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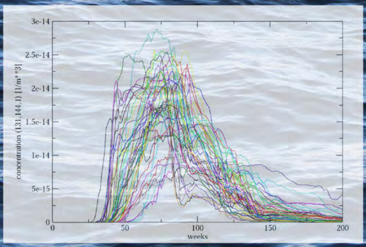
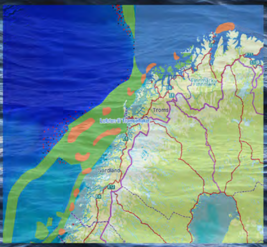
StrålevernRapport 2015:9





$$E_{int} = \sum_i (\tau)_i \times I_i$$

Geographical Categorisation of the Environmental Radio-sensitivity of the Northern Marine Environment

Risk indices for use in the aftermath of an accidental release of radioactivity to the marine environment

Reference:
Brown JE, Dowdall M, Hosseini A, Karcher M, Kauker F. Geographical Categorisation of the Environmental Radiosensitivity of the Northern Marine Environment.
StrålevernRapport 2015:9. Østerås: Statens strålevern, 2015.

Key words:
Radioactivity, marine advection dispersion modelling, radiological risk

Abstract:
Accidents involving the transport of radioactive materials in Northern seas may lead to detrimental consequences for Norwegian coastal environments. A method is presented in this report that allows radiological risk to be quickly derived accounting for the variability associated with the advection and dispersion of contaminants in marine systems.

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StrålevernRapport 2015:9. Østerås: Norwegian Radiation Protection Authority, 2015.
Language: English.

Emneord:
Radioaktivitet, marin spredningsmodellering, risikovudering fra stråling

Resymé:
Ulykker med transport av radioaktivt materiale i nordlige havområder kan føre til uheldige konsekvenser for norske kystmiljøet. En metode er presentert i denne rapporten som gjør at radiologisk risiko kan bli raskt derivert med hensyn til variasjon knyttet til adveksjon og spredning av forurensning i marine systemer.

Head of project: Justin Brown
Approved:

Per Strand

Per Strand, director, Department of Nuclear Safety and Environmental Radioactivity

25 pages.
Published 2015-09-07.
Cover design: 07 Media.
Printed number: 50 (09-15)
Printed by 07 Media.

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www.nrpa.no
ISSN 1891-5191 (online)
ISSN 0804-4910 (print)

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Geographical Categorisation of the Environmental Radio-sensitivity of the Northern Marine Environment

Risk indices for use in the aftermath of an accidental release of radioactivity to the marine environment

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Østerås, 2015

Innhold/Contents

1 Introduction

1.1 Background

Although land-based facilities have been and are the greatest contributors to radioactive contamination of the northern marine environment (1), the region is also vulnerable to marine releases from vessels carrying or powered by nuclear materials. Nuclear-powered vessels, both civilian and military, have operated in northern waters for many decades. These vessels have been primarily those of the Russian Northern Fleet and the nuclear icebreakers and service ships operated by Russia. Civilian transport of materials of the nuclear fuel cycle has been conducted along routes through the northern marine environment for many years. The largest dedicated fleet for the commercial transport of nuclear wastes and spent fuel is that operated by Pacific Nuclear Transport Limited (PNTL) owned jointly by International Nuclear Services (INS) of the UK, Japanese utilities and Areva of France. The PNTL fleet consists of the Pacific Egret, Pacific Grebe and the Pacific Heron while two other vessels - the Atlantic Osprey and the Oceanic Pintail - are owned by INS. The ships conform to all relevant international safety standards, notably INF-3, as set by the International Maritime Organisation (IMO), allowing them to carry highly radioactive materials such as high-level wastes, used nuclear fuel, mixed-oxide (MOX) fuel, and plutonium. The Atlantic Osprey has an INF-2 classification. Russia has taken delivery of a nuclear fuel and radioactive waste transport ship - the 4100 tonne Rossita, built in Italy and completed in mid-2011 after being launched at the end of 2010. The Rossita is designed for transporting spent nuclear fuel and other materials from decommissioned nuclear submarines from former Russian Navy bases in North-West Russia, and will be deployed on the Northern Sea Route serving Gremikha and Andreeva Bay, to cover requirements for spent nuclear fuel and radioactive waste shipments in northwest Russia over the period of rehabilitation of these territories. A number of other vessels have been or are involved in the transport of nuclear materials. These include the Danish flagged MV Puma, the Maltese registered Kapitan Lus, the Liberian registered MCP Altona and the Russian flagged MCL Trader. Sweden's SKB owns a purpose-built 2000 tonne ship, MS Sigyn, operated by Furetank Rederi AB, which has been used for transporting spent nuclear fuel from nuclear reactors to an interim waste storage facility. The Sigyn was replaced by the Sigrid during 2013-2014.

As has been evidenced by events over a number of decades, the public has been, and remains, very sensitive to radioactive contamination of the marine environment, a sensitivity that extends to even the potential for, or rumours concerning, contamination and which has the potential to generate socio-economic impacts that extend far beyond any impact arising directly from actual contaminant levels. This sensitivity to potential, yet ultimately unrealized contamination, was especially apparent after the sinking of the Komsomolets in international waters of the Norwegian Sea (73”43’16” N and 13”16’52” E) on the 7th of April 1989 (see: (2)) and the sinking of the Kursk on the 12th of August 2000 in the Barents Sea (see: (3)). Public concern as to potential contamination of the marine environment has not abated even as general contamination levels over the years have dropped and the focus of much attention in recent years has been on the transport of nuclear materials. Despite the fact that there has never been a significant release from a marine vessel carrying nuclear waste or spent fuel (4) and the stringent regulations under which such transports are conducted, this unease has recently focused on shipments of nuclear materials between the reprocessing plants of Europe and their customers and shipments conducted under the Global Threat Reduction Initiative (GTRI). The GTRI works towards, among other things, converting research reactors and isotope production facilities from highly enriched uranium (HEU) to low enriched uranium (LEU) and removing and/or disposing of excess nuclear and radiological materials. The repatriation of Russian-origin HEU fresh and spent fuel from research reactors in countries such as Poland, Germany, Ukraine, Romania, Bulgaria, Latvia and countries of the former Yugoslavia has resulted in a number of shipments of nuclear material taking place over the past years of relevance to the northern marine environment. Given the nature of the cargoes, information regarding such shipments is relatively scarce although a number of shipments have attracted specific attention. In 2010, the MV Puma carried a load of spent nuclear fuel from a research reactor in Belgrade along the Norwegian coastline to the Russian port of Murmansk. The same year, the MCL Trader transported HEU from Poland along the same route. The Polish material, originally from the research reactors Ewa and Maria, was transported under a 2009 agreement between Poland and Russia facilitating the transport of all HEU in Poland to Russia over a period of twenty years. Czech spent nuclear fuel was transported aboard the Mikhail Dudin to Murmansk early in 2013. These shipments were the subject of heightened attention in a number of countries once information became available and have served to set a spotlight on the transport of nuclear materials by sea as a cause of public concern.

Although the entire northern marine area is a rich fishery, the waters of the Lofoten archipelago between the 68th and 69th parallels are especially valuable as a fisheries resource (5). Of the 3 million tonnes of fish extracted yearly from the Barents and Norwegian Seas, approximately 70% have spawning grounds in the Lofoten area or utlise the area during their early life stages. The area is especially important for Northeast Arctic cod (*Gadus morhua*) and Norwegian spring-spawning herring (*Clupea harengus L.*), which constitute the largest populations (6). Other species for which the area is equally important but which are in themselves not as economically vital as the previous two species, include Northeast Arctic haddock (*Melanogrammus aeglefinus*), Northeast Arctic Pollock (*Pollachius virens*), deepwater redfish (*Sebastes mentella*), tusk (*Brosme brosme*) and ling (*Molva molva*). In total these species represent an equivalent economic importance to either herring or cod alone.

