StrålevernRapport ▪ 2011:3

Radioecological consequences after a hypothetical accident with release into the marine environment involving a Russian nuclear submarine in the Barents Sea

Statens strålevern
Norwegian Radiation Protection Authority

Reference:
Iosjpe M., Reistad O., Liland A.
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StrålevernRapport 2011:3. Østerås: Statens strålevern, 2011.

Key words:
Radioactivity, submarine, accident, cinsequences, concentration, doses, marine, biota

Abstract:
The report presents results concerning the potential consequences of a hypothetical accident involving a modern Russian submarine.

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Radioecological consequences after a hypothetical accident with release into the marine environment involving a Russian nuclear submarine in the Barents Sea
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Language: English.

Emneord:
Radioaktivitet, ubåt, ulykke, konsekvenser, konsentrasjon, doser, marint, biota

Resymé:
Rapporten viser resultater angående konsekvenser etter potensiell ulykke med moderne russisk ubåt.

Head of project: Mikhail Iosjpe

Approved:

Per Strand

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32 pages.
Published 2011-04-20.
Printed number 20 (11-04).
Cover design: LoboMedia AS.
Printed by LoboMedia AS, Oslo.
Cover illustration: Statens strålevern/Inger Sandved Arnfinsen www.koboltdesign.no

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

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Summary

The report presents results pertaining to a risk assessment of the potential consequences of a hypothetical accident involving a modern Russian submarine. The evaluation of the radioecological consequences is based on modelling of potential releases of radionuclides, radionuclide transport and uptake in the marine environment. Modelling work has been done, using a revised box model developed at the Norwegian Radiation Protection Authority. Evaluation of the radioecological consequences of a potential accident in the southern part of the Barents Sea has been made on the basis of the calculated collective dose to man, individual doses for the critical group, concentrations of radionuclides in seafood and doses to marine organisms. The results of calculations have been compared with the results of simulations with the recommendations and criterions for protection of the human population and the environment.

1 Introduction

Since the inception of its naval nuclear program, the Soviet Union/Russia has built a total of 255 nuclear-propelled surface and submersible military vessels – more than any other nation. Most of these have been fitted with two reactors. Today's Russian fleet consists of some 37 vessels stationed in the Russian High North and Far East (JIG, 2007). Two-thirds of these are third-generation vessels. Numerous accidents have occurred as a result of this activity (Reistad, 2008), and there are genuine concerns for large-scale releases that may come as a result of Russian activities involving marine reactors.

The largest accidental release – with the exception of Chernobyl, possibly the world's largest from an operating naval reactor – was the criticality accident in Chahzma Bay (1985), which involved the release to air of 200 x 10^{15} Bq (Sivintsev, 2000), with a high fraction of short-lived isotopes (Soyfer *et al*., 1995). For the dumped reactors in the Kara Sea, possible release rates averaged between 100-1000 x 10^{9} Bq per year for the various objects, with a peak release of 2700 x 10^{9} Bq per year by the year 2040 as the total for various first-generation reactor units with fuel in the rectors (IAEA, 1997). The main release mechanism was assessed to be pitting and bulk corrosion.

The estimated inventories for the reactors with fuel were up to 2.0 x 10^{15} Bq with ^{137}Cs as the dominant isotope (up to 23%). For a decommissioned vessel, the maximum release rate to seawater has been assessed to 2000 x 10^{12} Bq per year, falling to 60 x 10^{12} Bq/ year – however, with no significant doses, as the scenario was placed in a sparsely populated fiord close to Norway (NATO, 1998).

With respect to submarines in operation, for the *Komsomolets*, sunk due to fire and reactor shut-down, the main release mechanisms identified was fuel corrosion resulting in a maximum release of ^{137}Cs of 500 x 10^{9} Bq per year (Høibråten *et al*., 2003). The release rate of other radionuclides was assessed to be one order of magnitude lower. The inventory was given for three nuclides: 2.8 10^{15} Bq (^{90}Sr), 3.0 x 10^{15} Bq (^{137}Cs) and 4.4 x 10^{12} Bq (^{239}Pu). In Amundsen *et al.* (2002), the *Kursk* inventory was assessed to be 5 x 10^{15} Bq (^{90}Sr, ^{137}Cs each) after 24 000 MWd of operation, released after one year with the total collective dose from all nuclides estimated to 97 manSv.

This study is a response to the lack of updated assessments that take into account the increased nominal reactor power and the latest results from studies on reactor and fuel design and operational parameters (Reistad *et al*.,2005; Reistad 2008).

The objective has been to establish an upper threshold for the potential impact on the marine environment in the case of a Russian submarine accident involving a modern vessel with a maximum credible inventory of radionuclides and maximum release.

The geographical location of the accident, with subsequent release, has been set to the marine region outside the Russo-Norwegian border areas in the Barents Sea close to the operating naval bases in Northwest Russia.

The radioecological consequences after a hypothetical accident for a Russian nuclear submarine are based on modeling of potential releases of radionuclides, radionuclide transport and uptake in the marine environment. Modeling work has been done using a revised box model developed at the Norwegian Radiation Protection Authority (Iosjpe et al., 2002; Iosjpe, 2006). Evaluation of the radioecological consequences of a potential accident in the southern part of the Norwegian Current has been made on the basis of calculated concentrations of radionuclides in seafood, collective dose to man, individual doses for the critical group, and doses to marine organisms.

2 Methodology

This impact assessment is divided into two parts: the development of the radionuclide source term, and application of a relevant dispersion model for assessing the doses to humans and biota, with different methodological approaches for each part. As the objective was to complete a credible assessment, the focus has been on conservative but credible scenarios and assumptions.

2.1 Inventory

The source term consists of an inventory of radionuclides, released as a function of time and a release point. Each of these elements will be described below. The core inventory has primarily two components: the fuel matrix itself and the fuel burn-up. While the fuel matrix itself has only indirect influence on the amount of fission products, the amount of transuranics and release rates (discussed later) will depend directly on the type of matrix. In the current work, with its emphasis on a credible approach, the most probable representation of a Russian third-generation submarine core is a core load with 63% enriched fuel with 259.7 kg U-235 in a dispersion (UO_2-Al/ UO_2-Zr) or intermetallic configuration (UAl_x-Al) (Reistad *et al*., 2008). This composition has been verified to fit the only suggested core geometry for other than first-generation Russian submarines. However, as there exist an indefinite number of core configurations corresponding to various fuel volumes, the selection criterion has been to apply similar fuel density (4.5 Ug/cm^3) as that reported for Russian floating nuclear power plants under construction (Chuen and Reistad, 2007). A maximum credible inventory has been developed on the basis of a conservative approach to the average annual burn-up for third-generation reactors.

Average annual burn-up has been calculated to 30 effective full-power days (EFPD) and the maximum operational period hypothetically set to 20 years. At present, the average life-span for this class of vessels is 13.2 years. As the current decommissioning rate is higher than the commissioning rate, we may assume that this value will decrease slowly in the future. However, as the selection criterion has been a maximum credible burn-up, and normal vessel life is more than 13.2 years, we may assume 20 years of operation as a conservative estimate as a basis for calculating the radionuclide inventory at the time of the accident. The resultant burn-up is 114,000 MWd, or 269,000 MWd/tons of heavy metal (HM). We have also assumed an operating power fraction of 0.5 at the time of accident, resulting in a high inventory of short-lived isotopes when the hypothetical accident occurs. Most accident scenarios include a period where the reactor has been shut down or is operating on minimal power and subsequently a lower inventory of short-lived isotopes, before the release starts. The core inventory and core decay heat were developed using HELIOS 1.8 and SNF 1.2. HELIOS is a detailed reactor physics transport and burn-up code developed and supported by Studsvik Scandpower.

