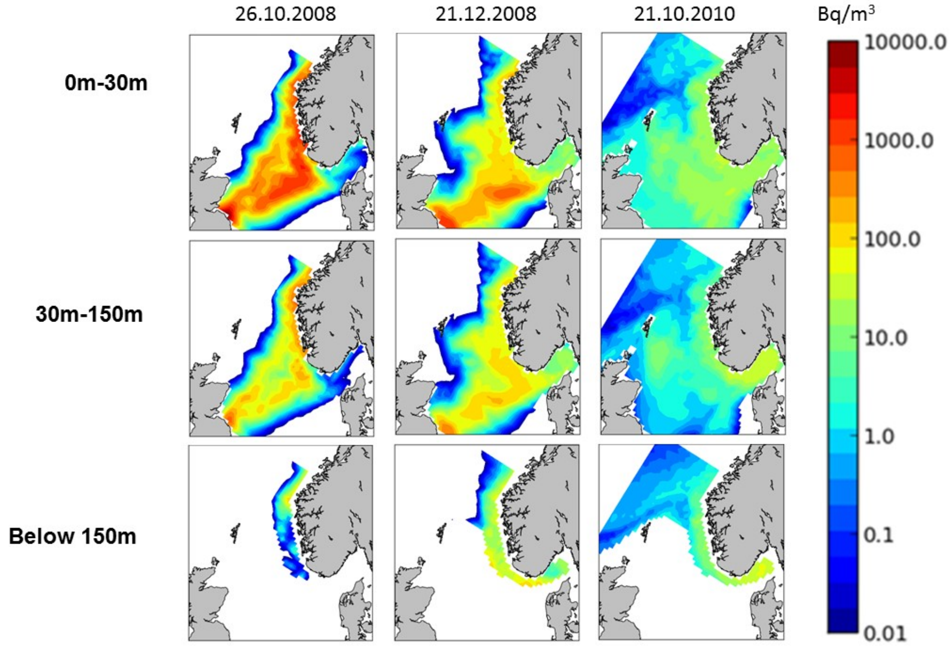

Figure 11: The dispersion calculated by ROMS for ^{137}Cs concentration (Bq m^{-3}) at three water depths: 0-30 m (top row), 30-150 m (middle row), below 150 m (bottom row) – and at three different dates after deposition: 26 October 2008 (left column), 21 December 2008 (middle column), 21 October 2010 (right column).

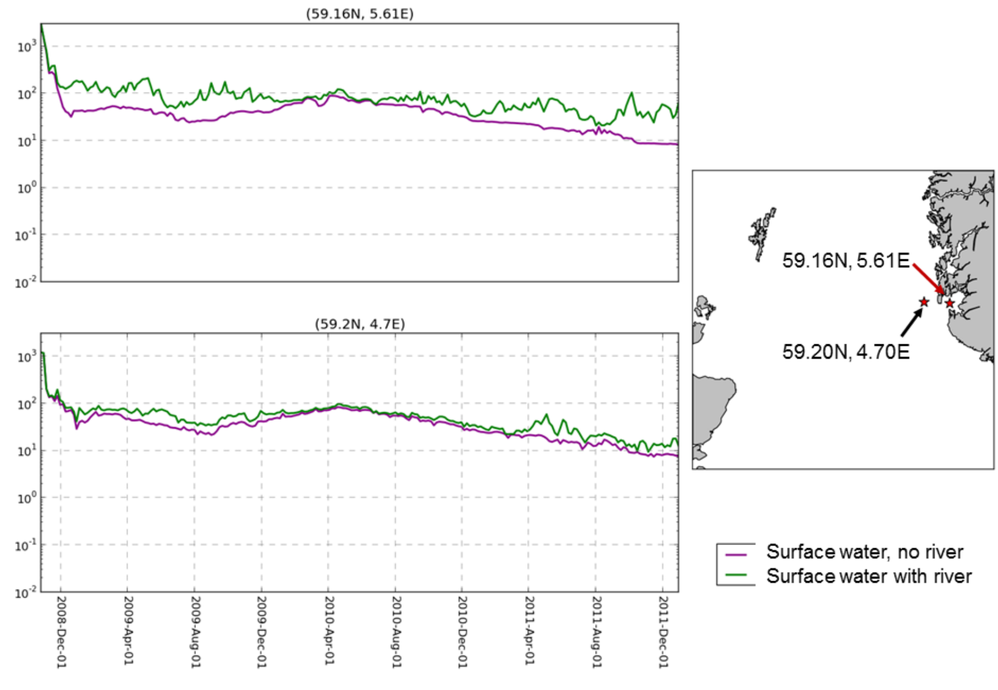

Figure 12: Transect concentrations for ^{137}Cs (Bq m^{-3}) in Boknafjorden (59.16M, 5.61E) (top) and off the coast (59.20N, 4.70E) (bottom) in surface water (0-30 m) without riverine input in purple. Green lines show the concentrations when riverine input is included in the model runs.

3.6.2 NRPA marine box model

The deposition on surface ocean water (Bq m^{-2}) was aggregated into total deposition per box and then used as input to the model with a deposition date of 20 October 2008. The total deposition in the box closest to Rogaland was assumed to be $3.25 \cdot 10^{15}$ Bq of ^{137}Cs. The model presumes that the radionuclides are instantaneously and uniformly mixed within the box. The radionuclides are then dispersed through the boxes and compartments with time. The concentration ratio values used were 100 L kg^{-1} for pelagic fish, 50 for crustaceans and 66 for molluscs according to IAEA (2004). The modelled concentrations were below 40 Bq kg^{-1} for all three species shortly after the accident, rapidly decreasing to < 5 Bq kg^{-1} within a few months. The corresponding dose rates predicted by the NRPA box model for the same organisms were all below 0.2 μGy h^{-1}.