1.2 Considerations for impact assessments

With respect to potential impacts from discharges of radioactivity, we are normally interested in risks to human health and the environment. In the event of an accident involving releases of radioactivity, priority has been and will continue to be focused on human welfare but once initial concerns about the health of the public are appeased and in the consideration of areas where humans cannot be impacted, emphasis will often be placed on defining environmental risk. In this regard, when addressing the requirement to consider post-accident environmental impacts from radiation, the International Commission on Radiological Protection (ICRP) argue that a suitable approach may be useful in communicating the implications of the situation to stakeholders, particularly in relation to environmental conditions where humans have been removed from an area, and food chains leading to human exposure have been severed (7).This subordinate environmental impact consideration has not been particularly well addressed until recently as, unlike the case of human dose assessments, there are no universally agreed methods of defining risk to the environment.

With regard to contextualizing the potential impacts for humans, one approach may be to compare predicted concentrations in seafood with current contamination levels in harvested species of fish etc. in particular sea areas or with relevant intervention limits. This is essentially the approach adopted in a study (8) to consider the impact on the Barents and Norwegian Seas from the Komsomolets and K-159 following theoretical releases from these sunken submarines. An intervention level of 600 Bq/kg fw for the activity concentration of ^{137}Cs in food, as set by Norwegian authorities, and a background level of 0.2 Bq/kg fw ^{137}Cs, equal to the current contamination level in cod in the Norwegian and Barents Seas, were used to contextualize model prognoses (8). Alternatively, a more direct means of defining risk to humans might be achieved through the comparison of human (committed effective) doses with corresponding benchmarks such as a De Minimis level. The alternative approach might be considered more 'direct' because the concept of dose is more conspicuously related to potential human health effects through, for example, the nominal coefficient for detriment-adjusted (cancer + heritable effects) risk (9) than an approach using, for example, intervention levels that have a somewhat indirect relationship with risk (as often other considerations such as political, societal and economical perspectives are introduced).

What is clear is that in conducting robust assessments, concerning the impact of radionuclide releases, methodological components quantifying both risks to humans and the environment are important requirements.

Another key issue in conducting an assessment relates to variability. It is known that environmental systems can be highly variable in both time and space and for the sake of undertaking a robust, defendable assessment some means of quantifying this variability needs to be found. Essentially, predictions cannot be many months in advance, because the weather conditions that might be prevalent when an accident might occur and in the subsequent period following the accident leading to the dispersion of contamination are essentially unknown. The best that can be aimed for is to provide a retrospective analysis, covering an acceptably long period of time (from practical considerations relating to the quality of atmospheric forcing data this has been taken to be several decades), wherein the chance of capturing extreme or unusual periods of contaminant dispersal is high. This might also allow us to ensure that the long-term variability of the system is reasonably characterized. The dispersion of radionuclides from a source is affected by the variability of oceanic currents with respect to flow speed, mixing intensity in the horizontal as well as the vertical, width and depth of the currents. This variability can be caused by local or small-scale forced processes and unforced processes (internal variability of the sea ice-ocean system). In addition large-scale flow pattern changes, often linked to specific atmospheric forcing patterns, have the potential to significantly alter the pathways of dispersion.

1.3 Objectives

With the above considerations in mind, the objectives of this study were: to characterize the underlying variability for a given release scenario and to develop a means of quantifying risk to man and the environment taking into account this variability.

2 Methodology

2.1 Particularly valuable and vulnerable areas

The 2006 management plan for the Marine Environment of the Barents Sea–Lofoten Area identified particularly valuable and vulnerable areas (10). These are areas that, on the basis of scientific assessments, were identified as being of great importance for biodiversity and for biological production in the entire Barents Sea–Lofoten area. These include the area from the Lofoten Islands to the Tromsøflaket, the Tromsøflaket bank area, and the Eggakanten area.

The areas are identified as particularly valuable and vulnerable based on a combination of qualities; such as, high nutrient content seawater and high phytoplankton production, their function as spawning grounds or part of a spawning migration route for fish, or as breeding, moulting and wintering areas for seabirds. Other areas may be valuable because there are colonies, breeding areas or other concentrations of marine mammals such as grey seals, common seals, common porpoises and killer whales. Others again are classified as particularly valuable and vulnerable because there are sponge communities and coral reef complexes on the seabed, which in turn provide habitats for other species. The presence of habitat types that are listed as threatened and/or declining by OSPAR have also been documented in these areas. Most seabird species in many of the valuable and vulnerable areas are declining, particularly along the mainland coast. There is no new information indicating that the status of any of the areas identified as particularly valuable and vulnerable in 2006 should be changed.

2.2 Selection of the impact area and release point

The main 'measurement' area was the "Lofoten to Tromsøflaket" (Figure 1). This area was selected because it has been identified by the Norwegian Authorities as being an area that is considered to be especially valuable and vulnerable to impact as described above and is also one of Norway's most commercially important fisheries. The data from the "Lofoten til Tromsøflaket" formed the basis of the probability density functions, pdfs, used in subsequent risk calculations.

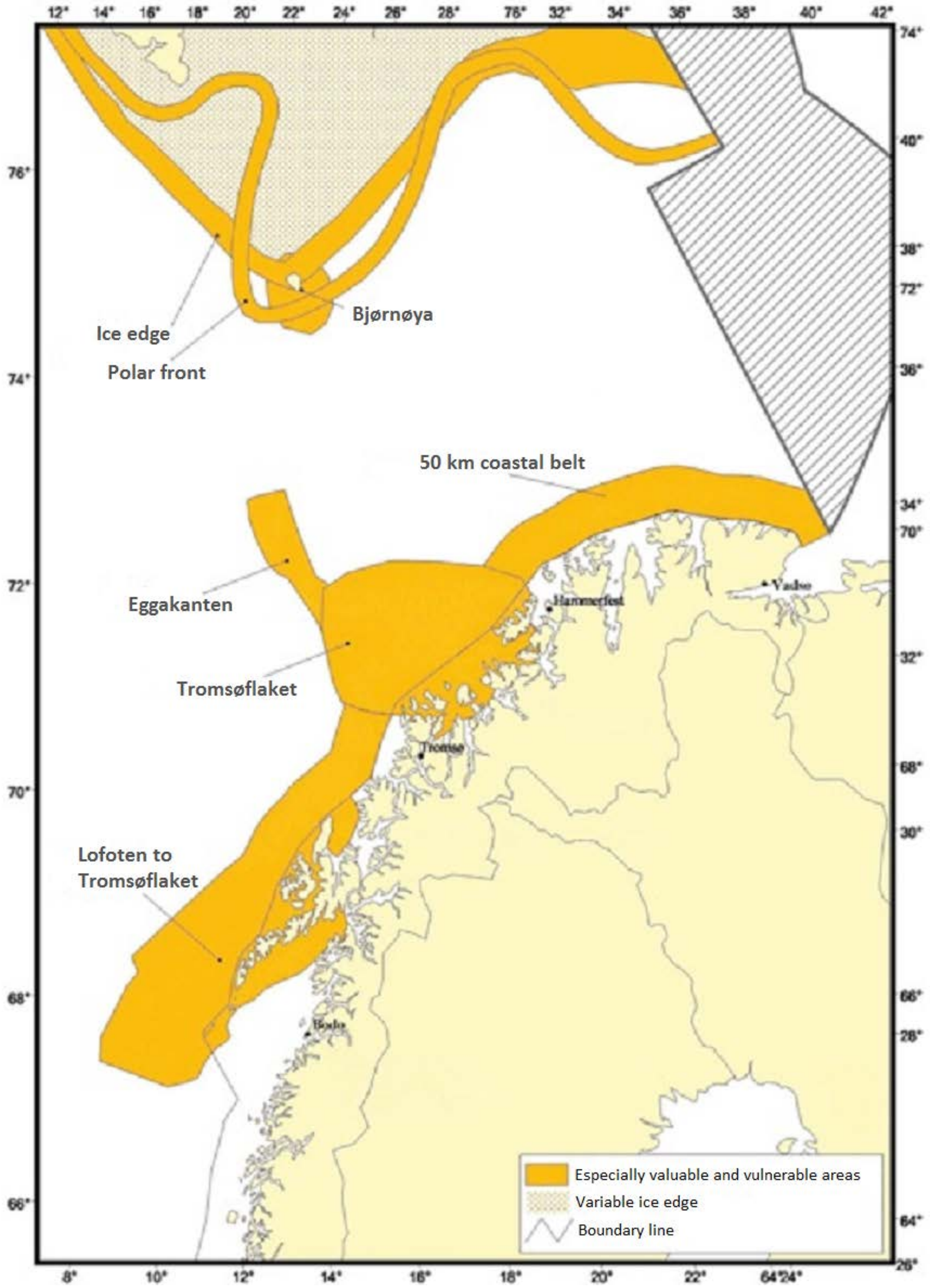

Figure 1. Map over Norwegian coast showing especially valuable and vulnerable areas (10)

The selection of the release point for radionuclides (Figure 2) was incidental and certainly not selected on the basis that the probability of an accident occurring at the selected point was any greater than that for any other point along the Norwegian coast. The sole criterion used was that the point of release would be located 'up-stream' of environmentally vulnerable and important commercial fishing areas along the coast. A selection of a release point in the Skagerrak was thus suitable in the sense that a large proportion of the radionuclides released would be expected to be transported via the Norwegian Coastal Current (see Figure 3) past important fish spawning areas on the south coast of Norway and subsequently past the important vulnerable environments and fisheries off the coast of Lofoten.