2.2 Accident scenario

There is no well–defined system for classification of accidents associated with naval vessels in general as for civilian power plants with Design Basis Accident (DBA), Reference Accident or Maximum Credible Accident (MCA). The secrecy maintained to this date have prevented the design and operational experiences from being adequately analyzed. The hypothetical scenario forming the basis for this study is if a core-melt / loss-of coolant accidents (LOCA) (Reistad *et al*., 2008) were to occur together with another type of incident, such as an explosion. Then there would be a credible risk of substantial damage to all parts of the submarine. An explosion that ruptured the hull and provided water intrusion in the reactor compartment would also contribute to cooling the corium. The specific incidents included in the assessment in Reistad et al. (2008) include the fire and sinking of the K-278 (*Komsomolets*) in 1989, the explosion and sinking of the *Kursk* in 2000, and the explosion and sinking of the K-219, a *Yankee*-class vessel, east of Bermuda in 1986. While the first two incidents did not result in any damage to the reactor section, in the third case the safety rods had to be manually inserted in the core, directly affecting the integrity of the core. In all, 165 safety-relevant events occurred for the Russian nuclear-propelled navy between 1959 and 2007.

While the release of radioactivity is one type of event that covers a large range of accident scenario, the mean time between each such event was calculated to 893 ± 138 vessel operating years (VOY) using statistical methods for reliability growth. At present, the Russian nuclear fleet is accumulating approximately 40 VOY annually. Other well-defined measures, e.g. core melt frequency, were methodologically difficult to obtain within reasonable confidence intervals due to accident clusters and too few events. No core-melt accidents have occurred for any third-generation vessel, though the operational experience for Russian third-generation submarines as of the end of 2007 is limited to 567 VOY, or 866 reactor-operating years. There have been several LOCA/ fuel damage accidents that have caused fuel damage and/ or led to the replacement of the reactor compartment or decommissioning of the vessel.

Kobayashi et al. (2001) use a scaled fuel inventory from the Genkai-2 plant based on the ratio between the nominal power for the power plant and the relevant propulsion plant (*Kursk*, 190 MW), resulting in an initial fuel content of 5.5 metric tons of low-enriched of ceramic UO_2-fuel and fuel burn-up of 194.75 MWd. Since the generation of important isotopes like Cs-137 and Sr-90 has a linear relationship to fuel burn-up, and low-enriched fuel leads to a large inventory of Pu, the resulting inventory, with similar release scenarios, will be significantly higher than this study, although not relevant for submarines. In Amundsen et al. (2001), calculations of the inventory of the *Kursk* were based on the description of the Russian cargo ship *Sevmorput*, using U-Zr alloy fuel matrix, 150.7 kg U-235 90% enriched, as given in the safety report. The burn-up was based on the fact that fresh fuel had been installed at the time of vessel commissioning, four years before the accident occurred in 2000. Two different burn-up estimates, 12,000 and 24,000 MWd, were established, using the same philosophy as in this paper. Similarly, for the *Komsomolets* a much lower inventory was taken forward, as the vessel had been in operation for only 4.5 years. Various versions for the *Komsomolets* inventory are shown in Table 1.

As a preliminary conclusion, two serious types of accidents have been identified. Each of them has the potential for substantial damage of the fuel and the vessel: a LOCA resulting in core-melt, and another event with the potential risk of water intrusion in the vessel compartments and the primary circuit taking place as part of or at the same time as the other event. Direct access between seawater and the primary circuit has been envisaged as part of the evaluation of the possible consequences of the case of the *Komsomolets* (Petrov, 1991), and cannot be excluded. Various possibilities for how the radionuclides may migrate to seawater have been described, including ventilation system and open access hatches. As a result, several scenarios are relevant, ranging from almost instantaneous release to seawater, to a staged approach, taking into account the hypothetical retention in the primary circuit, the reactor compartment and the submarine hull (Lewis and Morgan, 1999). In this study, we have established two scenarios as given in Table 2, with a priority for Scenario 1 in this study to establish the upper bound for the potential consequences. Scenario 2 includes a release of the remainder of the release fraction after a certain period of time, tentatively chosen as one year. This method may help us narrow down the area in which the maximum credible release can be expected. One year reflects, for example, a planning period for initiating salvaging or remedial actions like stabilization and vessel salvage, as in the case of the *Kursk*.

Table 1. Core inventory in _Komsomolets_ – various Russian studies

	Russian authorities – version 1	Russian authorities – version 2	Spassky (1991) "the moment of sinking"	Yablokov (1993)	Khlopkin (1994)
Kr-85			4.44E+14		
Sr-90	1.55E+15	2.80E+15	2.78E+15	1.55E+15	2.78E+15
Ru-106			8.88E+14		
Cs-134			1.89E+15		
Cs-137	2.00E+15	3.00E+15	3.07E+15	2.04E+15	3.07E+15
Ce-144			9.99E+15		
Pu-239					4.44E+12
Pu-240					1.70E+12
Pu-241					3.11E+14
Am-241					4.44E+10
Pu-242					9.99E+08
Cm-242					5.55E+12
Am-242m					1.48E+09
Am-243					1.67E+09
Cm-243					4.81E+08
Cm-244					3.15E+10

Table 2. Release fractions for maximum credible accident for third-generation submarine

<table>
<tr><th></th><th></th><th>Phase 1: From t = 0.1 days to t = 1 year</th><th>Phase 2: From t = 1 year</th></tr>
<tr><td rowspan="2">Scenario 1</td><td>Core-melt release</td><td colspan="2">Immediate release of release fraction as given in the release fraction in Table 3 (High Flux reactor)</td></tr>
<tr><td>Fuel corrosion</td><td colspan="2">Constant release of fuel corrosion products: corrosion rate: 0.01 % of fuel material annually</td></tr>
<tr><td rowspan="2">Scenario 2</td><td>Core-melt release</td><td></td><td>Immediate release of the remainder of release fraction as given in Table 3</td></tr>
<tr><td>Fuel corrosion</td><td>Corrosion processes initiated – corrosion rate: 0.01 % of fuel material annually – no releases</td><td>Immediate release of the accumulated corrosion products / continues corrosion rate: 0.01 %, release immediately</td></tr>
</table>

2.2.1 Release fractions

The main methodological problem here is the lack of relevant information on fuel materials, subsequently, on radionuclide behavior in fuel matrixes under extreme conditions (high temperature, saltwater intrusion etc.). However, a civilian nuclear system with potentially similar attributes to those of third-generation reactors – high power densities, high enrichment levels and moderate burn-up levels (50%) – is found in civilian research reactors. The hypothetical correspondence in fuel design and fuel properties has formed the basis for assessments of fuel consumption, as in Reistad et al. (2008b). Few civilian research facilities have been analyzed on the basis of probabilistic methods; deterministic accident analysis remains the most applied method for these facilities. For civilian facilities of similar size, the reference accident used as a basis for further analysis varies between a DBA describing complete and partial meltdown followed by water/ aluminum interaction and loss of mitigating systems. The source term evaluation for research facilities displays differences similar to those shown in Table 3, describing the release fractions for three HEU-fueled research reactors.

Table 3. Release fractions in the case of core meltdown following a LOCA (Abou Yehia and Bars, 2005)

	HIFAR	High flux reactor	SAFARI
Noble gases	1	1	1
I	0.3	0.8	1
Br		0.8	
Cs	0.3	0.8	0.163
Te	0.01	0.8	0.192
Rb	0.3	0.01	
Ru	0.01	0.1	0.005
Ba, Rh, Sr		0.1	
Actinides		0.01	0.1
Other		0.01	

The second component of the release fraction is fuel degradation and corrosion. Based on the hypothesis on the fuel matrix and the accident scenario, the corrosion processes of the uranium-loaded fuel component, UO_2 or UAl_x, starts immediately when the seawater enters the primary circuit. Experiments for long-term dissolution of fuel elements in seawater obtained dissolution rates from 0.1 to 1% of the fuel per year at temperatures from 10 to 20°C (Petrov, 1991). These assumptions were confirmed in experiments completed for the assessment of possible corrosion rates for fuel material in the dumped reactors at Novaya Zemlya, as specific corrosion rates for an UAl_x-matrix used in Russian icebreakers were identified to 0.3 mm/year (Yefimov, 1995): "If one considers this release as a result of corrosion destruction of the fuel matrix that takes place from the sample butt-ends then the corrosion rate of fuel composition with aluminum alloy in contact with seawater is about 0.3 mm/year." This was not bulk corrosion; regarding other than insoluble fission products, the base corrosion rate for U-Al alloy was in Lynn et al. (1995) identified to 0.03 mm/year. In any case, the corrosion rate for the other relevant fuel matrix (UO_2) was slower, ~0.0011 mm/year (Petrov, 1991). As temperatures in the Barents Sea at some depths are normally below this range, corrosion may be expected to proceed more slowly. However, in this study we have applied the conservative approach, assuming a U-Al matrix and 0.3 mm/ year.