3.6.3 Comparison of ROMS and NRPA box model results

The modelled concentrations and dose rates obtained by the dynamic ROMS model and by the NRPA box model have been compared, considering that only the former includes riverine input and dynamic dispersion. The ERICA Tool, using input from ROMS on water contamination, predicted concentrations about 7 times higher for pelagic fish in Boknafjorden than the NRPA box model for the box closest to Rogaland. However, water concentrations modelled by ROMS for the ocean area off the coast were about five times lower than within the fjord Boknafjorden closer to the river outlet (Figure 12). The concept of instantaneous and uniform mixing within the NRPA box model implies that the activity concentrations in the box closest to Rogaland should be more comparable to the ROMS area off the coast. Indeed, the modelled concentrations in fish is in relatively good agreement (differing with a factor of < 2) for the ocean areas off the coast of Boknafjorden.

3.7 Comparison with post-Chernobyl data

It is clear that an accident with an atmospheric release of this magnitude could potentially contaminate large areas substantially. The modelled deposition using a real historical weather scenario predicted deposition levels in Norway comparable to the Chernobyl accident (Figure 3). From experience, we know that this would lead to decades of challenges; over 30 years after the Chernobyl fallout event, countermeasures are still implemented in Norway to reduce the levels of ^{137}Cs in milk and meat to comply with the Norwegian food intervention levels for trade (Komperød et al., 2017) . A fallout of this magnitude would require decades of monitoring and countermeasures.

The predicted inhalation and external doses to humans were low and early protective actions such as evacuation or sheltering would not have been justified according to the Nordic guidelines and recommendations (2014) for protective measures in nuclear or radiological emergency. This is mainly due to the absence of short-lived radionuclides in the present scenario, which are usually contributing substantially to inhalation and external doses (e.g., ^{131}I, ^{134}Cs, ^{133}Xe, ^{132}Te).

The internal doses through a contaminated agricultural diet, however, would exceed the recommended maximum level of 1 mSv/y from ingestion. The predicted activity concentrations in agricultural produce were above maximum permitted levels and comparable to ^{137}Cs levels measured in Norway after the Chernobyl accident: i.e., up to 40 000 Bq kg^{-1} in lamb meat, 1200 Bq l^{-1} in cow`s milk and 2900 Bq $l^{-1}$ in goat`s milk (Liland and Skuterud, 2013). Agricultural countermeasures are clearly justified in this situation.

The predicted concentrations in freshwater fish are comparable to post-Chernobyl values as well, around 35 000 Bq kg^{-1} of ^{137}Cs was reported by Strand et al. (1992) and Brittain and Gjerseth (2010) for pelagic fish in Norwegian lakes. Also other forest foodstuffs such as mushrooms, berries and game were predicted to exceed the food intervention levels for trade the first years in some areas, which was also the case after the Chernobyl accident (Liland and Skuterud, 2013).

The contamination of marine species by ^{137}Cs was low after the Chernobyl accident, up to a few tens of Bq/kg in fish (IFE, 1986) since the deposition was very low in Norwegian marine areas. For this hypothetical scenario, we cannot rule out that food intervention levels might be exceeded for seafood close to the Vikedal river outlet (or other river outlets on the western coast) shortly after the accident. However, the transfer of ^{137}Cs to organisms in sea water is much lower than in freshwater so in general activity concentrations in marine fish and other seafood (wild and farmed) are predicted to stay below Norwegian food intervention levels. At the same time, the seafood market is very sensitive to information or rumours about contamination, and the export from Norway might be negatively affected even with low contamination levels due to public and market fear of radioactive contamination.

It should be noted that we have here compared true measurement data after the Chernobyl accident with conservative model estimates for a hypothetical accident. They are thus not directly comparable, but the model estimates are in the same order of magnitude as the measured data. This indicates that the model estimates are reasonable.

Furthermore, it should be noted that only ^{137}Cs was included in the pilot study described, although the HAL contains other long-lived radionuclides, too (see Appendix 2). Of these, ^{90}Sr would be present in large quantities and has a known high transfer in terrestrial food chains. This would add to the challenges presented for ^{137}Cs.

3.8 Key factors contributing to uncertainty

At the time this study was performed, the models did not have built-in uncertainty ranges when stating the results, although there are known variabilities and uncertainties in all of them. A single value, deterministic output from one model was used as input to the next model and this procedure was repeated along the whole chain of models. Each model was used to give a best estimate result, and no account was taken of the various inherent uncertainties associated with the calculations such as input, interpolation and extrapolation uncertainty, parameter uncertainty and variability, and algorithmic or structural uncertainty (Salbu, 2016). It is challenging to ascertain whether the results from the modelling chain can be deemed accurate or truly representative of the environmental system. The comparison between the modelled activity concentrations for the Sellafield scenario and the post-Chernobyl monitoring data on ^{137}Cs in Norway indicates, however, that the predicted values are within a realistic range.

Investigations have earlier been performed to identify the key factors contributing to uncertainties within each model, such as sensitivity analyses and expert judgements (e.g., Sørensen et al. (2014); Vives i Batlle et al. (2008); Iosjpe (2011); Iosjpe and Liland (2012); Iosjpe and Logemann (2014); Avila et al. (2004); Hansen et al. (2010)). Since one of the main CERAD goals is to reduce uncertainties in risk and impact assessments, further work has already been undertaken or is ongoing within CERAD to address this. Reducing the uncertainties in the models used in this pilot study is not only a question of finding the best parameter values and their underlying statistics/distributions; it is also important that the underlying model uses the most realistic representation of the geographical area and ecosystem studied.

