JOINT NORWEGIAN-RUSSIAN EXPERT GROUP
for investigation of Radioactive Contamination in the Northern Areas

INVESTIGATION INTO THE RADIOECOLOGICAL STATUS OF STEPOVOGO FJORD

The dumping site of the nuclear submarine K-27 and solid radioactive waste

Results from the 2012 research cruise

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Edited by Justin P. Gwynn and Alexander I. Nikitin

Preface

This report presents the results obtained by the joint Norwegian-Russian research cruise in 2012 to investigate the radioecological status of Stepovogo Fjord on the eastern coast of Novaya Zemlya, where the nuclear submarine K-27 and solid radioactive waste has been dumped. The research cruise was conducted onboard the R.V. Ivan Petrov of the Federal Service for Hydrometeorology and Environmental Monitoring (Roshydromet). This report has been written by a joint Norwegian-Russian working group with the following contributors:

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Executive summary

The joint Norwegian-Russian cruise to Stepovogo Fjord in 2012 to investigate the radioecological status of dumping sites for solid radioactive waste and the nuclear submarine K-27 was organized through the Norwegian-Russian expert group for investigation of radioactive contamination in the northern areas. The joint Norwegian-Russian cruise in 2012 followed on from previous joint Norwegian-Russian cruises to dumping sites of radioactive waste in the Kara Sea and Novaya Zemlya fjords in 1992, 1993 and 1994.

In 2012, the nuclear submarine K-27 was observed lying upright and clear of bottom sediments at a depth of around 30 m in the outer part of Stepovogo Fjord with no obvious corrosion damage of the outer hull. Based on in situ gamma measurements and the analysis of seawater and sediment samples taken around the submarine, there was no indication of any leakage from the reactor units of K-27.

In the inner part of Stepovogo Fjord, it was not possible to draw any direct comparisons with observations of dumped containers in the previous investigations in 1993 and 1994. Equally, it was not possible on the basis of the 2012 investigation to provide any overall assessment of the status of such dumped objects in the inner part of Stepovogo Fjord.

With regard to the radioecological status of Stepovogo Fjord, activity concentrations of all radionuclides in seawater, sediment and biota in 2012 were in general lower than reported from the previous investigations in the 1990s and were comparable to or lower than reported values for other marine areas for a similar time period. However in 2012, the activity concentrations of Cs-137 and, to a lesser extent, those of Sr-90 remained elevated in bottom water from the inner part of Stepovogo Fjord compared with surface water and the outer part of Stepovogo Fjord.

Although the current environmental levels of radionuclides in Stepovogo Fjord are not of immediate cause for concern, further monitoring of the situation is warranted. In particular, a better understanding of the amount, source and status of waste that has been dumped in the inner part of Stepovogo Fjord is required. Additionally, the situation with regard to the nuclear submarine K-27 in the outer part of Stepovogo Fjord should be followed, especially in connection with any future plans involving the recovery of K-27 from the fjord.

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1. Introduction

1.1 Norwegian-Russian cooperation

In 1992 a Norwegian-Russian expert group was established to investigate radioactive contamination in the northern areas under the joint Norwegian-Russian Commission for Cooperation in the Environmental Sector. At first headed by the Ministry of Environmental Protection of the Russian Federation and the Norwegian Ministry of Environment, the Norwegian-Russian expert group is currently jointly led by the Federal Service for Hydrometeorology and Environmental Monitoring (Roshydromet) from the Russian side and by the Norwegian Radiation Protection Authority (NRPA) from the Norwegian side. The Norwegian-Russian expert group was formed in the light of new information concerning dumping of radioactive waste in the Barents and Kara Seas by the Former Soviet Union and to develop a joint plan to investigate this issue.