2.2.2 Accident location

The release point, usually also part of the source term, has been established on the basis of the objectives of this study, but also in accordance with probable sailing routes leading from the main Russian submarine bases to the Atlantic Ocean (Figure 1). The location chosen for the accident was also based on an evaluation of the radiological sensitivity of marine areas relevant to the study. Radiological sensitivity analysis of Arctic marine regions shows that the North Norwegian coastline and the Barents Sea can be considered as the most vulnerable areas in the Arctic region, in terms of the effects of possible radioactive contamination (Iosjpe et al., 2003).

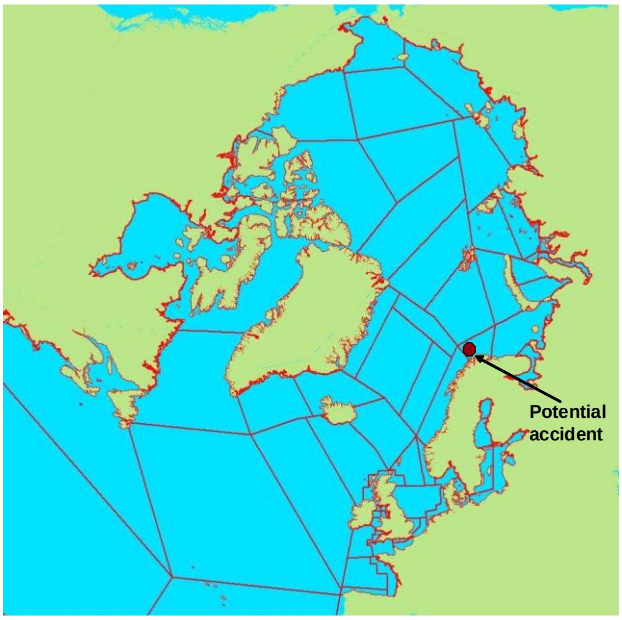


Figure 1. The structure of the NRPA box model and location of the potential accident.

2.3 Brief description of the NRPA box model

The box model developed at NRPA uses a modified approach for compartmental modeling (Iosjpe et al., 2002) which allows for dispersion of radionuclides over time. The box structures 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 (Karcher & Harms, 2000) generated from the 3D hydrodynamic model NAOSIM (the surface box structure is shown in Figure 1). The volume of the three water layers in each box has been calculated using detailed bathymetry together with a 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. Radioactive decay is calculated for all compartments. The contamination of biota is further calculated from the radionuclide concentrations in filtered seawater in the different water regions. Collective doses to the world population are calculated on the basis from seafood consumptions, in accordance with available data for seafood catches and assumptions about human diet in the respective areas (Nielsen et al., 1997, EC, 2000; IASAP, 2003).

The collective dose *D* can be determined using the following expression:

$$D = \sum_{j=1}^{m} DCF_j \sum_{l=1}^{k} \varphi_l \cdot CF_{lj} \sum_{i=1}^{n} A_{il} \int_0^T C_{ij}(t)dt, \qquad (1)$$

where [0, T] is the time interval; DCF_j is the dose conversion factor for radionuclide *j* (j = 1,2,…, m); CF_{lj} is the concentration factor for radionuclide *j* in seafood of type *l* (*l* = 1,2,…, k); A_{il} is catchment of seafood of type *l* in the model compartment *i*; (*i* = 1,2,…,n); C_{ij} is the concentration of radionuclide *j* in filtered seawater in model compartment *i*; and φ_l is the edible fraction for seafood of type *l* (the following assumptions (CEC, 1990; EC, 2000; IASAP, 2003) for the edible fractions of marine produce to the human diet have been used: 50 % for fish, 35 % for crustaceans and 15 % for molluscs).

Collective dose rates DR can be defined using the following expression:

$$DR = \frac{D(t_2) - D(t_1)}{t_2 - t_1}, \qquad (2)$$

where $D(t_1)$ and $D(t_2)$ are collective doses at times t_1 and t_2, respectively.

It is necessary to note that the model can also easily be used to provide information about impact to doses/dose rates from different marine regions, and provide dose assessment to different groups of population. Furthermore, the dose rate will be used in the present dose assessment because this parameter can easily indicate dose dynamic and is therefore widely used in present investigations (EC, 2003; IASAP; 2003).

Dose rates to biota are developed on the basis of calculated radionuclide concentrations in marine organisms, water and sediment, using dose conversion factors (Brown et al., 2006; Iosjpe, 2006). Expressions used for dose rates determination, which are used in the NRPA box model, are detailed in (Thørring et al., 2004; Brown et al., 2006).

It is important to note that the concentration factors used for calculating dose rates to biota (Hosseini et al., 2008) can differ significantly from IAEA recommendations (IAEA, 2004). This is largely because concentration factors given in ERICA data base (Hosseini et al., 2008) are calculated for the whole organism, whereas IAEA concentration factors are often defined only for edible parts of the organism i.e., that which has a potential consequence for dose assessments to man. In the present report, dose rates to man were calculated on the basis of concentration factors from the IAEA recommendations. For the calculations of dose rates to biota a conservative approach was chosen using concentration factors from the ERICA database, when these concentration factors were higher than the corresponding concentration factors from IAEA recommendations.

3 Results and discussion

The radioecological consequences of the potential scenarios leading to accidental releases of radioactivity have been evaluated on the basis of the calculated concentrations of radionuclides in typical sea foods, collective dose rates to man, individual doses for the critical groups and doses to marine organisms. Scenario 1 in Table 2 is the worst case scenario and, therefore, radioecological consequences of this scenario are the most conservative.

3.1 Release Scenario

The total and the individual releases of the radionuclides that had the most significant effect on the release rates during the initial and later phases of accidental releases are presented in Figures 2 and 3 for scenarios 1 and 2, respectively. As expected, the maximum release occurs during the initial period after the accident (the instant release fraction) with maximum values of $1.6{\cdot}10^{18}$ Bq and $4.6{\cdot}10^{16}$ Bq at the beginning of release for scenarios 1 and 2, respectively. Figure 1 shows that short-lived radionuclides of iodine and barium are most significant during the initial phase of release according to scenario 1, while ^{90}Sr and ^{137}Cs dominates in the final period of release. According to Scenario 2, release of radioactivity starts one year after the accident. Therefore, the impact of short-lived radionuclides to the total release of activity is negligible or strongly reduced under an initial phase of Scenario 2, which is dominated by ^{137}Cs, ^{134}Cs and ^{90}Sr. It is interesting to note that the same radionuclides (^{137}Cs, ^{90}Sr and ^{126}Sn) dominate the final phase of the total release for both Scenarios 1 and 2.