Table 4 lists the key factors contributing to uncertainty for each model. Some of these key factors were prioritized in 2016 for updating through CERAD research (see Table 4, last column), based on their magnitude and the feasibility of reducing their influence on the total uncertainty in model calculations. Work to reduce these uncertainties continues to be a priority of CERAD, but is too extensive to be fully described here. Some of the work already undertaken since this study was performed, is presented below with due references.

Table 4: Key factors contributing to uncertainty for each model and those prioritized for updating in CERAD

Model	Key factors contributing to uncertainty	Prioritized updating in CERAD
SNAP	1. Model formulation 2. Input data – Inaccurate or wrong source term description *(location; time; particle size, density, shape; release rates and vertical range)* – inaccurate or missing meteorological input data *(precipitation field, velocity field, temperature field)* – Incomplete or over-simplified parametrizations in the model *(wet and dry deposition, advection and diffusion)*	• Use ensemble meteorological prediction instead of deterministic predictions • Improve parametrization of: – wet deposition (particle size/ density, precipitation) type/form/vertical distribution – dry deposition (particle shapes, receptor conditions)
ARGOS DSS	1. For human dose assessments: – particle sizes, breathing rate, occupancy and shielding effects from houses	• Particle sizes, will be done as part of SNAP
FDMT	1. Default data representative of Southern Germany: – Growth periods, leaf area indices, crop yields, migration rates – Transfer factors (TF), animal specific feeding rations, period of preparing winter feed – Human age-dependent consumption rates, seasonality of consumption rates	• Regional updating to Norwegian parameter values for: – growing season, harvest periods – feeding rations of cows, goats and lamb/sheep – ^{137}Cs and ^{90}Sr transfer factors for milk and meat – human age-dependent diets
STRATOS	1. Deposition 2. Aggregated transfer factors (T_{ag}'s) 3. Production/hunting/gathering statistics	• Fieldwork to establish regional Tag's • Extension to include ^{90}Sr
INCA-RAD	In order of relative importance for final sensitivity: 1. Sediment transport *(entrainment rate, splash erosion rate, flow erosion rate)* 2. Mineralization rate of solid organic matter 3. K_d 4. Residence time of water in the organic layer 5. Rate of solid organic matter to dissolved organic matter	• Sediment transport • K_d *(distribution coefficient between particle and water phase)* • $K_{w\text{-}doc}$ *(partitioning coefficient between water and dissolved organic carbon)* • Speciation codes • Transformation processes

	6. $K_{w\text{-}doc}$ 7. Temperature correction constant 8. Hydrological parameter for calculating the velocity of flow	
ROMS	*1.* Ocean model dynamics *(initial or boundary conditions, model resolution, description of turbulent mixing)* 2. Particle processes not included *(assumes passive, non-interacting radionuclides)* 3. Missing description of mixing zones in estuaries	• Increase resolution • Include radionuclide particle interactions • Include chemical speciation, particle sizes and kinetics of transformation processes, in particular for estuaries
NRPA box model	For ^{137}Cs in the Norwegian Current: 1. Porosity 2. Sediment distribution coefficient (K_d) 3. Reworking rate 4. Concentration factor (CF) *(For other radionuclides also: suspended sediment load, sedimentation rate, and coefficient of molecular diffusion)*	• For dispersion in water and sediments: porosity, K_d, sedimentation rate, suspended sediment load, coefficient of molecular diffusion • For bioaccumulation: CF, assimilation efficiency for different trophic levels, ingestion and excretion rates • Include biokinetics
ERICA Tool	1. Assumes equilibrium conditions 2. Concentration ratios 3. K_d 4. Bioavailability and mobility of radionuclides in the environment 5. Bioaccumulation factors 6. Gut uptake fraction	• Develop dynamic approach • Include biokinetics • Characterize time-dependent uptake, assimilation and depuration of radionuclides

The main sources of uncertainty for the SNAP model are the model formulation including the conceptual understanding of the source term, and in the parametrization of the input data. Uncertainties in the model formulation are related to our incomplete knowledge of the physical processes responsible for atmospheric transport and deposition of radioactive debris. In all severe nuclear accidents, radioactive particles ranging from submicrons to fragments will be released (IAEA, 2011; Salbu 2016). Particulate fallout would result in non-homogeneous hot-spots and single particles could become internal point sources if inhaled or ingested by humans or animals (Ytre-Eide et al., 2009), but this was not taken into account in our present study. If particles are present in the release, the size and density would influence the particle transport pattern and the deposition map as well as the subsequent ecosystem transfer estimates, being quite different from that of aerosols. If particles are ignored, the conceptual uncertainty in the source term is high.

Another example of source term uncertainty is the timing of an event. In this case study, MET Norway showed that a 3 hours difference in start time of the accident gives concentration and deposition changes with a factor of four. Likewise, uncertainty related to parametrization of wet deposition (due to variations in particle size and density, precipitation amount) in the present model structure is estimated to ±50% while for advection and diffusion it is estimated to only ±20%.