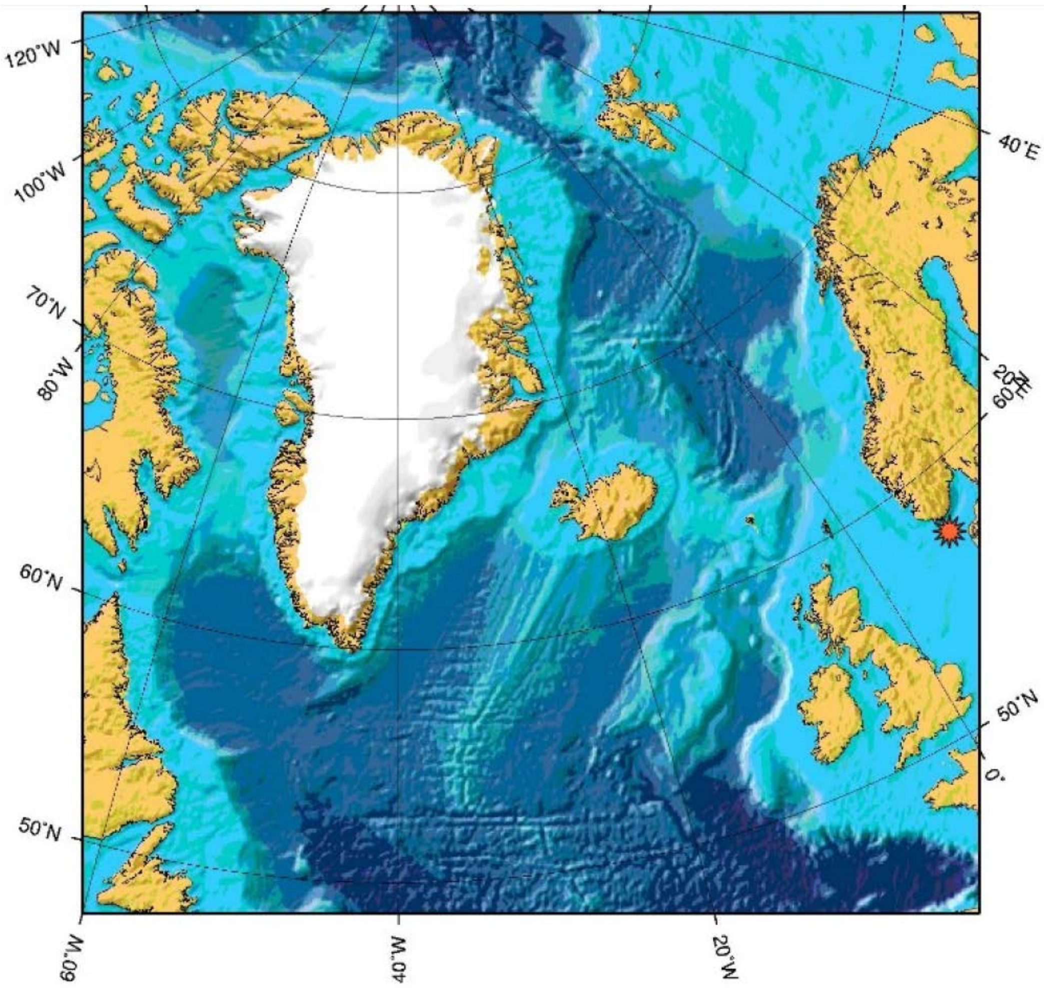

Figure 2. Release point for the model runs: Orange star: Skagerrak, South of Norway, (source of the map: DMI).

A release was set up for a non-specific, fully soluble radionuclide. Radioactive decay was assumed to be insignificant over the period of the experiment. A unit of contaminant was released over a time span of one day into a model grid box (horizontal size: 28 km x 28 km), which allowed for subsequent scaling according to different release cases.

Figure 3. The general circulation pattern of marine water in the North Sea (from (11)).

2.3 Modelling advection and dispersion of radionuclides

The simulation of radionuclide advection and dispersion has been achieved by using a large scale numerical model called NAOSIM (North Atlantic/Arctic coupled Ocean Sea Ice Model). A version of NAOSIM, which covers the Northern Seas plus the central Arctic and northern North Atlantic, was used. The use of this large simulation area ensures a proper representation of the system to ensure a correct embedding into the important large-scale circulation.

NAOSIM has been used successfully in a number of applications focusing on Northern Sea circulation (12, 13, 14) and tracer dispersion (15, 16). For the scenario experiments NAOSIM was forced with daily atmospheric (at 2-meters) air temperatures, surface wind stress, 2-meter dew-point temperature, cloudiness and precipitation derived from the NCEP-NCAR reanalysis for each year from 1948 to 2011. By this approach, a database has been constituted which allows statistical analyses of the data and the use of data probabilistically based on several decades of weather situations and oceanic circulation.

The aim was to derive probability density functions (PDFs), reflecting local as well as non-local variability of the oceanic current fields in the respective regions on daily to inter-decadal timescales. For this purpose, each release scenario was repeated at different starting times. For the period 1952 to 2007 every year the radionuclide was released into the top 20 m of the water column on the 1. January in the Skagerrak. To cover the subsequent dispersion of the radionuclide in Norwegian and adjacent waters each experiment (i.e. each ensemble member) was run over a period of 4 years. This produces a total of 224 years (56 experiments of 4 years length each, constituting a so-called ensemble of 56 members) of three dimensional concentration data for the entire model domain. The data are stored as weekly means. The first model run was from 1952 to 1956, the second 1953-1957 up to and end date of 2011 (starting 2007). The ensemble reflects the different states of the flow field over the last decades. From the ensemble members, PDFs for the simulated concentrations of the contaminant can be generated at every grid point in the model domain.

The scenario with a release in the Skagerrak was conceptualized as a hypothetical accident of a ship carrying nuclear material or a hypothetical accident of a nuclear-driven vessel south of the southern Norwegian coastline involving the release of soluble radionuclides into surface waters.

2.4 Statistical treatment of the datasets

For each grid point and depth (in 20 m intervals for the upper 100 m of the water column) we have information for 54 model runs constituting the ensemble. For a given grid point and single model run, time dependent concentrations for each depth are selected. The depth averaged concentration at a single position (grid box) is calculated for the entire (simulation) time period. For the following analysis the maximum depth-averaged concentration is selected at the given point.

As the observed variation between depth averaged data for all the considered grid-points was not significant (min and max differed by a factor of 2) a decision was made to pool all the available data for the considered region together. In this way we get 1960 (56 x 35) data points to work with. In order to find out which probability density function (PDF) fit the data better we first applied a default method of fitting distribution parameters based on the maximum likelihood method. In this method, the values of the parameters of a distribution are taken to be those that maximize the likelihood function.

Then, after considering various PDFs a criterion was needed to rank the considered distributions according to their goodness-of-fit. For this purpose, the Kolmogorov-Smirnov (KS) test statistic was applied to each PDF. The KS test statistic is defined as the maximum deviation between the hypothesized cumulative distribution function and the empirical cumulative density function and is a measure of the discrepancy of the tested PDF and the data.