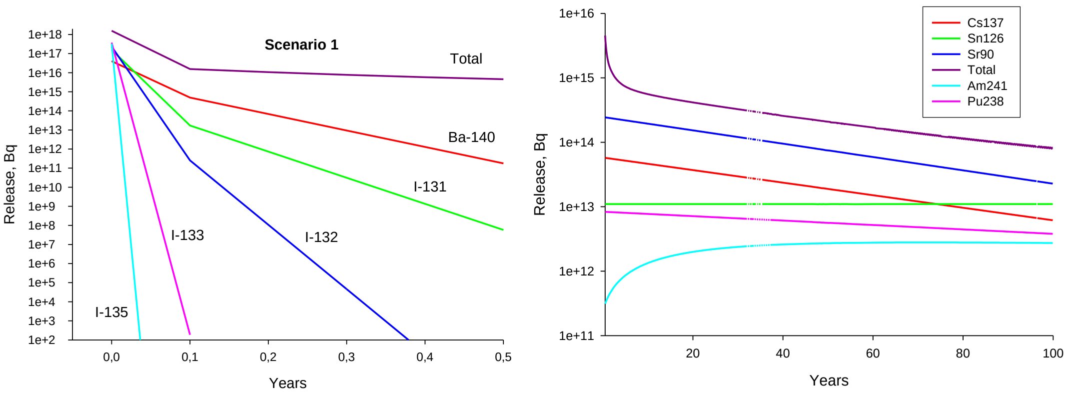


Figure 2. The worst case release scenario (Scenario 1 in Table 2) for the initial time of 0-0.5 year (left) and for time of 0.5-100 years (right)

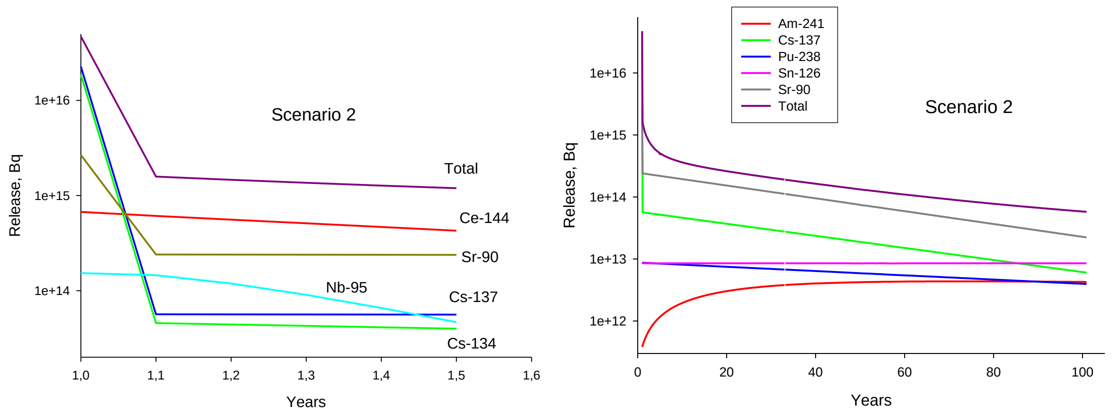


Figure 3. The release scenario 2 in Table 2 for the initial time of 1-1.5 year (left) and for the whole time period (right).

3.2 Concentration of radionuclides in seafood

Food and Agriculture organisation of the United Nations and World Health Organisation have provided recommendations (guideline levels) for the concentration of radionuclides in foods, when contaminated after an accidental release of radionuclides (CAC, 2006). According to CAC (2006) radionuclides can be separated into four groups. Examples of some typical radionuclides for each group are presented in Table 4.

Table 1. Examples of international guideline levels for radionuclides in food.

Radionuclides in Foods		**Guideline Level (Bq/kg)**	
		Infant Foods	**Other Foods**
Group 1	^{238}Pu, ^{239}Pu, ^{241}Am	1	10
Group 2	^{90}Sr, ^{106}Ru, ^{129}I	100	100
Group 3	^{60}Co, ^{134}Cs, ^{137}Cs	1000	1000
Group 4	^{3}H, ^{14}C, ^{99}Tc	1000	10000

Following the CAC (2006) recommendations, the model calculations for fish, crustaceans and molluscs are provided separately for each group of radionuclides presented in Table 4.

Some results of calculations of the radionuclide concentration in seafood (fish, crustaceans and mollusks) are shown in Figures 4 - 7 for the Barents Sea.

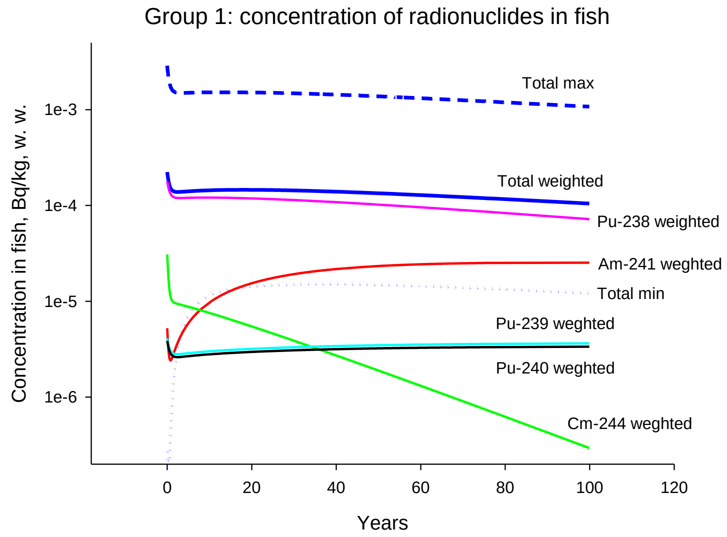


Figure 4. Concentration of radionuclides (Group 1) in fish.

Results in Figures 4 – 6 correspond to weighted / "average" concentration of radionuclides in seafood in the Barents Sea: $C_w^{(\xi)} = \dfrac{\sum_{i=1}^{n} C_i^{(\xi)} M_i^{(\xi)}}{\sum_{i=1}^{n} M_i^{(\xi)}}$, where $C_w^{(\xi)}$ is weighted concentration of radionuclides in seafood of type ξ , $C_i^{(\xi)}$ is concentration of radionuclides in seafood of type ξ in part *i* of the Barents Sea, $M_i^{(\xi)}$ is a catchment of seafood of type ξ in part *i* of the Barents Sea, and *n* is a number of water boxes in the Barents Sea. This concentration corresponds to samples, where species from different regions of the Barents Sea have been mixed.

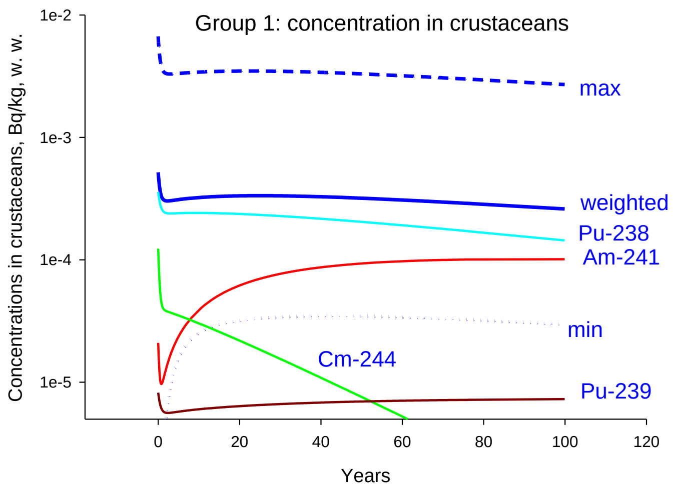


Figure 5. Concentration of radionuclides (Group 1) in crustacea.

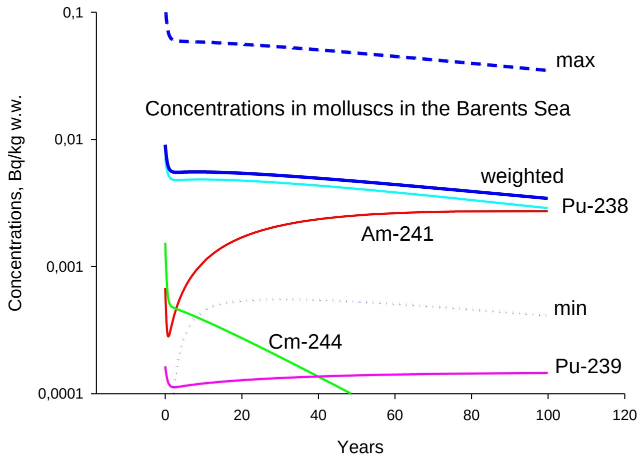


Figure 6. Concentration of radionuclides (Group 1) in molluscs.