Regarding FDMT, the model outcomes are highly sensitive to the deposition time and to variations in a number of other input parameters. These parameter sensitivities are case specific (Müller and Pröhl, 1993), and does not provide general answers regarding the model uncertainties. However, general recommendations are available within the RODOS or ARGOS communities in relation to regional updating – where different parameters have been rated as being of high, moderate or low priority by the developers of ECOSYS or other experienced users of FDMT (Pröhl and Müller, 2005); (Raskob et al., 2000); (Hansen et al., 2010). Our priority parameters shown in Table 4 are largely based on these expert judgement recommendations. Important parameters in relation to adaptation to Nordic conditions have recently been identified and updated (Thørring et al., 2016) – focusing on (1) parameters of relevance to growing season and harvest periods of crops and grass, including seasonal development of leaf area indices (LAI), (2) animal feeding practice, and (3) human consumption of foodstuffs. Regional adaptation continues with work initiated on e.g., transfer and time-development of relevant radioelements. Furthermore, probabilistic simulations in FDMT is presently under development, as opposed to the present day deterministic runs.

For STRATOS, the focus is on continuing work done in relation to distribution/production maps (Thørring et al., 2010) and to include ^{90}Sr into the model. Such information is highly important in relation to vulnerability/preparedness – and for evaluating potential economic consequences of a particular fallout.

The long-term marine dispersion of historic ^{99}Tc discharges from Sellafield has been studied by Simonsen et al (2017), where the Lagrangian dispersion was modeled off-line, using the ROMS-TRACMASS model system (See Appendix A.1.6). Here, it was shown that increasing the model resolution improved the agreement with observations. It was also shown that a tidal forced Lagrangian drift could only be resolved in the simulations with relatively high resolution. This Lagrangian drift was particularly strong in the Irish Sea, heading northwards in those waters. The Lagrangian drift in the Irish Sea was found to impact the estimated activity concentration as far as in the Barents Sea. As element speciation and transformation processes are essential for marine transport, further work to include this in the ROMS-TRACMASS model has been achieved (Simonsen et al., 2019).

Sensitivity analysis has shown that a series of factors will contribute to the overall uncertainty in the INCA-RAD output. The analysis has shown that the initial value used for suspended particulate matters (mg L^{-1}) has probably been too high overestimating the sediment concentration of ^{137}Cs (Lin et al., 2019). The status of the INCA-RAD model at NIVA/CERAD is that the whole suite of INCA models will be subject to re-coding. A key issue is to implement a proper element speciation code including transformation processes that would affect the river transport and the input to estuarine zones.

The results of the implementation of a kinetic model for bioaccumulation processes into the NRPA box model (Iosjpe et al., 2016) clearly demonstrated that there is a significant quantitative difference between the kinetic modelling approach and the approach based on the constant concentration ratios. It is noteworthy that such differences can be observed over relatively long time periods. For example, the maximum differences between the two approaches can be seen 3, 4 or 5 years after start of the releases depending on scenario. It clearly shows that kinetic modelling of the bioaccumulation processes can provide a more correct description of the concentration of radionuclides in biota. Results also demonstrate that kinetic modelling of bioaccumulation processes leads to a better harmonisation between the different calculations (for example, between doses to the critical group and concentrations in marine organisms for short-life radionuclides) and to better logical explanations of the results.

The ERICA Tool is under a continued state of development, the latest publication describing improvements have been given in Brown et al. (2016b). A conspicuous limitation of the model is linked to the assumption of steady state distribution coefficients and constant transfer factors (concentration ratios) under conditions when ambient radionuclide activity concentrations are known to be changing rapidly with time. Recent work has involved the development of biokinetic models to more realistically account for the dynamics of food-chain transfer and as exemplified for particular cases involving instantaneous releases from dumped nuclear objects in the Arctic (Brown et al., 2016a); (Hosseini et al., 2017). Such models can be simply linked to components of the ERICA Tool to provide estimates of the environmental exposures and associated dynamics as demonstrated in the aforementioned publications.

4 Conclusions and further work

Predicted impacts from hypothetical, yet realistic, scenarios are important in emergency preparedness work to scale the necessary emergency plans, response strategies and measurement capacities. Our study has shown that using specialized and generic models together with DSS in a chain is useful to predict possible impacts from a large fallout of ^{137}Cs

over Norwegian territories. We used a combination of the ARGOS DSS and its modules (FDMT) and other modelling codes developed for Norwegian conditions (SNAP, NRPA box model, INCA-RAD, STRATOS) or of a generic character (ROMS, ERICA Tool) to predict activity concentrations and doses / dose rates to both humans and the environment via various exposure routes.

In the pre- or early accident phases, rapid assessment of possible impacts are necessary to take the right decisions to protect life, health and societal interests. DSS like ARGOS and RODOS, coupled with national meteorological atmospheric dispersion models, are very valuable for rapid support to decision makers for protecting people and the production of food, feed and goods. In these phases, decisions need to be taken within hours or 1-2 days. For other impacts, such as contamination of aquatic areas, forests and recreational areas, and likewise long-term consequences for food production systems, the decision makers have more time to decide. Time would allow the use of various regionally adapted models in combination with predicted deposition and measurement results before deciding on the necessary countermeasures. These models could, of course, be modules in a DSS that have been adapted to a region or country, or they could be national models developed specifically for an ecosystem and/or a region. Our study has shown that a chain of different regional/national models works well in combination with ARGOS DSS and could be used to assess impacts in a variety of ecosystems.

The modelled activity concentrations and doses / dose rates can be used further in optimising countermeasures for various sectors and ecosystems e.g., using the AgriCP module for agricultural countermeasures, or performing cost-benefit analyses for remediation actions. Recently, a cost-benefit analysis framework was adapted to radioactively contaminated sites and tested on a site contaminated by naturally occurring radionuclides in the Euratom TERRITORIES[2] project (Liland et al., 2019). This framework will now be used for evaluating agricultural countermeasures for this hypothetical Sellafield scenario.