The objectives defined for the expert group were:

- To obtain information on the handling, storage, discharge and dumping of radioactive material in the northern areas.
- To investigate, through field work, the actual levels of radioactive contamination in the open Kara Sea and at the dumping sites.
- To locate dumped nuclear waste and identify if any leakage of radioactive substances has taken place.
- To undertake impact and risk assessments for man and the environment.
- To inform the public of the results of these investigations

The Norwegian-Russian expert group has enabled greater cooperation between Norway and Russia with regard to nuclear safety and radiological environmental assessments. In recent years, the Norwegian-Russian expert group has made significant strides in a number of target areas including:

- Cooperation with Russian nuclear regulatory authorities on regulations, inspections, licensing and permissions with regard to risk reduction, with a particular focus on Andreeva Fjord.
- Risk and consequence assessments for actual and potential sources of radioactive contamination in North West Russia.
- The wider use of consequence assessments as a component of decision based processes by Russian regulatory authorities.
- The removal and safe disposal of 180 radioactive sources (RTGs) from light beacons in North West Russia and 71 similar sources from the Russian coastline in the Baltic Sea.
- Continued cooperation through the Norwegian-Russian environmental monitoring programme to document trends in radioactive contamination in the Barents Sea and the extension of the programme to include terrestrial monitoring.
- Cooperation on the environmental impact of radioactive contamination, including joint fieldwork, in the area surrounding the Russian nuclear facility at Majak and consequence assessments for potential accidents at the facility.
- Joint Norwegian-Russian cruises to dumping sites of radioactive waste in the Kara Sea and Novaya Zemlya fjords in 1992, 1993 and 1994.
- The joint Norwegian-Russian cruise to the Kara Sea and Stepovogo Fjord in 2012 to investigate the status of dumped radioactive waste including the nuclear submarine K-27.

1.2 Dumping of nuclear waste in the Barents and Kara Seas

Regular dumping of liquid and solid radioactive waste in the Arctic was practiced by the former USSR and later by Russia from the early 1960s until the early 1990s. Assessments of the total activity of liquid and solid radioactive waste dumped into the Barents and Kara Seas were first reported in the White Book (1993), then revised by the International Arctic Seas Assessment Project (IASAP) in 1993-1996 and subsequently summarised in the IAEA technical document 'Inventory of radioactive waste disposal at sea' (IAEA, 1999). More recently, the White Book 2000 (Sivintsev et al., 2005) reassessed the information originally published in the White Book (1993) and identified a number of errors, inaccuracies and omissions. The total activity of liquid and solid radioactive waste dumped in the Barents and Kara Seas reported by the White Book 2000 is 38801.81 TBq (Sivintsev et al., 2005), equivalent to approximately 45% of the total activity of radioactive waste dumped in the global oceans. However it is likely that the true figure is somewhat higher as the White Book 2000 (Sivintsev et al., 2005) identified a number of dumping operations within the Barents and Kara Seas without providing any information on associated activitites of the dumped waste.

Table 1.1. *Total activity (TBq) at time of dumping of different types of radioactive waste dumped in the Arctic region by the Former USSR and Russia as reported in the White Book 2000 (Sivintsev et al., 2005).*

Waste type	Total activity at time of dumping (TBq)	Percent of total activity
Reactor units with spent nuclear fuel	21781	56.1
Reactor units without spent nuclear fuel	14802	38.1
Reactor components	20.8	0.1
Low level solid waste	1240.21	3.2
Low level liquid waste	957.8	2.5
Total	**38801.81**	**100**

Liquid radioactive waste of an activity of 435.2 TBq was deliberately dumped into the Barents Sea within five specially allocated areas, while an additional 522.6 TBq was dumped as a result of operational accidents in the Barents, Kara and White Seas (Sivintsev et al., 2005). Low- and intermediate-level solid radioactive waste (SRW) was prinicpally dumped in eight main areas covering the fjords east of Novaya Zemlya and the Novaya Zemlya trough in the open Kara Sea. By volume, the bulk of the SRW dumped consists of waste produced during the operation of the naval ships, icebreakers, and submarines with nuclear reactors.