Among the total weighted concentration of radionuclides in seafood, the total maximal and minimal concentration curves are also shown in Figures 4 – 6. The maximum and minimum concentrations correspond to the narrow and most remote regions of the Barents Sea in reference to the accident location. Figure 7 shows the maximal concentration of radionuclides in mollusks. It is necessary to note that the total concentration curve in Figure 7 is identical to the total maximal concentration curve in Figure 6.

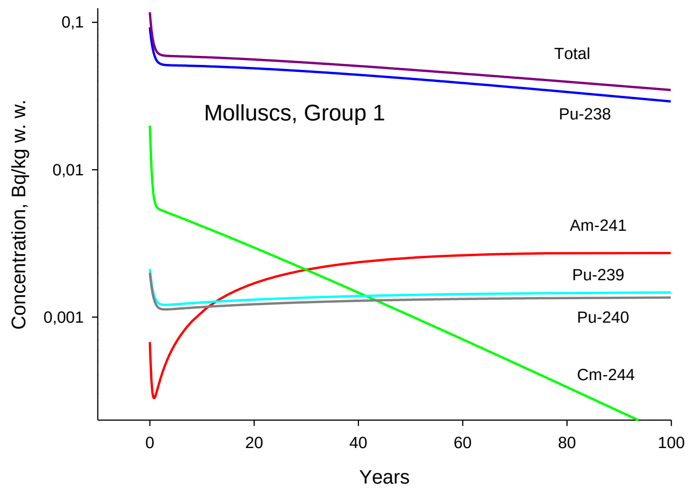


Figure 7. Maximal concentration of radionuclides (Group 1) in molluscs.

Results of the model calculations indicate that the concentration of radionuclides from group 1 in seafood lies significantly under the CAC guideline levels. Concentration reaches its maximum during the initial time after the accidental release began, and is followed by a relatively low decrease. The concentration level of radionuclides in seafood for group 1 of radionuclides is strongly dominated by ^{238}Pu.

Concentration of Group 2 radionuclides in seafood is shown in Figure 8.

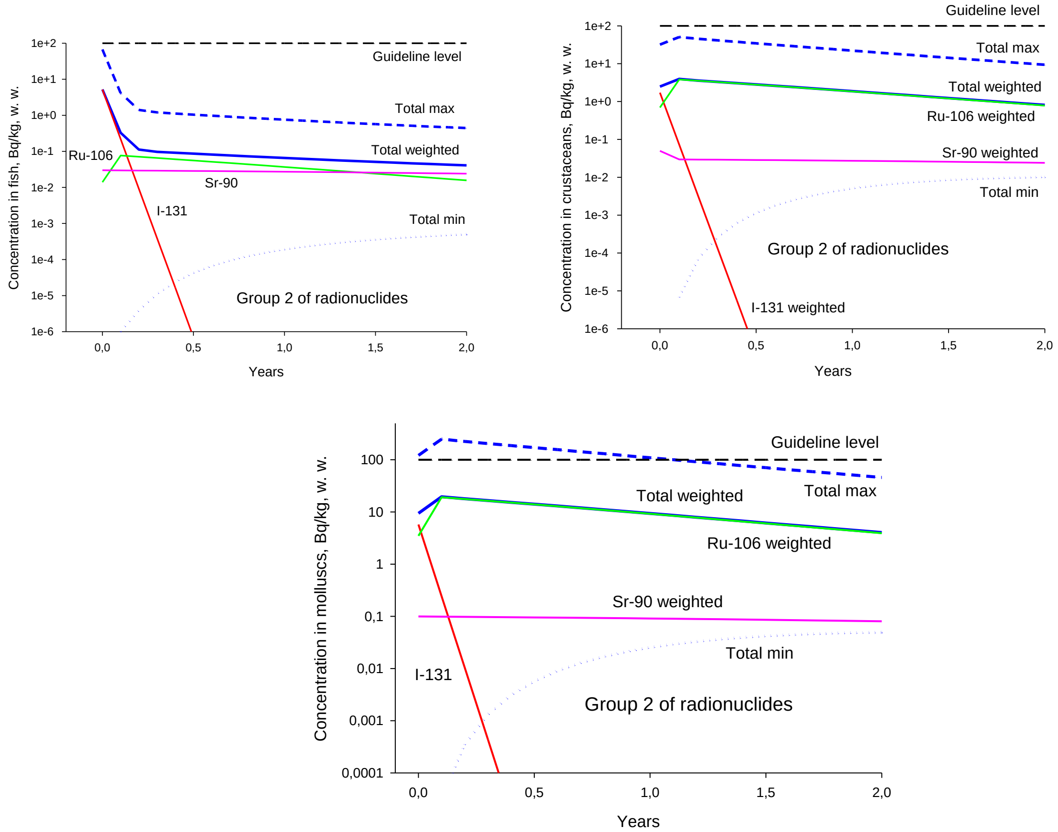


Figure 8. Concentration of radionuclides (Group 2) in seafood.

Concentrations of radionuclides in fish and crustaceans are lower then the CAC guideline level for group 2 nuclides, but contrary to radionuclides from group 1, concentration curves corresponded to the accident location narrow zone (total max curves) are close to the guideline level during initial time of release with relatively fast decreasing. Results of calculations for mollusks indicate that radionuclides concentration in the accident narrow zone exceeds the CAC guideline level during one year after accident, approximately. The maximal value of the concentration level of radionuclides in mollusks is around of 250 Bq/kg. The radionuclides that impacted the concentration levels in seafood (for group 2) the most were ^{106}Ru, ^{131}I and ^{90}Sr.

Maximal concentrations of radionuclides in seafood from group 3 are shown in Figures 9. Results of calculations in Figure 9 demonstrate that the concentration curves lies significantly under the CAC guideline levels. Nuclides ^{127}Sb, ^{133}I, and ^{137}Cs, ^{134}Cs dominate the concentration level of radionuclides in seafood for group 3 under the initial and following time after accident, respectively.

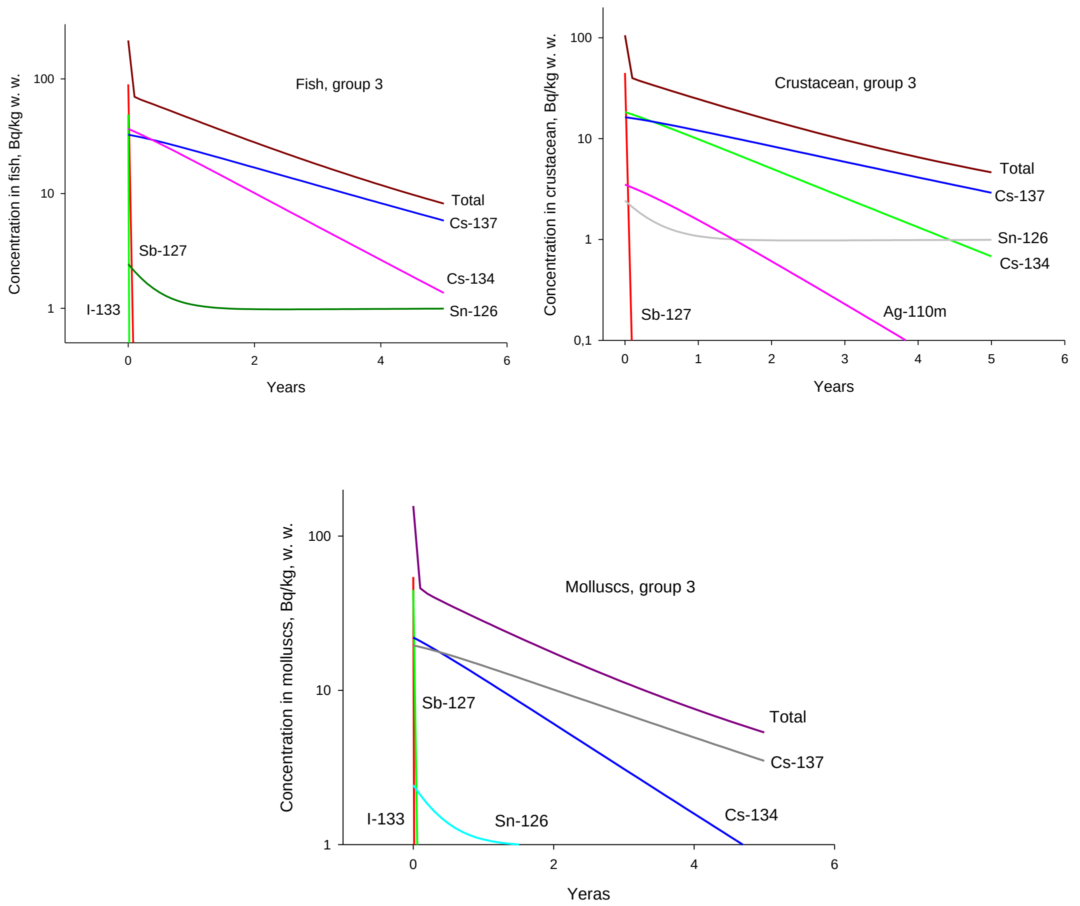


Figure 9. Concentration of radionuclides (Group 3) in seafood.