The choice between a fully-fledged DSS with specific modules for all ecosystems or a combination of DSS modules and regional/national models, needs to be taken on a national level by the relevant experts. The suitability of the former and the resources needed for regional/national adaptation of parameter values, must be evaluated compared to the latter. The need for specific regional/national models could be more prominent for some countries, for instance marine models in Norway where fisheries and aquaculture are important for diet, employment and export. In any case, the modelling tools need to be set up and tested before an accident happens, if they are to have any value in a crisis management situation.

Last, but not least, CERAD will continue the effort of reducing model uncertainties to improve our health and environmental impact assessment tools for future nuclear accidents.

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[2] To Enhance unceRtainties Reduction and stakeholders Involvement TOwards integrated and graded Risk management of humans and wildlife In long-lasting radiological Exposure Situations (https://territories.eu/)

Appendix 1 – Description of DSS and models used to assess the impacts in Norway from a hypothetical accident scenario at the Sellafield site.

A.1.1 SNAP atmospheric dispersion and deposition

SNAP is a Lagrangian particle model developed at MET Norway (Saltbones et al., 1996). It simulates transport and deposition of radioactivity from an atmospheric plume of radioactive contaminants. The input data are the source term (activity in Bq, particle size and density) for the released radionuclides, the accident site coordinates, the release height and release time. The emitted mass of radioactivity is distributed among a large number of model particles. After the release, each model particle carries a given amount of the radioactive substances which can be in the form of a gas, a noble gas, aerosols or particles. Although aerosols and particles of varying size and density can be included in the model, only aerosols were assumed in the present work. The HIRLAM[3] and ECMWF[4] models are used as the meteorological input provider for SNAP.

The atmospheric boundary layer can have different depths usually ranging from 300 to 2500 m and the model assumes instantaneous and homogenous mixing of particles within the layer.

The basic outputs from the dispersion modelling are air concentration (Bq m^{-3}), time integrated air concentration (Bq·h m^{-3}), and deposition on ground (Bq m^{-2}). This information is given for each isotope of interest, and over a time period which is covered by the available meteorological forecast.

[3]HIRLAM, the High Resolution Limited Area Model, is a Numerical Weather Prediction (NWP) forecast system developed by European meteorological institutes (https://en.wikipedia.org/wiki/HIRLAM)

[4] ECMWF, European Centre for Medium-Range-Weather Forecasts

The model version for the present simulation (Bartnicki et al., 2011) used meteorological data from the HIRLAM model with a spatial resolution of approximately 10 km x 10 km.

A.1.2 ARGOS decision support system

ARGOS is a decision support system designed for response to nuclear and radiological accidents (Hoe et al., 2009). Its main purpose is to provide a set of tools which can help in assessing the consequences of an accident both prior to a release (predictive) and after. The main tools in ARGOS are dispersion modelling, management of measurement data and impact assessments in agricultural and urban areas. Furthermore, ARGOS contains a database of nuclear reactors including reactor inventory, source terms, radionuclides, dose conversion factors, population data etc. ARGOS supports different atmospheric dispersion models. Such models require access to large Numerical Weather Predictions (NWP) data and, in some cases, super computers to do the modelling. Thus, long range runs are done on remote servers often hosted by meteorological institutes. MET Norway provides this service through the long range model SNAP which is integrated with ARGOS. The ARGOS operator defines the position and amount of radioactive material released into the air over time (source term) and sends a request through ARGOS to MET Norway to perform dispersion calculations with SNAP. The results are received in ARGOS within 15 minutes.

The outputs are raster files with resolution in this work of 10x10 km on total deposition and concentrations in air (Bq m^{-3}, Bq·h m^{-3}) which ARGOS uses to calculate other aggregated outputs like dose to thyroid (if iodine is present), total effective dose, dose rate etc. With a good numerical estimate of the source term, the output from dispersion models can give an early assessment of potential consequences for humans following a nuclear accident.

It should be noted that the direct combination of SNAP through ARGOS DSS only works for recent and forecasted weather situations. For historic weather situations as used for this work, SNAP had to be started offline from the ARGOS DSS.

A.1.3 FDMT food chain modelling

FDMT is used in the Decision Support Systems ARGOS and RODOS to simulate transfer of radioactive substances in food chains following radioactive fallout. The user can select radioactive isotopes from 26 elements.

FDMT is based on the ECOSYS dynamic model developed in the early 1990's (Müller and Pröhl, 1993). Based on input data on deposition to soils and vegetated soils (Bq m^{-2}) and air concentrations (Bq m^{-3}), radionuclide concentrations in food and feedstuffs (Bq kg^{-1}) can be calculated for a chosen time period, from days to years. It includes a number of processes such as interception, translocation, weathering, root uptake, growth dilution, processing of feedstuffs, transfer to animal products, and processing of foodstuffs.

A large number of adjustable parameters are included in the module where some are dependent on site and situation, whereas others have a more general validity. A regional adaptation of these parameters is recommended as many of the default values are representative for Southern Germany. For a detailed description of FDMT parameters including default values reference is given to e.g., Müller et al. (2004).

A range of products are included in the model: feed stuffs such as grass, hay and maize; edible plants such as varieties of leafy vegetables, root vegetables and cereals; animal products (milk, meat, eggs); and processed food such as cheese, butter and beer. By also including a defined human diet, FDMT can calculate ingestion doses to humans (mSv) for various age groups, from infants to adults, over a chosen time period.