Table 1.2. *Liquid radioactive waste dumped in the Arctic region as reported in the White Book 2000 (Sivintsev et al., 2005).*

Location	Dumping area	Years	Total volume (m^3)	Total activity (TBq)		Remarks
				At time of dumping	In 2000	
Barents Sea	1	1968-1989	15639	297.9	130.3	
Barents Sea	2	1960-1990	66811	133.3	37.5	
Barents Sea	3	1966-1989	53300	77.0	24.3	
Barents Sea	4	1975-1991	8507	2.0	0.9	
Barents Sea	5	1966-1992	49838	12.4	5.4	
Andreeva Fjord	-	1982-1986	9000	42.6	16.5	Leaks from storage facility
Ara Fjord	-	1989	20	74.0	37.2	Nuclear submarine accident
White Sea	-	1959, 1965	610	3.7	0.83	Principally from explosion at Severodvinsk shipyard
Abrosimov Fjord	-	1967	370	0.01	0.002	Barge MNN-231500
Kara Sea	-	1964-1977	1095	315.0	96.0	Principally from LRW of icebreaker in lighter PSSN-328
Total			**205190**	**957.8**	**348.9**	

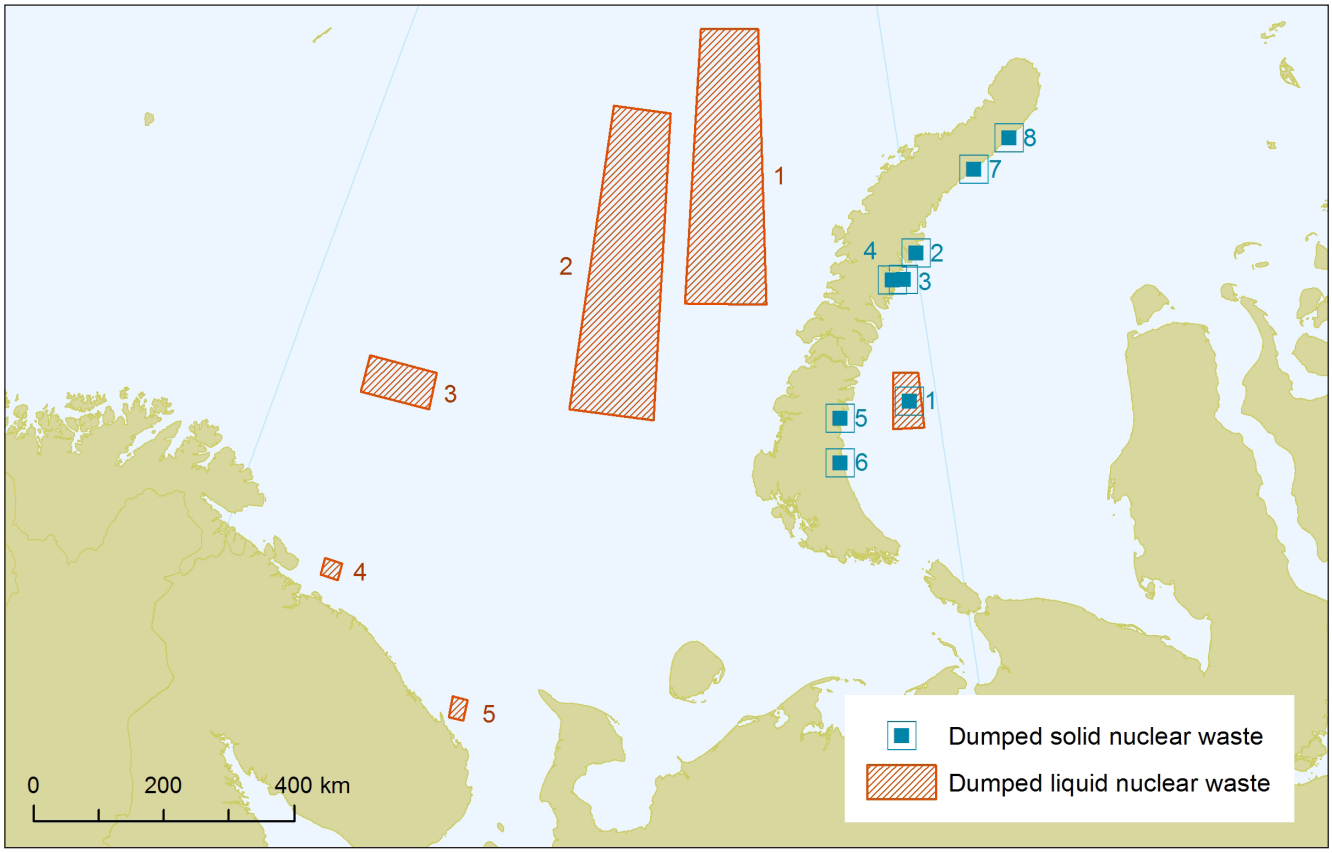