Through this approach it was observed that a log-normal distribution (mean = 2.70E-20 and SD = 1.04E-20) fit the data much better than a normal distribution. This is shown in an illustrative way in Figure 4.

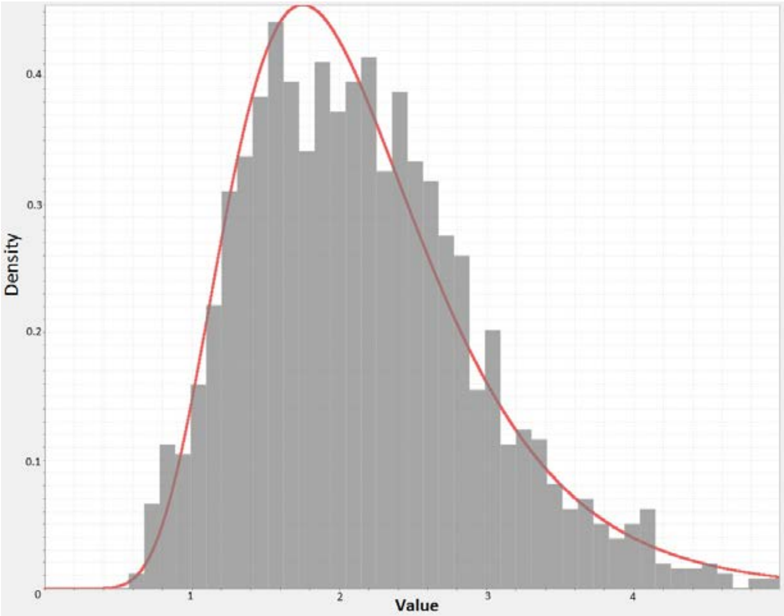

Figure 4. Fitting a log-normal distribution to data (numbers used in the figure are for illustrative purposes only and do not pertain to the actual values used in the analysis).

2.5 Quantifying risk to humans

For the human assessment, the critical pathway has been identified as arising from the consumption of seafood only, as opposed to external exposure from contaminated inter-tidal sediments or a combination of pathways. Consumption rate data are required and are normally selected to characterise groups of high-rate consumers. One source of data is the European Food Safety Authority (17) which provides statistical information on ingestion rates of different food products for many countries across Europe. For Norway, the daily ingestion of 'Fish and fish products' and 'Fish based preparations' amounts to 78.2 g (99^{th} percentile) or 28.6 kg per year. This value compares to a value of 93 kg per year of Plaice and cod for a 'critical' (High rate consumption) group used in earlier studies for impact studies in the UK. Although there are reservations regarding the 28.6 kg/y because the values seems rather low for a high rate consumer, the value has been nonetheless selected for subsequent calculations because it is specific to Norway and comprises of a quite recent (less than 10 years old) analysis.

Since the data provided from model simulations are in terms of Bq per unit volume seawater, there is a requirement to convert these values to an activity concentration in the seafood product being studied. The standard dataset used in routine human radiological assessments from IAEA Technical report series (TRS) 422 (18) which employs concentrations factors[1], CFs. The approach however has its limitations. Although in no way a justification of adopting steady state transfer parameters in this work, the recent study (8) also adopted an approach using CFs.

No information on the underlying variability in CFs is provided in TRS-422 in a quantitative sense but it is stated in the report that, with regard to these transfer parameters, *"In most cases maximum and minimum values can be assumed to be within one order of magnitude of the recommended value".* In view of the lack of specific information on pdfs a decision was made to include variability by employing a uniform distribution with a minimum and maximum one order of magnitude below and above the 'recommended' value respectively in line with the information available. In this way the uncertainty associated with the transfer component of the risk

[1] Defined as the activity concentration in biota (Bq/kg f.w.) divided by activity concentration in sea water (Bq/l).

assessment, which is believe to be an important part of the assessment in terms of its potential for introducing uncertainty (see (19)), can be accounted for through error propagation when running probabilistic analyses.

The (Annual) Committed effective dose, E_{int}, can be derived using the following equation:

$$E_{\text{int}} = \sum_i e(\tau)_i \times I_i \qquad (1)$$

Where : $e(\tau)$ = Dose conversion factor (Sv Bq^{-1}) for radionuclide "i"

I_i = Annual Intake of radionuclide "i", (Bq y^{-1})

Dose conversion factors have been taken from ICRP-72 (20).

The final piece of information required for the analysis is the appropriate benchmark with which to compare the doses calculated. The IAEA (21) have suggested a de minimis level of 10 µSv y^{-1} to any member of the public below which a practice may be exempted from further consideration. This was also considered to be the most appropriate benchmark to use in the context of radiological assessments involving a screening approach that was recently applied (22).

The (Radionuclide specific) Human health Risk Index, RI_{hum}, is thus defined by Equation 2 :

$$RI_{hum} = \frac{E_{\text{int}}}{D_{\text{de minimis}}} \, per\, Bq\, release \qquad (2)$$

Where : E_{int} = (Annual) Committed effective dose (µSv)

$D_{\text{de minimis}}$ = (Annual) benchmark de minimis dose (µSv)

Since the dose is expressed as a pdf, the Risk Index also has an associated distribution. A 95th percentile has been selected to introduce conservatism into the dose and hence the risk index metric to allow for the uncertainty in applying an equilibrium-based model to an arguably incompatible dynamic system.

Tables were generated (see Results) for those radionuclides that might be expected to constitute the largest components of the release for the type of accident scenario involving nuclear submarines.

The selection was made based on a recent comprehensive analyses (23). From information provided for radionuclide inventories of nuclear submarines, such as Komsolmolets, that lie on the seabed and what is known about release fractions in relation to potential scenarios that could occur for reactors under particular conditions it would appear that focus should be placed on deriving information for radioisotopes of caesium and Sr-90. The inventories and or release fractions of other radionuclides are typically lower than the aforementioned radionuclides and where this is not necessarily the case, for example in releases involving isotopes of iodine, the half-lives of the various radioisotopes considered, I-131 (T0.5 = 8.02 d), I-132 (T0.5 = 2.295 h), I-133 (T0.5 = 20.8 h), are so short that substantial decay would have occurred before the contamination reached the measurement location.

Decay has been accounted for in the case of Cs-134 only because the half-life was such that the approximate transit time from source to assessment area of approximately 75 weeks was long enough that decay would have occurred to a non-trivial degree.

The applicability of the approach varies according to which radionuclide we are interested in.

The conservative behavior of radiocaesium in seawater (see (24)) means that the model outputs from NAOSIM that pertain to a passive tracer are highly relevant. Similar arguments hold true for ^{90}Sr.

2.6 Quantifying risk to the environment

The ERICA integrated approach (25) is a means of quantifying risk to the environment from ionising radiation. The approach is based around the use of reference organisms, selected as being representative of terrestrial, freshwater and marine ecosystems and encapsulating information on transfer, dosimetry (through defined geometric shapes) and effects data that are specific to these organism groups. The approach is supported by the ERICA Tool (26) which consists of software that connects the various underlying databases that are required to perform an assessment allowing ease of manipulation by an assessor through an easy-to-use graphical user interface.

In a similar fashion to the analysis conducted for humans, use is made of concentration ratios in the derivation of activity concentrations of radionuclides in organisms from activity concentrations in seawater. The same reservations in using these ratios, which assume steady state conditions, of course apply in this case. Use has been made of up-to-date information from the wild-life transfer database (see (27)). It should be noted that although there is some overlap in the transfer data used for humans and those for the environmental impact assessment, in the sense that some of the same published datasets have been used, there are important differences primarily relating to the selection of species for inclusion (the data compilation for humans focuses on edible species whereas the environmental assessment does not) and the part of the organism body to which the values pertain (edible part for the human assessment compared to the whole body for 'wild' organisms). The underlying databases used for the non-human biota assessment include detailed information on statistics, in contrast to the data compilation for humans, and therefore allows us to make use of this information when undertaking error propagation and probabilistic calculations.