As the same time a random fish sample in the zone, which is closest to the accident location, can indicate that concentration of radionuclides from group 3 is considerably in excess of the CAC guideline level during three weeks after the accidental release began (Figure 10). In this case, the level of concentration of radionuclides in benthic fish is dominated by ^{132}Te.

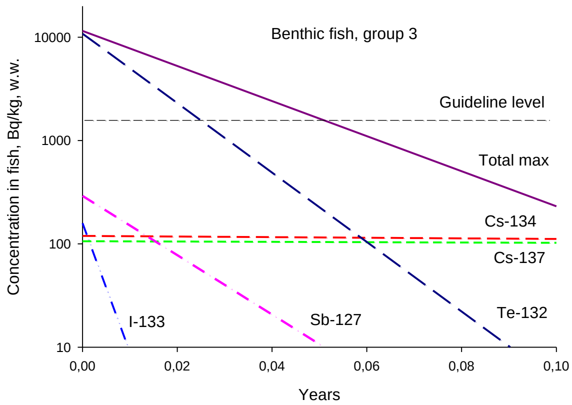


Figure 10. Concentration of radionuclides (Group 3) in benthic fish in the bottom waters near the accident location.

Figure 11 shows that contamination curves of seafood by Group 4 radionuclides lies under the guideline levels, similar to the Group 3 case, but the concentration of radionuclides in mollusks (970 Bq/kg) lies very close to the infant guideline level. ^{127}Te, ^{129m}Te, and ^{103}Ru dominate the radionuclide concentration.

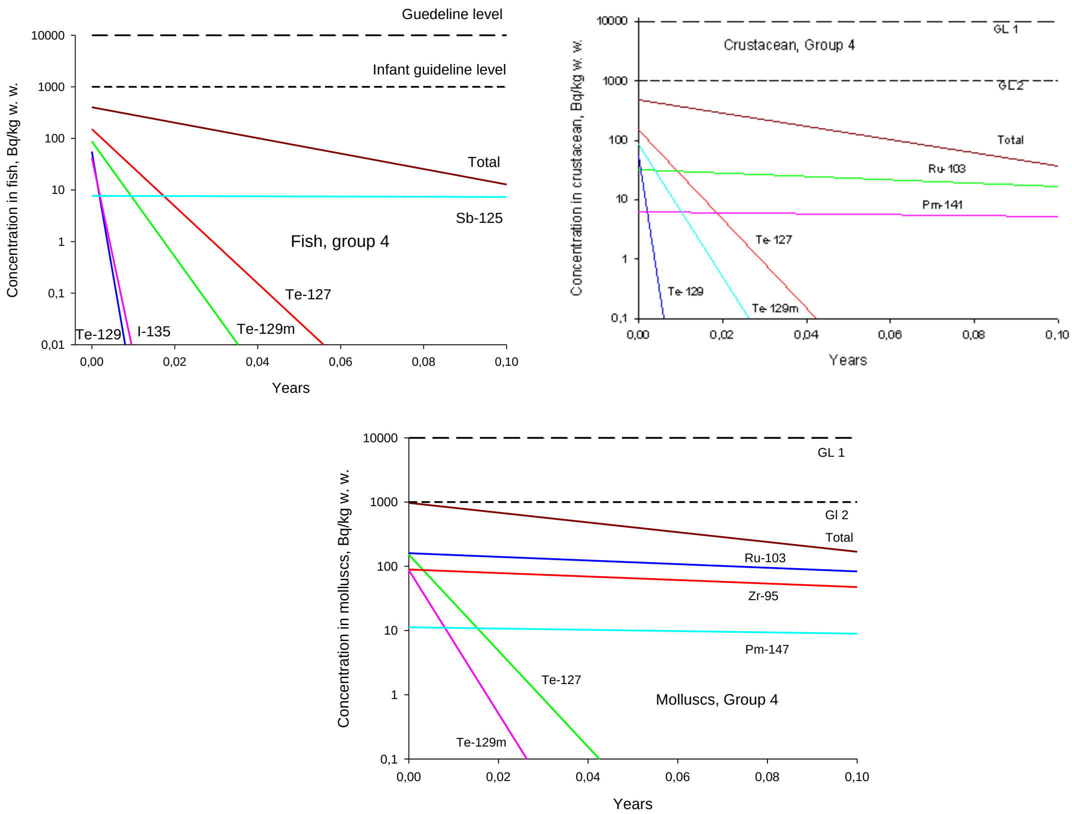


Figure 11. Concentration of radionuclides (Group 4) in seafood (GL 1 and GL 2 are guideline levels for infant and other food).

It is necessary to note that during the human habit assessment for infants (Smith and Jones, 2003; US DoHHS, 1998), which was used for the CAC guideline levels development, the consumption of fish was found to be very low, while consumption of crustaceans and molluscs was not found at all, probably because it is generally recommended to avoid feeding children seafood before the age of 12-36 months, due to allergy concerns (Fiocchi, et al., 2006; Kull et al., 2006). Therefore the close values of radionuclide concentration in mollusks to infant guide level for Group 4 nuclides will not lead to any restrictions. As the same time, similar to the case concerning Group 3, a random fish sample in the zone, which is closest to the accident location, indicates that the concentration of radionuclides from group 4 is in excess of the CAC guideline level during two-three days after the accidental release began (Figure 12). ^{127}Te, ^{129m}Te, and ^{129}Te dominate the radionuclide concentration in benthic fish during the time.

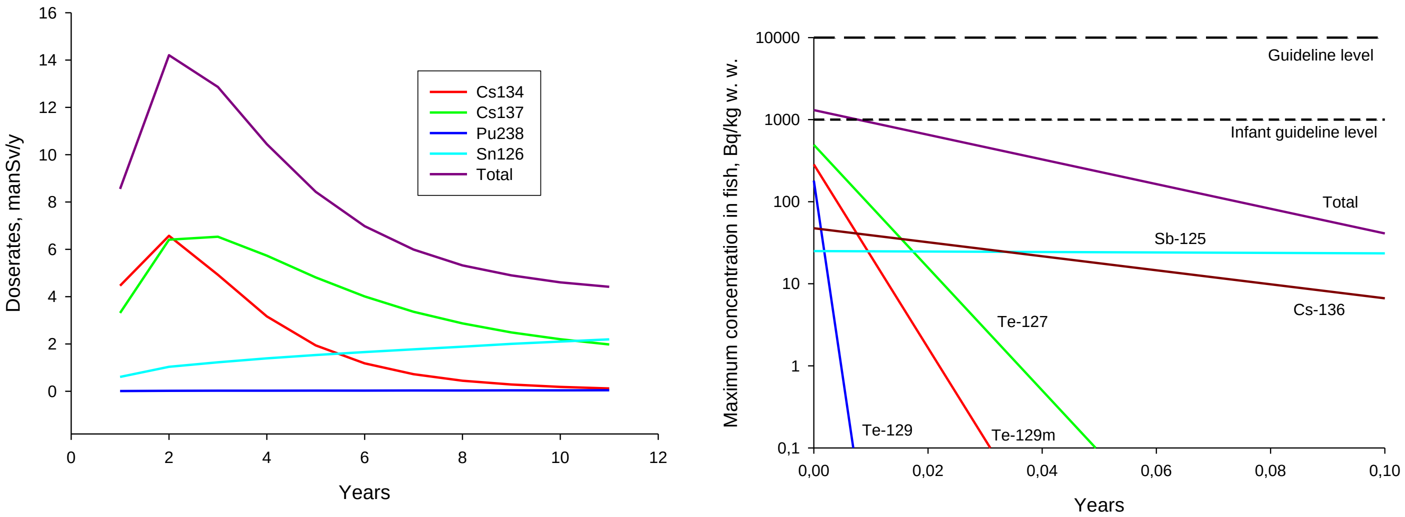


Figure 12. Concentration of radionuclides (Group 4) in benthic fish in the bottom waters near the accident location.