Figure 1.1. *Main dumping areas in the Barents and Kara Seas as reported in the White Book 2000 (Sivintsev et al., 2005).*

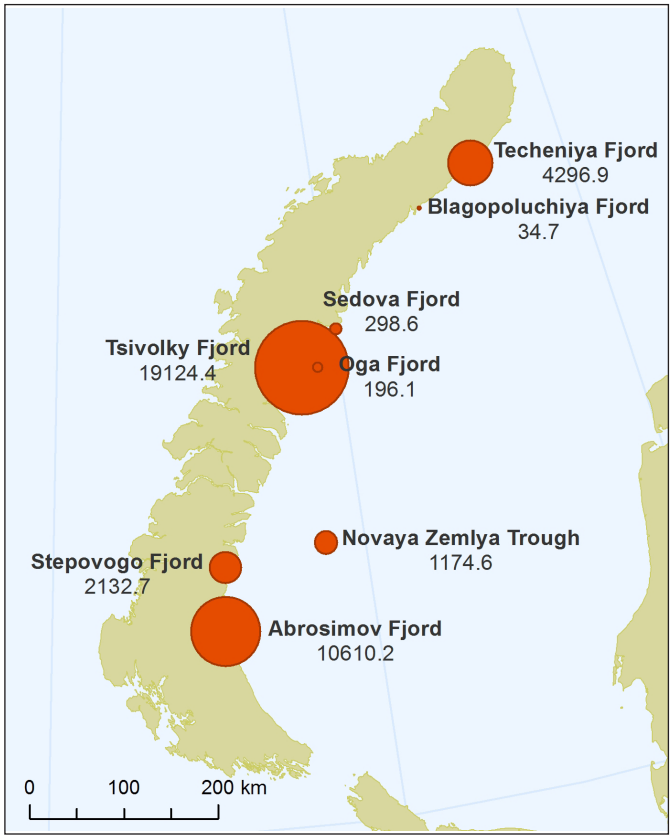


Figure 1.2. *Revised estimates of the maximum total activity (TBq) of the dumped solid waste at the time of dumping as reported in the White Book 2000 (Sivintsev et al., 2005).*

As a rule, low and intermediate-level SRW dumped in the Kara Sea was enclosed in metal containers. Large waste objects were dumped separately or inside specially allocated ships, such as barges, lighters and tankers. The total activity of low and intermediate-level SRW dumped in the Kara Sea amounts to 1240.21 TBq, with a further 11.1 TBq having been dumped in the Barents Sea (Sivintsev et al., 2005).

Table 1.3. *Low and intermediate-level solid radioactive waste dumped in the Barents and Kara Seas as reported in the White Book 2000 (Sivintsev et al., 2005).*