A.1.4 STRATOS pasture and wild foodstuffs modelling

STRATOS (Thørring et al., 2010) is a terrestrial model developed by the Norwegian Radiation and Nuclear Safety Authority to predict long term impacts on wild foodstuffs and rough grazing animals in Norway, i.e., wild berries, mushrooms, game, reindeer, lamb, and goat milk – foodstuffs not presently covered by FDMT in a satisfactory way. It is a screening model with the purpose of distinguishing between areas where food intervention levels might be exceeded and areas where they are not. The model incorporates information on (a) deposition of ^{137}Cs and ^{134}Cs, (b) aggregated transfer factors (T_{ag}) for Cs to vegetation or animals, (c) Norwegian food intervention levels, and (d) geographical information on distribution of grazing animals / hunting statistics.

The aggregated transfer factor is defined as the ratio between the activity concentration in a given animal or plant (Bq kg^{-1} fresh weight) and the total deposition density in the grazing area (Bq m^{-2}). Uncertainty/variability regarding transfer is reflected in the model by using three different T_{ag}: minimum, expected and maximum. The values used are based on Norwegian post-Chernobyl research, see Table A.1, and are generic for the whole of Norway without regional specific values.

Table A.1: Aggregated transfer factors used in the STRATOS model (Thørring et al., 2010).

Product	**Harvest period**	**Transfer factor**		
		Expected	min	max
Wild berries	Jul-Sep	0.007	0.0003	0.04
Mushrooms	Jul-Oct	0.02	0.0005	0.2
Moose	Sep-Nov	0.02	0.005	0.2
Red deer	Sep-Nov	0.02	0.005	0.2
Roe deer	Oct-Des	0.05	0.005	0.2
Reindeer	Late Oct-Mar	0.25	0.05	1.5
Reindeer	Sep-early Oct	0.15	0.05	0.5
Lamb	Oct-Des	0.04	0.01	0.2
Goat milk	Jun-Sep	0.007	0.001	0.02

The maximum T_{ag} can typically be representative of the first period after an accident or for particularly vulnerable areas. The expected transfer factor is the most likely transfer based on the existing amount of data. If the food activity concentration exceeds the food intervention level using this T_{ag}, it means that countermeasures will probably be necessary for many years. The minimum T_{ag} may be interpreted as being representative of areas of very low sensitivity to radioactive caesium and/or for the situation decades after an accident. If the results from a model run shows that the food intervention level is exceeded in a given area using the minimum T_{ag}, the foodstuff will probably exceed the intervention level for many years, even decades.

The model does not include an explicit time function. The estimations of how long the contamination of a given foodstuff will be above the food intervention level is based on the range of T_{ag}'s from min to max.

Activity concentrations can be coupled with geographical information on the number of grazing or hunted animals to estimate the proportion of the annual production that is exceeding intervention levels for a given area.

A.1.5 INCA-RAD freshwater modelling

The INCA (INtegrated CAtchment model) family of codes include a set of tools aimed at predicting hydrological and biogeochemical processes controlling lateral transfer of nutrients such as sodium, phosphorus and carbon (Futter et al., 2007); (Wade et al., 2004); (Whitehead et al., 2011), and environmental contaminants such as mercury (Futter et al., 2012) from soils to rivers, as well as riverine transport and sediment dynamics as a function of climate, hydrology and land use. A new synthesis model - INtegrated CAtchment model for RADionuclides (INCA-RAD) was developed to integrate several features of previous INCA models and adding the capability of simulating and predicting the transport and retention of radionuclides in river basins at catchment-scale and in daily time steps. INCA-RAD contains a rainfall-runoff hydrological module, a sediment transport and particle erosion module, a biogeochemical cycling module, and a radionuclides geochemistry module. Based on user-defined land-use type and river stretches geometry, and using time series of radionuclides deposition (Bq m^{-2}) as inputs, INCA-RAD simulates the temporal variation in radionuclides export from different land-use types within a river system, as controlled by reactive transport and by radioactive decay. INCA-RAD is able to simulate the transport of radioactive elements in several phases, such as dissolved phase, associated with suspended particles and bed sediment. A summary of the main mass transfer pathways is presented in Figure A.1. Combined with the hydrological simulations, INCA-RAD outputs the radionuclide concentrations in the water phase, the particulate phase and the river bed sediment. Aggregate values, such as annual flux of radionuclides at a catchment's outlet are also computed and utilized as input to marine models.

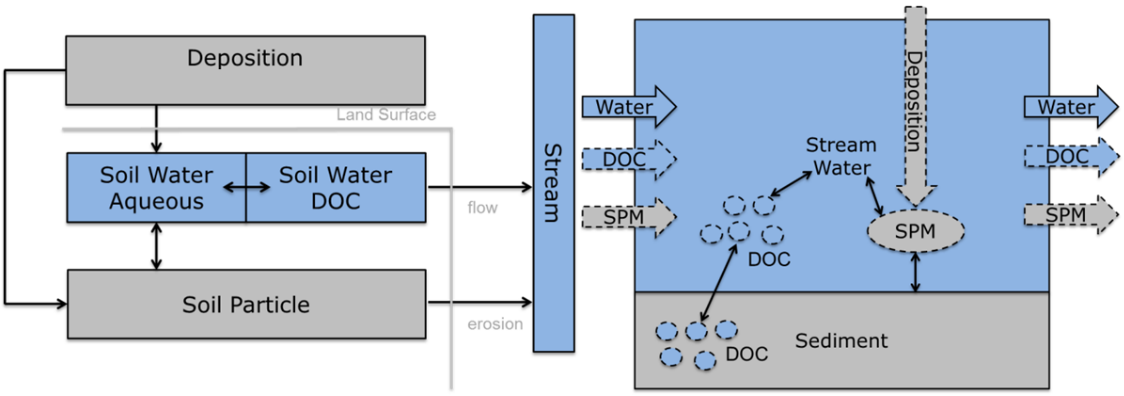

Figure A.1: Summary of the main mass transfer pathways in the INCA-RAD model. DOC = Dissolved Organic Carbon, SPM = Soil Particulate Matter.