One additional piece of information that is required for the environmental assessment relates to the physical dimensions of the selected organism and its associated occupancy factors, i.e. position (and fraction of time spent) within a given habitat that a selected organism can be found. These are important considerations in relation to the internal and external dose calculations that are performed. The starting point for this analyses was the map over Norwegian coast showing especially valuable and vulnerable areas (Figure 1). Information was then accessed (see (28)) providing details on the life history for various fish species common to Norwegian coastal-marine environments. The information included spawning areas for these fish (Figure 5), and where these coincided, or at least showed partial overlap, with the area delineated within Figure 1 these species were selected for further potential consideration. For the sake of efficiency (because there seemed little point in exhaustively reporting information for every single fish species where the above criterion were met because the differences introduced by slightly different geometries, reflecting size and mass, will introduce only small changes to the calculated dose rates), 1 pelagic and 1 demersal and 1 benthic species were finally selected to report upon in this work. The selected species were Greenland Halibut (*Hippoglossus hippoglossus*), Herring (*Clupea harengus*) and Cod (*Gadus morhua*).

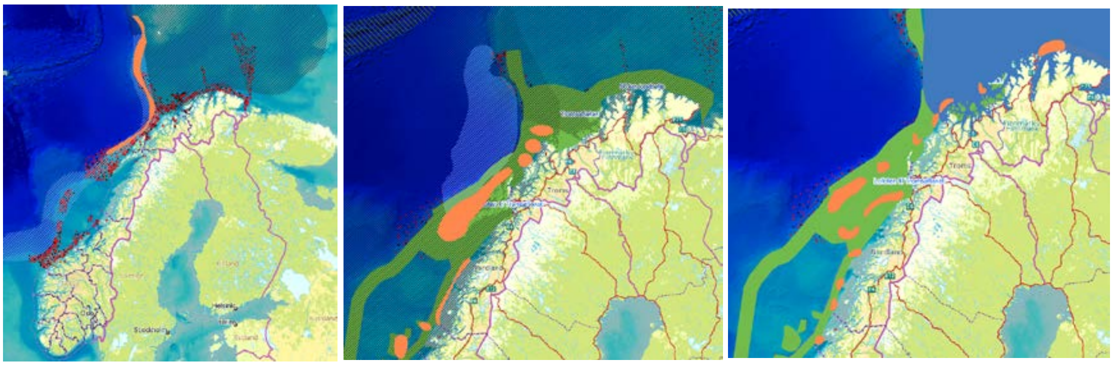

Figure 5. Spawning area (marked with light brown color) for Greenland Halibut, Herring and Cod (from left to right).

From life history information, appropriate geometries and masses of these organisms were generated by selecting appropriate elliptical representations. Following this, dose conversion coefficients could be calculated using functionality in the ERICA Tool (26). The particular information used is presented in Table 1.

Table 1 : information used to create geometries for fish species and concomitant occupancy factors

Organism (habitat)	**Mass**	**Occupancy factor**	**Relative Dimensions of equivalent ellipsoid**
Greenland Halibut (Benthic)	44 kg[1]	Sediment water interface =1	1 x0.6250 x 0.0630[1]
Herring (Pelagic)	0.5 kg	Water column	1 x 0.1875 x 0.1875
Cod (Demersal)	55 kg	Sediment water interface = 0.5; Water column = 0.5	1 x 0.24 x 0.24

[1] Relative dimensions based on ICRP flatfish (7).

Details for other organisms were more generic in nature, i.e. specific life history data were not compiled nor were dosimetric geometries generated placing reliance instead upon default dose conversion factors from the ERICA Tool. It was considered appropriate to include a marine mammal and a sea bird in the analyses. Information concerning the radiobiology of different organism groupings suggests that vertebrates, such as mammals and birds, are among the most radiosensitive of all organism groups (29). Furthermore, the 'Lofoten to Tromsøflaket' is considered to be an important habitat for numerous species of seabirds and mammals.

Total (internal plus external irradiation) weighted (to account for radiation quality) absorbed dose rates can be derived through application of the ERICA Tool using information entered as single numbers or as probability distribution functions, pdfs.

In conformity with the human assessment, there was a prerequisite for the environmental assessment to select an appropriate benchmark against which to compare the doses calculated. Within ERICA, "effective" dose rates resulting in a 10% change in observed effect (EDR_{10}) data for mortality, morbidity and reproduction endpoints for a range of organism types (plants, vertebrates and invertebrates) for three ecosystems have been collated and data have been screened to remove results that do not meet with strict quality assurance criteria (30). From this information, species sensitivity distributions are constructed by plotting remaining robust EDR_{10} data onto a single figure allowing the hazardous dose-rate 5 %, HDR_5 i.e. the dose rate at which you might expect to observe a 10 % effect in 5 % of species). A 'Safety Factor' of 5 is finally applied to account for uncertainties associated with the dose-effects data, *per se*, and extrapolations (from groups of individuals to populations, from external irradiation effects data to multiple irradiation pathways and from laboratory to field etc.) to derive a Predicted No Effects Dose-rate, (PNEDR). At the ecosystem level, the no-effect values lie in the dose range giving rise to minor cytogenetic effects or minor effects on morbidity in vertebrates. Those effects are not expected to be directly relevant at higher organisational levels, such as the structure and functioning of ecosystems

The PNEDR used by ERICA is 10µGy/h. This has been used as the appropriate benchmark around which to base the radioecological Environmental Risk Index.

The (Radionuclide specific) Environmental Risk Index, RI_{env}, is defined by Equation 3 :

$$RI_{env} = \frac{D_o}{PNEDR}\ per\ Bq\ release \qquad (3)$$

Where : D_o = Dose rate for organism 'o' (µGy/h)

And PNEDR = predicted no effects dose rate (µGy/h)

The dose rate will have additional variability introduced from parameters such as CR values (the ERICA Tool has distributions defined for example for Cs-137 fish CRs).

Since the dose rate is expressed as a pdf, the Risk Index also has an associated distribution. A 95th percentile has been selected to introduce conservatism into the dose rate and hence the risk index metric to allow for the uncertainty in applying an equilibrium based model to an arguably incompatible dynamic system.

As for the human risk assessment, a table was generated for those radionuclides that might be expected to constitute the largest components of the release for the type of accident scenario involving nuclear submarines.

2.7 Case study – application of the approach to a theoretical scenario

In the following section, a demonstration is provided on how the methodology presented above might be used in practice.

For a given release of 'x' Bq of a particular radionuclide we simply multiply the Risk Index for that radionuclide by 'x'. The Risk Index relates to a particular radionuclide and therefore the radionuclide specific Risk Index needs to be summed over all radionuclides being considered in the assessment. For those radionuclides not considered in the tables provided in this report specific derivations may be required by applying the ERICA Tool.

If the sensitivity exceeds 1 then there is a potential for impact on marine wildlife and a more detailed assessment would be required reflecting the fact that the dose rate criteria are for screening assessments.

3 Results and discussion

3.1 Variability in seawater activity concentrations with time

An example for the available information from the set of ensemble experiments is illustrated in Figure 6. In this figure, the surface concentrations in the Norwegian Coastal Current off Tromsø for the 4 years subsequent to the 1-day release on the southern Norwegian coast are shown. It is evident that the shapes of lines characterizing concentrations versus time are very different for the different starting years of release.

There are also differences in the maximum concentrations observed (cf. high maximum transfers close to 3E-14 with low maxima below 1.5E-14), as well as the duration of the period with the largest concentrations and the arrival time of the contamination front. For instance, the arrival time of the peak concentration varies roughly between week 50 and week 100. This is an important result as details concerning the arrival time of a contamination peak would be an obvious piece of information requested post-accident and the uncertainty associated with any prognoses given should be apparent. Although any given Emergency Crisis Committee is unlikely to enthuse over the news that the timing of maximum concentrations, based on knowledge a priori, cannot be specified with accuracy beyond a 1 year interval, for a case like the one chosen here, the data provides a true reflection of the variability in the system and as such needs to be accounted for.