Location	Dumping area	Years	No. of containers	No. of unpacked items	Total activity (TBq)		Remarks
					At time of dumping	In 2000	
Novaya Zemlya trough	1	1967-1991	4824	561	288.5	112.1	Operational waste and components from the "Lenin"
Sedova Fjord	2	1982-1984	1100	112	296.6	111.8	Operational waste
Oga Fjord	3	1968-1983	2190	101	191.2	64.0	Operational waste
Tsivolky Fjord	4	1964-1978	5242	166	229.4	53.3	Operational waste
Stepovogo Fjord	5	1968-1975	1917	3	106.0	28.1	Operational waste
Abrosimov Fjord	6	1966-1981	646	-	55.8	16.7	Operational waste
Blagopoluchiya Fjord	7	1971-1972	992	2	27.7	7.7	Operational waste
Techeniya Fjord	8	1982-1988	194	28	33.9	15.9	Operational waste
NW of Kolguyev Island	-	1978	-	18	2.56	0.83	Various ship components
Barents Sea	-	1959	-	-	8.55	1.64	Barge with SRW
Total			**17105**	**991**	**1240.21**	**412.07**	

Reactors and reactor compartments, both with and without spent nuclear fuel (SNF), were also dumped in the Kara Sea with a total activity of 36583 TBq. In addition, a number of reactor components have been dumped at various locations with a total activity of 20.8 TBq. The objects containing SNF are of the greatest potential radioecological hazard among all the radioactive waste dumped in the Arctic seas. A total of 5 objects containing 6 reactors with SNF and a screening assembly with 60% of the fuel from the OK-150 unit from the icebreaker "Lenin" as well as 5 objects containing 10 reactors without SNF have been dumped (Sivintsev et al., 2005). All reactors containing SNF were dumped with their reactor compartments filled with a furforal mixture as a protective barrier. According to design specifications, the filling of the reactor compartments with furforal should prevent SNF from coming into contact with seawater for a period of up to 500 years (Sivintsev et al., 2005).

Table 1.4. *Total activity at time of dumping in reactor units dumped in the Kara Sea as reported in the White Book 2000 (Sivintsev et al., 2005).*

Location	Year of dumping	Unit No.	No. of reactors in unit		Total activity at time of dumping (TBq)
			Without SNF	With SNF	
Abrosimov Fjord	1965	No. 285	1	1	3968
	1965	No. 901	-	2	3644
	1965	No. 254	2	-	1839
	1966	No. 260	2	-	1097
Tsivolky Fjord	1967	OK-150	3	0.6[a]	18891
Novaya Zemlya trough	1972	No. 421	-	1	884
Stepovogo Fjord	1981	No. 601	-	2	2018
Techeniya Fjord	1988	No. 538	2	-	4242
Total			**10**	**6.6**	**36583**

[a] *- SNF was contained in a screening assembly not a reactor.*

Table 1.5. *Total activity at time of dumping in reactor components dumped in the Barents and Kara Seas as reported in the White Book 2000 (Sivintsev et al., 2005).*

Location	Year of dumping	Description of components	Total activity at time of dumping (TBq)
Stepovogo Fjord	1966	Reactor lids x4	3.7
Olga Fjord	1976	Reactor lids	1.9
Barents Sea	1978	Reactor lids x 7 in barge Nickel	0.7
Abrosimov Fjord	1980	Reactor vessel in lighter L-8711	0.4
Novaya Zemlya trough	1985	Reactor lids x 15 in barge Kureika	1.1
Techeniya Fjord	1988	Screening assembly in Lighter-4	13
Total			**20.8**

1.3 Dumped nuclear waste in Stepovogo Fjord

1.3.1 Nuclear submarine K-27

The nuclear submarine K-27 was designed with two experimental 73 MW VT-RM-1 nuclear reactors with lead-bismuth liquid metal coolants, within the modified hull of a November class submarine. Launched in 1962 and commissioned in 1963, the K-27 suffered a reactor accident on the 24th of May 1968 while on naval exercises. The accident resulted in damage to approximately 20% of the fuel in the port reactor, release of radioactive material to the primary circuit and subsequently to the reactor compartment leading to a number of fatalities amongst the crew from radiation exposure. Following the accident, K-27 remained at Gremikha Fjord while the reactors were cooled before being officially decommissioned in February 1979.