A.1.6 ROMS marine dynamic modelling

The Regional Ocean Model System (ROMS, http://myroms.org) is a three-dimensional, free-surface, Bossinesq, hydrostatic ocean circulation model (Haidvogel et al., 2008). The model is an open-source software, with numerous users and application areas. At MET Norway, the model is used in the operational ocean forecasting system, as well as for research purposes within ocean modelling. In the operational MET Norway system, this model system is set up in a production line with different configurations: from a regional model with 4 km x 4 km resolution covering the Nordic Seas to an 800 m x 800 m model covering the Norwegian coastal waters.

In the dispersion simulations for radioactive discharges in ROMS used in this study, the radionuclides are assumed to be non-reactive, conservative and totally dissolved. Consequently, ^{137}Cs was computed as a tracer that follows the three-dimensional ocean currents passively. Interaction with sediments and suspended particles was not considered in this experiment, even if we know that this is an over-simplification that causes relatively large uncertainties (Simonsen et al., 2019). Computation of the tracer dispersion is performed on-line simultaneously with the ocean model.

In addition to the on-line simulations with ROMS, an off-line dispersion model, TRACMASS (www.tracmass.org) is used for simulations of particle transport paths. A finite number of numerical particles are released in the model, according to a given discharge scenario. The release can be a spatial distribution, and/or a time-dependent function. Transport of the particles is computed from the previously simulated ocean model velocity fields from the hydrodynamic ocean model (ROMS). The spatial resolution will depend on the output from the ocean circulation model. Each numerical particle will represent a given activity. The concentration of radioactivity in sea water can be computed from the density of particles in a certain water volume.

We considered two sources for ^{137}Cs in this marine model: atmospheric fallout and riverine input. The atmospheric fallout was modelled as an instantaneous deposition on 22 October 2008, which corresponds to the accumulated wet and dry deposition the three first days after the hypothetical accident. The surface deposition is distributed vertically into the water column by:

$$C(z) = C_0 e^{\frac{z}{\lambda}}$$

Equation A.1

where C_0 is surface concentration, z is water depth (a negative number) and λ is an e-folding depth, here chosen to be 4 m. In addition to marine surface deposition, riverine input of ^{137}Cs can be included as point sources with a time-dependent flux of ^{137}Cs into the marine model.

The outputs are dynamic activity concentrations in sea water (Bq m^{-3}) at various depths for the chosen time period.

A.1.7 NRPA marine box model

The marine box model developed at NRPA uses a modified approach for compartmental modelling which allows for dispersion of radionuclides over time based on either point source discharges or deposited radionuclides on the surface as input data. It assumes instantaneous and uniform mixing of radionuclides within the total box volume. The box structures (Figure A.2, left) for surface, mid-depth and deep water layers have been developed based on description of polar, Atlantic and deep waters in the Arctic Ocean and the Northern Seas and site-specific information for the boxes generated from the 3D hydrodynamic model NAOSIM (Karcher and Harms, 2000).

The volume of the three water layers in each box has been calculated using detailed bathymetry together with GIS. The box model includes the processes of advection of radioactivity between compartments, sedimentation, diffusion of radioactivity through pore water in sediments, resuspension, mixing due to bioturbation, particle mixing, and a burial process of radioactivity in deep sediment layers (Figure A.2, right). Radioactive decay is calculated for all compartments. The output is radionuclide seawater and pore water concentrations (Bq m^{-3}) and sediment concentrations (Bq kg^{-1}). In addition, the model calculates contamination of marine organisms (Bq kg^{-1}) based on concentrations ratios (CR) which is the ratio of concentration in the organism tissue (fresh weight) to that in water (IAEA, 2004). Dose rates to biota (µGy h^{-1}) are then calculated using dose conversion factors. Doses to man can be calculated from the concentration in biota and data for human consumption in the respective areas. More detailed descriptions are given in Iosjpe et al. (2002), Iosjpe (2006) and Iosjpe et al. (2009).