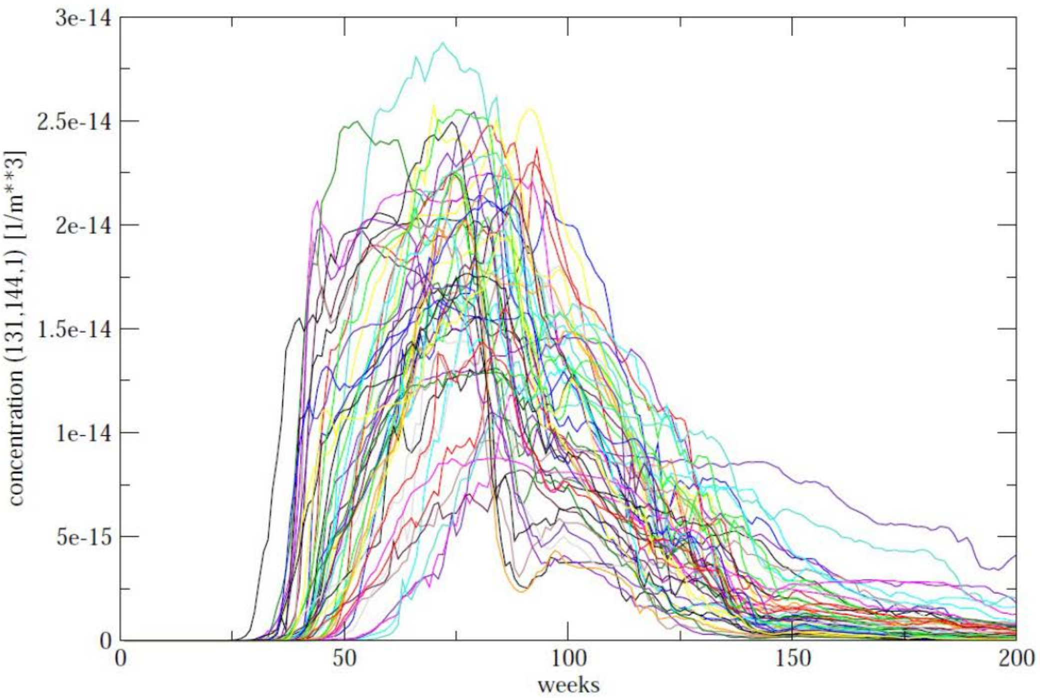

Figure 6. Concentrations of the radionuclide released in the Skagerrak at a surface position in the Norwegian Coast Current close to Tromsø for the full ensemble of 56 model experiments (weekly means, time axes provides week after release, each color represents a different starting year of release).

3.2 Transfer Factors

In previous work (31, 32) involving conservative radionuclides (or tracers) from known releases and measurements in seawater), "transfer factors" were estimated from the Sellafield reprocessing complex to coastal locations in Northern Norway (Tromsø coast, Barents Sea). The transfer factors from these publications were in the range 5-50 Bq per m^{-3} per PBq a^{-1} released i.e. 5E-15 to 5 E-14 Bq per m^{-3} per Bq a^{-1} released. The presumption, made in the current report but not by the aforementioned authors, would be that if Sellafield transfer factors were related to an instantaneous release and not one occurring over 1 year, the transfer factors would be somewhat greater than the values provided above. The argument is based on the consideration that for a pulsed release, the maximum values attained at a position downstream would attain higher values than those associated with a continuous release because for the latter the activity is already, at the initial stage of release, spread out over time (in effect adding an extra level of dispersion). The corresponding value for the current work based on a release from the Skagerrak is 27 ± 10 Bq per m^{-3} per PBq released. This value is of the same order of magnitude as those cited above from previous studies notwithstanding the view that direct comparison is difficult because the derived transfer factors are for different source locations. Arguably the values derived from the current modelling work (i.e. with release in the Skagerrak) is somewhat lower than might be expected if the empirical information cited above is taken as the 'true' indication of physical transfer because the release point in the model was located much closer to the assessment area than those for the empirical studies and would thus be expected to yield higher activity concentrations downstream. This discrepancy may have numerous explanations but a consideration that becomes immediately evident relates to the fact that linking measurements of activity concentrations at some distance from a source to the amount of activity released at a source is fraught with difficulty if releases have been occurring over protracted periods and where other sources may confound the input signal. These conditions would appear to have prevailed with some of the earlier studies on transfer from Sellafield (31) where releases(of e.g. radiocaesium) had been occurring for some time before the transit factors were derived (thus increasing the likelihood of contamination recirculation in the system and introducing what might be referred to as secondary sources of contamination). Furthermore, the inputs of other sources including global fallout etc. cannot be discounted easily nor can the possibility that the maximum in environmental concentrations is missed during empirical studies as the frequency of sampling is often not very high. Where studies have been conducted accounting for 'background' seawater concentrations of conservative radionuclides and where the input signal of the tracer to the system is more clearly defined (e.g. Sellafield ^{99}Tc discharges in the 1990s where an order of magnitude or so abrupt, or step, increase in discharges over historical levels was observed), the calculated transfer factors (from Sellafield to Tromsø) lie very much towards the lower end of the range expressed above (from (31, 32)) of ca. 5 Bq per m^{-3} per PBq a^{-1} (see (32)). Such a transfer factor would appear to be much more consistent with the modelled data from inputs to the Skagerrak in the sense that it lies substantially below the latter simulated result.

3.3 Limitations in coupling 3d hydrodynamic models to exposure models

Spatial averaging is an issue that is often raised in the processes of undertaking environmental impact assessments. The grid points are approximately 30 km apart (the grid is resolved to 28 km by 28 km) in a horizontal direction. For mobile organisms, such as migratory fish, there seems little point in emphasizing any difference between *metrics* in adjacent grid points because the 'spatial averaging' resulting from radionuclide uptake over the much larger areas associated with the organisms movement will probably render this unimportant. For more sedentary organisms however an argument can be made for distinguishing between grid-points although the fact that seawater concentrations at any given point for any given release are likely to change quite rapidly renders elaborated discussions on spatial differences less elucidating.

The tacit assumption in using CFs is that the system is under steady state, a condition that clearly may not be applicable when environmental concentrations are changing rapidly with time and in space following an accidental, pulsed release of radioactivity. Although other types of approaches have been adopted under such conditions, such as kinetic models applied post-accident at Fukushima Dai-ichi in 2011 (33) in deriving marine biota activity concentrations, a similar way forward was not deemed necessary here. This primarily reflected two further considerations that (i) the input data (activity concentrations with time at a given location) would need to be deterministic, providing no information on underlying variability, thus defeating the objective of the study and (ii) we would need to expand the data-set so that a time series was available for each grid point an approach that (although such data were available) was deemed practically challenging at this stage of the analysis. For this reason, the 'simplification' has been made to use CFs.

A further point relates to the consideration of benthic organisms. In calculated external exposure, distribution coefficients, k_ds, are applied in deriving sediment concentrations from seawater concentrations for application in screening tools like ERICA (26). For more realistic modeling, the assumption of instantaneous equilibrium between seawater and deposited sediment underpinning a robust application of k_ds is less appropriate. The rate functions dictating the flux of radionuclides between the water column and the seabed are presumably not constant reflecting the rapid passage of a contaminated plume in the water column above bed sediments where the possibility for interaction is limited.

3.4 Risk to humans and the environment

The risk indices pertaining to human welfare for selected radionuclides are presented in Table 2.

Table 2 : (Human) Risk indices (as defined in Equation 2) for "Lofoten til Tromsøflaket" in relation to particular radionuclides and releases in the Skagerrak

Radioisotope	**RI_{hum}** (deterministically derived values in brackets)*
Cs-134	**1.0 E-15** (8.7E-17)
Cs-137	**1.2 E-15** (1.0E-16)
Sr-90	**7.7 E-17** (6.5E-18)

*Values not derived using underlying uncertainty in parameters

The Risk indices relate to a unit release, i.e. 1 Bq, in the Skagerrak and thus the actual (or hypothetical) risk is derived by multiplying by the actual (or hypothetical) release at this location. The calculated RI_{hum} can be used to identify a potential problem if the index exceeds unity. In other words a single release in excess of 1 PBq (1 E+15 Bq) of ^{134}Cs would constitute a threshold beyond which further action, which might involve something as rudimentary as performing more detailed analyses, would be warranted.