3.3 Collective dose-rates to man

The scenario entails the release of 110 kinds of radionuclides into the marine environment. More than 50 radionuclides were considered; Figure 13 presents the results for the radionuclides that were calculated to have the most significant impact regarding doses to man during the initial period of release. The results presented in Figure 13 show that maximum collective dose rates in the studied scenario, calculated for the world population, occur during the second year after the release of radioactivity. The maximum collective dose rate is approximately 14 manSv per year, with ^{137}Cs and ^{134}Cs giving the highest impact on maximal total collective dose rate, while ^{126}Sn and ^{137}Cs dominate collective dose rates after six years since release.

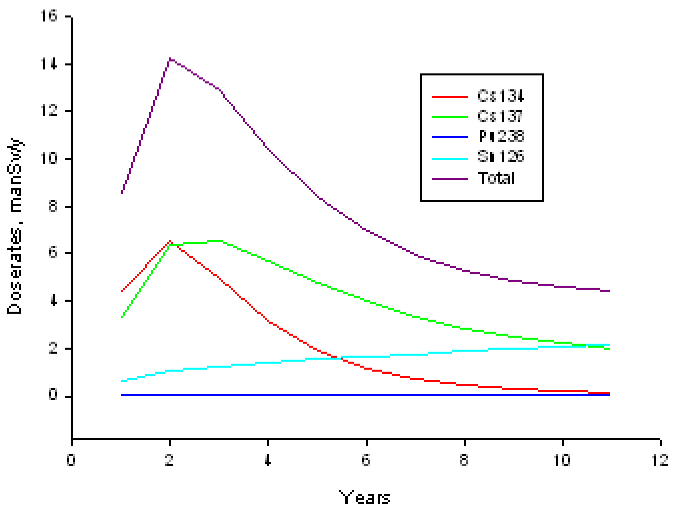


Figure 13. *Collective dose-rates*

There are different approaches for evaluating effects from low-dose radiation (ICPR, 2007) and, for example, according to a non-threshold model, any dose is harmful, no matter how small. Therefore, it is interesting to compare the present results for the collective dose rate with dose rates for exposure from natural sources.

Considering the accident location, radionuclides will be mainly dispersed in the Barents Sea, during the first two years after the release began (Iosjpe et al., 2002). Therefore, with a rough approximation, it is possible to evaluate the size of the "local" population (Norwegian and Russian) which will be affected by the consequences after the accident. Since the total fish catchments from these marine regions are approximately $7.2 \cdot 10^8$ kg per year, while typical fish consumption for Norwegian and Russian population (for Russian population, data corresponds to the Kola Peninsula) are 25 and 50 kg per year, respectively (Bergsten, 2003; IASAP, 2003), we can estimate the actual population to be in the range of $1.4 \cdot 10^7$- $2.9 \cdot 10^7$. According to (UNSCEAR, 2000), the annual exposure from natural sources can be expected to be in the range of 1 – 10 mSv, with an average annual exposure of 2 mSv. Knowing this information, we can evaluate the collective dose per year from natural sources DR_n using the approximation (ICRP, 2007) $DR_n = \sum_i E_i N_i$, where E_i is the average effective dose per year for group *i* and N_i is the number of individuals in this group.

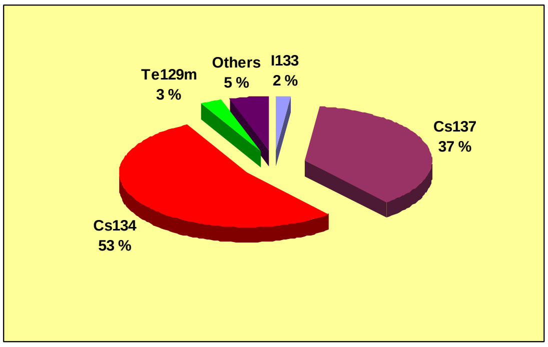
Thus, a rough estimation for the collective dose per year for the above mentioned population group can be expected to be in the range of $1.4 \cdot 10^4$ - $2.9 \cdot 10^5$ manSv. Furthermore, the exposure rates in the range of 1-10 µSv per year constitutes, according to (UNSCEAR, 2000), a negligible component of the annual effective dose from natural sources. With this assumption, the value 14 manSv per year (1 µSv · $1.4 \cdot 10^7$) can be used as conservative estimation of the negligible component of the annual effective dose from natural sources. Since calculations provide the value 14.2 manSv for the collective dose-rate (second year after accident), it is not possible to ignore the present results as negligible in comparison with nature sources.

3.4 Doses to the critical group

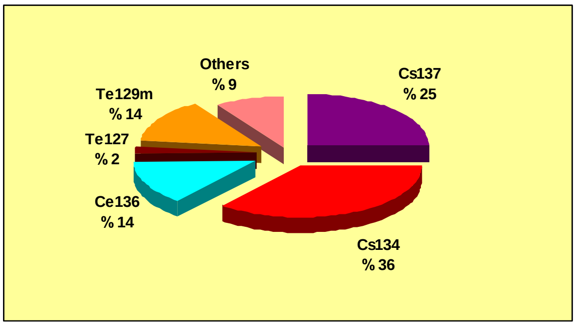
According to an investigation of consumption patterns for different populations living on the Norwegian coast and inland (Bergsten, 2003), maximum seafood consumption is 200, 40 and 4 g/day for fish, crustaceans and mollusks, respectively. Based on this investigation, the hypothetical group with heavy consumption of seafood from the most affected sea region (the southern part of the Barents Sea) was chosen for the evaluation.

The individual dose rates for the ingestion pathway have been calculated on the basis of expressions (1) and (2), where catchments of seafood were replaced by consumptions for the critical groups.

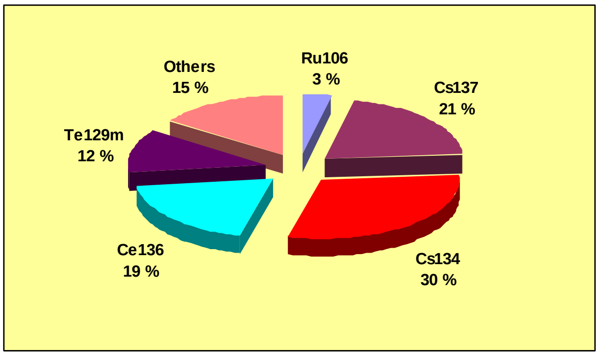
The proportions of the total calculated dose attributable to the different types of seafood are presented in Figures 14, corresponded to maximal dose-rate. ^{137}Cs and ^{134}Cs were the two radionuclides that gave the most significant contribution to doses (Figure 14).



Fish (0.07 mSv per year)



Crustacean (0.01 mSv per year)



Mollusks (0.002 mSv per year)

Figure 14. *Dose impact to critical group from fish, crustaceans and molluscs*

The calculated maximal dose-rate equals 82 μSv yr^{-1}, which is significantly lower than the average annual dose of 2 mSv from nature sources. At the same time, this dose-rate is significantly higher than range 1 - 10 μSv yr^{-1} for the negligible component to the annual individual dose from natural sources (UNSCEAR, 2000) and, therefore, has to be under consideration during evaluation of the accident consequences.