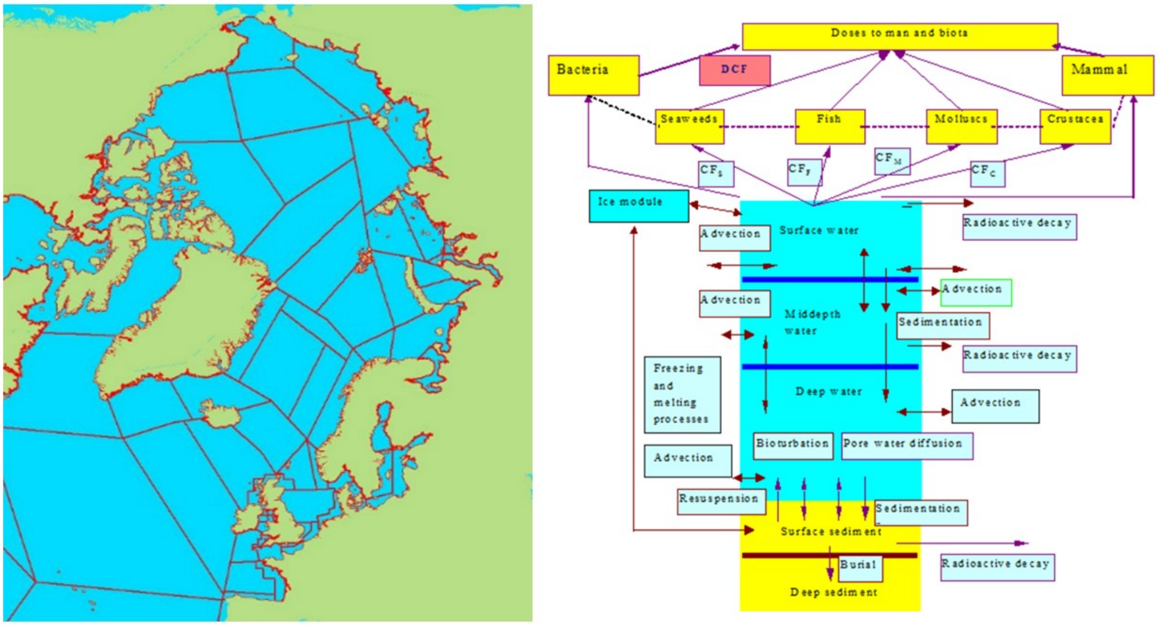

Figure A.2: The division of ocean areas in compartments (left) and processes included (right) in the NRPA marine box model

A.1.8 The ERICA Tool environmental risk assessment

The ERICA Tool (Brown et al., 2008) provides a means for calculating radiological environmental risk based on input data either in the form of discharges to the environment or from radionuclide activity concentrations measured in biota and environmental media (e.g., water, sediment). It includes default values for a suite of radioisotopes from 31 elements selected to cover a wide variety of conceivable exposure situations, including accidental releases.

The models used to quantify transfer from water to plants and animals are simple in nature having been based upon concentration ratios (CR) with an implicit assumption of equilibrium (steady state conditions for radionuclides between abiotic and biotic compartments). Similarly, sediment activity concentrations are derived using distribution coefficients (K_d's) derived from various compendia e.g., IAEA (2010). The underlying transfer databases have been updated to be compatible with comprehensive international compilations documenting these parameters (Copplestone et al., 2013). The concentration values are coupled to occupancy factors and internal and external dose conversion factors to calculate total dose rates (μGy h^{-1}) to a selection of reference organisms.

The Tool uses a default screening dose rate of 10 μGy h^{-1} applicable to incremental exposures. The derivation of this value is described in Garnier-Laplace et al. (2008). For dose rates below 10 μGy h^{-1} the environmental risk is arguably negligible (Brown et al., 2008), while dose rates above this screening value indicates that a more detailed assessment with site-specific data should be performed to determine the potential environmental risk.

Appendix 2 – Estimation of the inventory in the HASTs and the source term

The vitrified waste comes in canisters that equates to 10 TeU (‘Tonnes equivalent Uranium’) of spent fuel (Nuclear Decommissioning Authority, 2010). The activity of different nuclides in such a canister is given by e.g., Chubu Electric Power (2014), a company who receives vitrified waste from Sellafield Ltd after reprocessing of spent Japanese nuclear fuel. In their press release, we find that for ^{137}Cs we have 3.0 - $4.7 \cdot 10^{15}$ Bq per canister. Prognoses from a meeting between the Norwegian Radiation Protection Authority and the Nuclear Decommissioning Authority (NDA) in June 2013 show an expected stock of about 6300 TeU of HAL for April 2014. With 10 TeU/canister, this amounts to 630 canisters. The total amount of ^{137}Cs is therefore approximately 1.9 - $3.0 \cdot 10^{18}$ Bq. This agrees well with the plans and signals given by the NDA.

We have not done any probability calculations for a possible accident scenario, be it the loss of cooling, a natural disaster or a malevolent act. We just assume that there will be loss of the HAL tanks’ integrity and that 1% of all the estimated ^{137}Cs in the tanks is released to the air ($3.0 \cdot 10^{16}$ Bq) as an aerosol at a height where it can be mixed with the atmosphere and then transported to Norway by atmospheric dispersion.

Only ^{137}Cs was included in the pilot study described. It should be noted that the HAL contains other long-lived radionuclides besides ^{137}Cs that would add to the problem. ^{90}Sr and ^{90}Y are present in the liquid waste at the same order of magnitude as ^{137}Cs. ^{241}Am is present in quantities between 1 and 10 %, ^{244}Cm and $^{154/155}$Eu around 1% while other radionuclides are all in the range 0.001-0.1 % of the ^{137}Cs content (Chubu Electric Power, 2014). Americium and curium are transuranium elements that exhibit low transfer in the terrestrial foodwebs. For instance, in IAEA TRS472 (IAEA, 2010) the recommended gastrointestinal fractional absorption values for ruminants are 1 for Cs, 0.3 for Sr and 0.0005 for Am and Cm.

Similar differences are apparent for transfer factors from soil to plant and transfer coefficients to milk and meat. Data are scarce for yttrium (Y) and europium (Eu), but the latter is one of the lanthanides known to exhibit low transfer in terrestrial ecosystems, and Y is a transition metal with similar chemical properties as the lanthanides. ^{241}Am, ^{244}Cm, $^{154/155}$Eu and ^{90}Y would thus be of minor importance compared to ^{137}Cs and ^{90}Sr when it comes to terrestrial food chain transfer. The two latter are elements that resembles the essential elements potassium (K) and calcium (Ca) that are important nutrients in all living matter, contributing to a significantly higher transfer of these elements in terrestrial foodwebs.

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