An important consideration relates to the comparison of the conservative values provided in bold font in Table 2, which pertain to conservative estimates that account for variability in the marine system, and the values in parentheses, which pertain to deterministic calculations using best estimate values (and which might have been derived had little attention been paid to the variations in activity concentrations that could occur in seawater and ignoring uncertainty in the key parameters relating to transfer). The conservative estimates are at least one order of magnitude greater than those determined deterministically which, although certainly not of a magnitude that might be considered extreme, leads us to support the notion that understanding, or at least making efforts to characterize, the underlying variability and uncertainty within environmental systems should constitute an important prerequisite for subsequent robust defendable assessments. Without making such analyses, substantial underestimates in risk could occur. Furthermore, using long term averages to characterize the currents and circulation leading to the advection and dispersion of contaminants or arbitrarily selecting a given flow regime might lead to misleading results in relation to the determination of risk.

The risk indices pertaining to environmental protection for selected radionuclides are presented in Table 3.

Table 3 : (Environmental) Risk indices (as defined in Equation 3) for "Lofoten to Tromsøflaket" in relation to particular radionuclides and releases in the Skagerrak

Radioisotope	**RI_{env}** (median)*	**Limiting biota**
Cs-134	**2.3E-17** (2.1E-18)	Greenland Halibut
Cs-137	**1.3E-17** (1.2E-18)	Greenland Halibut
Sr-90	**1.1E-18** (1.1E-19)	Bird

* in this case, in contrast to the case for humans, it was defendable to present a median as oppose to 'best estimate' value because detailed statistical information was used in the probabilistic simulation (whereas this wasn't the case for the human assessment).

The limiting biota reported in the table above concern those organisms for which exposure and therefore risk were calculated to be greatest among the full suite of organisms included in the analysis as described in Section 2.6 above. The identification of Greenland Halibut as the 'limiting biota' for releases of radiocaesium isotopes undoubtedly reflect the position of this organism type at the seabed and thus the relative importance of an external exposure pathway from sediment. The Risk indices for isotopes of radiocaesium that were derived for herring a pelagic species, for example, were substantially lower (cf. 95% 1.4E-19 with 1.3E-17 for Cs-137) than halibut whereas cod, located nearer the sediment, Risk indices were only a factor of ca. two-three lower (cf. 95% 5E-18 with 1.3E-17 for Cs-137). Nonetheless, some caution is required in viewing these results because of a heavy reliance on K_ds when deriving sediment activity concentrations and the limitation in applying such ratios, as described above, to a dynamic situation. If anything, the RI_{env} might be considered as being overly pessimistic because the likelihood is that a system would not have equilibrated and the contamination levels in sediments would not have had time to reach levels that are normally expressed through the application of K_ds.

In a similar way to the considerations made for the human risk indices above, it is apparent that one would require a release exceeding 70 PBq of ^{134}Cs in the Skagerrak before serious concerns based on radiological criteria for the Lofoten/ Tromsøflaket would be raised.

For this particular case and accepting the specific radiological criteria adopted in this report, it would seem quite evident that considerations with regard to potential human radiological impacts would drive management decisions related to a marine release.

This specific view would clearly hold in a more generic sense for all cases where an environment is exploited for human use and where the assessment area is co-located for humans and wildlife because the only parameter that can change in the proposed approach is the amount of activity released and this will of course be identical for both components of the assessment. Nonetheless the environmental risk criteria still allow valuable information to be conveyed in the event that limitations are put in place with regard to human resources exploitation – it might be foreseen, for example, that restrictions could be placed on fishing and the consumption of seafoods following an accident. In this regard the use of an RI_{env} could be used to contextualize or interpret the meaning of a given release in line with the intentions expressed by the ICRP (7). Furthermore, there is a clear sensitivity to the choice of benchmark. The application of a De Mininis level for humans is not universally approved and the selection of alternative metrics to scale the potential harm to humans, for example the application of dose limits and reference levels (see (9)), might render the putative predominance of human radiological criteria less evident. Finally, it may not be appropriate under particular cases to co-locate the environmental and human assessment areas (22) a consideration that might clearly lead to differences from the example given above, in terms of which radiological criteria drive management decisions.

3.5 Case study

The inventories of Komsomolets and K-159 have previously been estimated (8) as approximately 3.1 E15 Bq ^{137}Cs (as of 1989) for the former and 1.2E15 to 5.2E15 Bq ^{137}Cs (with a similar inventory of ^{90}Sr) for the latter in the process of deriving input estimates for their studies concerning releases to and 'impacts' upon Norwegian coastal environments. Although these input estimates are highly conservative because no consideration was given to release fractions as notably analyses in other related studies (23), they suffice for the purposes here which is simply to illustrate the application of the methodology without dwelling too specifically on the repercussions following a severe nuclear accident in the Skagerrak. Using the inventories given then, the RI_{hum} would be exceeded by some margin – the releases of Cs-137 alone would be enough to trigger a response to consider some form of action which might, for example, involve introducing restrictions on seafood sale and consumption. However, the derived risk to the environment would be substantially below a level where any realistic concerns might be justified. Had account not be taken of the uncertainty in the assessment endpoints a quite different conclusion may have been drawn in the sense that neither the ΣRI_{env} nor the ΣRI_{hum} would have exceeded unity. This once more draws attention to the essential requirement of having an understanding of variability and of accounting for it when conducting this type of assessment.

4 Conclusions

Understanding, or at least attempting to quantify, the variability associated with activity concentrations in seawater, which might be expected from a nuclear accident involving the release of radioactivity upstream from vulnerable locations, is arguably an important component of robust emergency preparedness planning for the marine environment. One way of achieving this goal is through the application of appropriate modelling tools. In this report the NAOSIM model has been applied to model the advection and dispersion of conservative tracers and provide quantitative information on radionuclide concentrations in seawater over a range of different weather conditions and oceanic circulatory patterns (corresponding to conditions observed in previous years). The resultant probabilistic data have, in turn, been used as input to transfer and dose models to provide an indication of potential impacts for man and the environment. The concomitant radionuclide specific risk indices presented in this report can be applied by simply multiplying the reported values by the magnitude of the source term and thereafter summing over all radionuclides to provide an indication of total risk. For the particular case study presented, for which the locations selected for environmental impacts were co-located with the fisheries used in the human assessment, it would appear that considerations with regard to potential human radiological impacts would drive management decisions related to a marine release. However, if different radiological protection criteria had been applied or if dissimilar locations corresponding to human and environmental impacts had been selected, it is evident that environmental criteria could become more prominent drivers.

5 References

1. AMAP, 2010. AMAP (2010). Arctic Monitoring and Assessment Programme, AMAP Assessment 2009: Radioactivity in the Arctic. (AMAP publication 978-82-7971-059-2, 2010; http://www.amap.no/documents/doc/amap-assessment-2009-radioactivity-in-the-arctic/99).

2. Høibråten et al., 1997. Høibråten, S., Thoresen, P. E., Haugan A. (1997). The sunken nuclear submarine Komsomolets and its effects on the environment. Science of The Total EnvironmentVolume 202, Issues 1–3, 25 pp. 67–78.

3. Amundsen et al., 2002. Amundsen, I., M. Iosjpe, O. Reistad, B. Lind, K. Gussgaard, et al. The accidental sinking of the nuclear submarine, the Kursk: monitoring of radioactivity and the preliminary assessment of the potential impact of radioactive releases. Mar. Pollut. Bull. 44, 459–468 (2002).

4. IAEA, 2001 IAEA (2001). International Atomic Energy Agency, "Inventory of accidents and losses at sea involving radioactive material" (IAEA-TECDOC-1242, 2001; http://www pub.iaea.org/books/IAEABooks/6353/Inventory-of-Accidents-and-Losses-at-Sea-Involving-Radioactive-Material.

5. Olsen et al., 2010. Olsen, E., S. Aanes, S. Mehl, J. C. Holst, A. Aglen, et al., (2001). Cod, haddock, saithe, herring and capelin in the Barents Sea and adjacent waters: a review of the biological value of the area, ICES J. Mar. Sci. 67, 87–101.