3.5 Doses to marine organisms

Dose rates calculated for different reference marine organisms (fish, crustacean and mollusks) in the southern part of the Norwegian Current (the location for the hypothetical accident) are presented in Figure 14. Figure 14 also indicates that dose rates in fish, crustaceans and mollusks vary similarly with time, with the maximum concentration of radionuclides corresponding to the initial time after the accidental releases began and with a following relatively fast decrease with time. The most significant impact to the total dose rate to pelagic fish and crustacean was observed for ^{132}Te, while the dose rates to mollusks are dominated by ^{242}Cm and ^{132}Te.

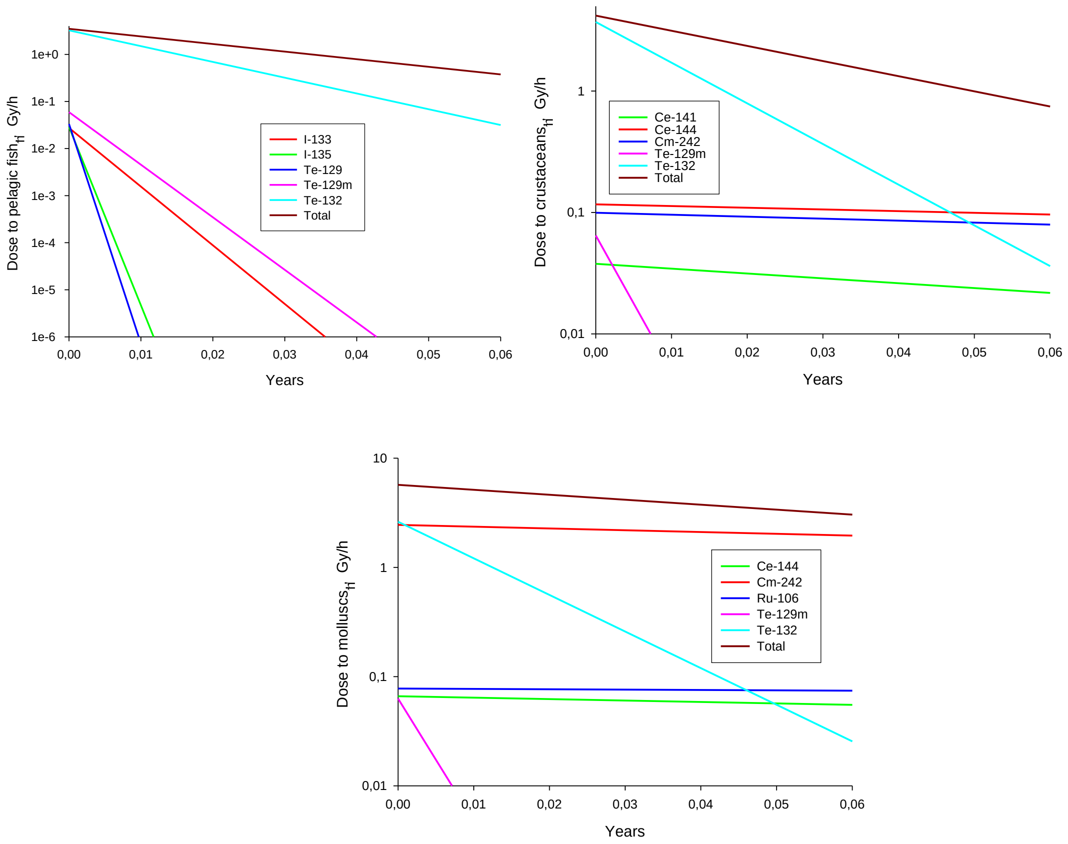


Figure 14. Doses to biota

It has previously been suggested that the screening dose rate of 10 μGy h^{-1} or less are not harmful to marine biota (Brown et al., 2006). On the other hand, according to the US Department of Energy, the dose rate limit of 400 μGy h^{-1} can be accepted for native aquatic animals (US DoE, 2002). This difference can primarily be explained by the different approaches for evaluating these dose rate limits. The estimation of the screening dose rate is based on the evaluation of radiation effects to individual organisms, while the dose rate limit, provided by US DoE (2002), is based on the evaluation of effects for the population of aquatic organisms, where minor effects for individual organisms are not expected to be significant for the viability of the population. Figure 14 indicates that doses to marine organisms are significantly lower than the screening dose rate. At the same time, Figure 15 shows that the results of calculations for the polychaete worm, living in sea sediment, are significantly higher than the screening dose rate and the dose rate limit according to Brown et al. (2006) and US DoE (2002). Figure 15 indicates that the dose rate calculated for the polychaete worm, contrary to results presented in Figure 14, does not decrease over time. This will affect many generations of this particular marine organism because the polychaete worm lifespan is 2-3 years.

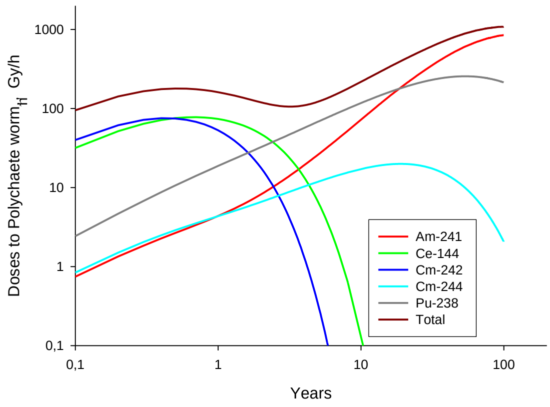


Figure 15. Doses to the Polychaete Worm

It is interesting to note that radiation effects for fish reproduction are reported for dose rates higher than 100 μGy h^{-1} (Real et al.,_2004), while for the earthworm the first radiation effects (reproduction capacity) were not observed for dose rates lower than $4 \cdot 10^{3}$ μGy h^{-1} (Hertel-Aas, 2008). Thus, a fact, that dose to the *Polychaete Worm* exceed the dose rate limits does not automatically mean damage to organism colony, but it means that this situation has to take a special consideration.

4 Uncertainties

The accuracy of the calculations can be improved by the development of a more detailed source term, through refinements of the concentration factors and sediment distribution coefficients, which are now defined with a precision of up to one order of magnitude (IAEA, 2004), and increased knowledge about water-sediment interaction with regards to sedimentation and remobilisation processes for radionuclides. Furthermore, accuracy of calculations can be improved by considering low doses effects and the effects to population dynamic of marine organisms

5 Conclusions

The consequences after an accident with the modern Russian submarine were calculated on the basis of most conservative scenario considered in the present investigation.

Calculations indicate that, generally, concentration of radionuclides on seafood is under the international guideline levels for different groups of radionuclides. Simultaneously, results of calculations indicated that concentrations of radionuclides for some marine organisms during initial time of release near the accident location exceeded guideline levels. Elevated levels of radionuclides in marine food products may lead to economic consequences in a market very sensitive to contaminants.

Calculated collective dose rates to man as well as doses to a critical group are significantly lower than doses from natural sources, but at the same time, these dose-rates are significantly higher than the negligible component to the annual individual dose from natural sources (UNSCEAR, 2000) and, therefore, have to be taken under consideration during evaluation of the accident consequences.

Calculations of the doses to marine organisms indicate that doses to marine organisms are lower than the screening dose rate (not harmful dose-rate level to marine biota). At the same time, the results of calculations for the organisms, living in sea sediment near the accident location, can be significantly higher than screening dose rate during lifespan for many generations of these marine organisms, which means that statistically significant effects can be expected for colonies of such organisms.

As a consequence, extensive additional monitoring of marine environment as well as assessment of contamination levels in the environment, as well as doses to man and biota are expected following an analogue accident with a modern Russian submarine sinking in the coastal waters in the Barents Sea.

The accuracy of the conclusions can be boosted by improving the methodology of the source term evaluation, dose assessment modeling concerning man and biota, and considering low dosage effects and the effects to population dynamic of marine organisms

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Lobo Media 46188

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