6. IMR, 2013. IMR (2013). Institute for Marine Research, "KunnskapsInnhenting Barentshavet – Lofoten" (KILO Rapporten nr. 3-2013, 2013; http://brage.bibsys.no/xmlui/handle/11250/113923)

7. ICRP, 2014. ICRP (2014). Protection of the environment under different exposure situations. ICRP Publication 124, Ann. ICRP 43(1).'

8. Heldal et al. (2013). Heldal, H E, Vikebø, F., Johansen G. O. (Dispersal of the radionuclide caesium-137 (137Cs) from point sources in the Barents and Norwegian Seas and its potential contamination of the Arctic marine food chain: Coupling numerical ocean models with geographical fish distribution data. Environmental Pollution 180 (2013) 190-198.

9. ICRP, 2007. ICRP, 2007. The 2007 Recommendations of the International Commission on Radiological Protection. ICRP Publication 103. Ann. ICRP 37(2–4).

10. St.meld. nr. 8, 2005-2006. St.meld. nr. 8 (2005-2006) Helhetlig forvaltning av det marine miljø i Barentshavet og havområdene utenfor Lofoten (forvaltningsplan).

https://www.regjeringen.no/nb/dokumenter/stmeld-nr-8-2005-2006-/id199809/

Accessed : March 04, 2015.

11. Brown et al., 1999. Brown, J. E., Kostad, A. K., Bringot, A. L., Lind, B., Rudjord, A. L., & Strand, P. (1999). Levels of ^{99}Tc in biota and seawater samples from Norwegian coastal waters and adjacent sea. Marine Pollution Bulletin, 38(7), 560–571.

12. Karcher et al. 2008. Karcher, M, Gerdes, R, and Kauker, F, 2008. Long-term variability of Atlantic water inflow to the Northern Seas: insights from model experiments In: Arctic-Subarctic Ocean Fluxes: Defining the role of the Northern Seas in Climate, eds: B. Dickson, J. Meincke and P. Rhines. Springer, 111-130.

13. Gerdes et al., 2001. Gerdes R., Karcher M., Kauker F., Köberle C. (2001). Predicting the spread of radioactive substances from the Kursk. Eos, Transactions American Geophysical Union, Volume 82, Issue 23, pages 253–257.

14. Gerdes et al., 2005. Gerdes, R, Drange, H, Gao, Y, Karcher, MJ, Kauker, F, and Bentsen, M, 2005. Ocean General Circulation Modelling of the Nordic Seas , in: The Nordic Seas: An Integrated Perspective, Drange, Furevik, Gerdes and Berger (Eds.), AGU Monograph edition). 199-220.

15. Karcher et al., 2004. Karcher, MJ, Gerland, S, Harms, IH, Iosjpe, M, Heldal, HE, Kershaw, PJ, and Sickel, M, 2004. The dispersion of Technetium-99 in the Nordic Seas and the Arctic Ocean: a comparison of model results and observations J. Env. Rad. 74(1-3):185-198.

16. Karcher et al., 2012. Karcher, M., et al. "Recent changes in Arctic Ocean circulation revealed by iodine-129 observations and modeling." *Journal of Geophysical Research: Oceans (1978–2012)* 117.C8 (2012).

17. EFSA, 2014. [EFSA], European Food Safety Authority. 2014. The EFSA Comprehensive EuropeanFood Consumption Database. [cited October 2014]. Available from: http://www.efsa.europa.eu/en/datexfoodcdb/datexfooddb.htm

18. IAEA, 2004. IAEA, 2004. Sediment Distribution Coefficients and Concentration Factors for Biota in the Marine Environment. In: Technical Reports Series No. 422. International Atomic Energy Agency, Vienna.

19. Avila et al., 2004. Avila, R., Beresfold, N., Broed, R., Brown, J., Iospje, M., Agüero A., Robles, B., Suañez, A. (2004). Study of the uncertainty in estimation of the exposure of non-human biota to ionizing radiation. Journal of Radiological Protection, 24, pp. A105-A122.

20. ICRP, 1995. ICRP, 1995. Age-dependent Doses to the Members of the Public from Intake of Radionuclides - Part 5 Compilation of Ingestion and Inhalation Coefficients. ICRP Publication 72. Ann. ICRP 26 (1).

21. IAEA (1999). IAEA 1999 Application of radiological exclusion and exemption principles to sea disposal. The concept of 'de minimis' for radioactive substances under the London Convention 1972 IAEA Techdoc 1068 (Vienna: IAEA).

22. Copplestone et al., 2010. Copplestone, D., Brown, J.E. and Beresford N.A. (2010). Considerations for the integration of human and wildlife radiological assessments. J. Radiol. Prot., 30, 283–297.

23. Iosjpe et al., 2011. Iosjpe M., Reistad O., Liland A. (2011). Radioecological consequences after a hypothetical accident with release into the marine environment involving a Russian nuclear submarine in the Barents Sea. StrålevernRapport 2011:3. Østerås: Statens strålevern, 2011.

24. Jefferies et al., 1982. Jefferies, D. F., Steele, A. K., & Preston, A. (1982). Further studies on the distribution of Cs-137 in British coastal waters. I. - Irish Sea. Deep Sea Research, 29, 713–738.

25. Larsson, 2008. Larsson, C.M., 2008. An overview of the ERICA Integrated Approach to the assessment and management of environmental risks from ionising contaminants. Journal of Environmental Radioactivity 99, 1364-1370.

26. Brown et al., 2008. Brown, J.E., Alfonso, B., Avila, R., Beresford, N.A., Copplestone, D., Pröhl, G., Ulanovsky A. (2008). The ERICA Tool. Journal of Environmental Radioactivity 99, Issue 9, pp.1371-1383.

27. Howard et al., 2013. Howard, B. J.; Beresford, N. A.; Copplestone, D.; Telleria, D.; Proehl, G.; Fesenko, S.; Jeffree, R. A.; Yankovich, T. L.; Brown, J. E.; Higley, K.; Johansen, M. P.; Mulye, H.; Vandenhove, H.; Gashchak, S.; Wood, M. D.; Takata, H.; Andersson, P.; Dale, P.; Ryan, J.; Bollhofer, A.; Doering, C.; Barnett, C. L.; Wells, C., The IAEA handbook on radionuclide transfer to wildlife. J. Environ. Radioact. 2013, 121, 55-74.

28. Wienerroither et al., 2011. Wienerroither R., Johannesen E., Dolgov A., Byrkjedal I., Bjelland O, Drevetnyak K., Eriksen KB., Høines Å., Langhelle G., Langøy H., Prokhorova T., Prozorkevich D., Wenneck T., 2011. Atlas of the Barents Sea Fishes. IMR/PINRO Joint Report Series 1-2011, ISSN 1502-8828.

29. UNSCEAR, 2011. United Nations Scientific Committee on the Effects of Atomic Radiation. Sources and Effects of Ionizing radiation, 2008 Report to the General Assembly, with scientific annexes; Annex E: Effects of ionizing radiation on non-human biota; United Nations: New York, 2011.

30. Garnier Laplace et al. 2008. Garnier-Laplace, J., Copplestone, D., Gilbin, R., Alonzo, F., Ciffroy, P., Gilek, M., et al. (2008). Issues and practices in the use of effects data from FREDERICA in the ERICA Integrated Approach. Journal of Environmental Radioactivity, 99/9, 1474–1483.

31. Dahlgaard (1995). Dahlgaard, H. (1995). Transfer of European coastal pollution to the Arctic: radioactive tracers. Marine Pollution Bulletin, 31, 3–7.

32. Brown et al., (2002). Brown, J.E., Iospje, M., Kolstad, K.E., Lind, B., Rudjord, A.L. & Strand P. (2002). Temporal trends for ^{99}Tc in Norwegian coastal environments and spatial distribution in the Barents Sea. Journal of Environmental Radioactivity, 60, 49–60.

33. Vives I Battle et al. , 2014. J Vives i Batlle, T Aono, JE Brown, A Hosseini, J Garnier-Laplace et al. (2014). The impact of the Fukushima nuclear accident on marine biota: Retrospective assessment of the first year and perspectives. Science of The Total Environment 487, 143-153.

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ISSN 1891-5191 (online)
ISSN 0804-4910 (print)