Prior to the final disposal of K-27 the liquid metal reactors and the reactor compartment were subjected to a series of conservation actions (Sivintsev et al., 2005). On the port side, the liquid metal coolant in the reactor and other equipment of the primary circuit was in a "frozen" state at the time of disposal. On the starboard side, the liquid metal coolant was discharged from the primary circuit but maintained at a level of 80 cm above the core in the reactor. Free spaces of the starboard primary circuit, the steam generators of the port side secondary circuit, the lead-water shielding tank, the steam generator enclosures, the enclosures of the primary circuit pumps and the pump box of the leak collection canister were then filled with bitumen. Free spaces of the reactors and associated equipment, specifically the reactor hull cooling and warming coil pipes, steam jackets, reactor water casings, cooling condensers and gaps in the control rod covers were then all filled with furfural. The control rod covers were then cut and plugged and the gaskets of all removable reactor parts were plugged and welded shut. A steel enclosure was then installed and welded to completely cover each reactor lid which was then filled with bitumen. Upon completion of these operations, the remaining free space in the reactor compartment was then filled with bitumen to a height of 70 cm above the reactor lids.

Following these preparations, K-27 was towed to Novaya Zemlya and sunk with an open hatch to the reactor compartment at a depth of 33 m in September 1981 in the outer part of Stepovogo Fjord. According to the White Book 2000 (Sivintsev et al., 2005), the total activity within the 2 reactors at the time of dumping was 2018 TBq.

Table 1.6. *Inventory of reactors of K-27 at time of dumping as reported in the White Book 2000 (Sivintsev et al., 2005).*

	Activity at time of dumping (TBq)
All actinides	1.69
All fission products	1060
All activation products	956
Total	**2018**

Table 1.7. *Inventory of selected radionuclides in reactors of K-27 at time of dumping as reported in the White Book 2000 (Sivintsev et al., 2005).*

	Activity at time of dumping (TBq)
Pu-239	0.49
Pu-240	0.015
Am-241	0.0069
Cs-137	486
Sr-90	496
Co-60	658

1.3.2 Other dumped waste

According to the White Book 2000 (Sivintsev et al., 2005), various other wastes were dumped either in containers or in unpacked states within Stepovogo Fjord. Available coordinates indicate that these wastes were dumped within the inner part of Stepovogo Fjord at various depths. A total of 1917 containers have been reported to be dumped within Stepovogo Fjord, containing a wide range of operational waste ranging from clothing to components with a total activity of 106 TBq (Sivintsev et al., 2005). In addition, the White Book 2000 (Sivintsev et al., 2005) reports the dumping of 4 reactor lids within Stepovogo Fjord with a combined activity of 3.7 TBq and makes reference to unspecified dumping operations of unpacked SRW for which no information on associated activities, contents or number is available.

Table 1.8. *Low-and intermediate-level solid radioactive waste dumped Stepovogo Fjord as reported in the White Book 2000 (Sivintsev et al., 2005).*

Year of dumping	SRW description[1]	No. of containers	No. of unpacked items	Total activity (TBq)	
				At time of dumping	In 2000
1968	Cleaning cloth, overalls, flexible PVC	463	-	15.54	3.73
	Containers of control rod holders	2	-		
	Assembly 18	-	1	0.09	0.02
	Filters S-14	-	2	0.17	0.04
	Unpacked SRW of the icebreaker Lenin[2]	-			
1970	Metal chips, cleaning cloth, overalls	166	-	9.57	2.43
	Unknown SRW	77	-	22.14	5.62
	Unpacked SRW[2]	-			
1972	Unknown SRW	223	-	15.13	4.06
	Unknown SRW	19	-	2.99	0.80
	Unpacked SRW[3]	-			
1973	Steam generators, main circulating pump, individual protective means, removable equipment	270	-	4.89	1.35
	Operational waste from icebreaker	247	-	19.49	5.39
	Unpacked SRW[4]	-			
1975	Scrap metal, parts, tools, removable equipment, cleaning cloth etc	257	-	1.71	0.50
	-	193	-	14.27	4.18
	Unpacked SRW[2]	-			
Total		**1917**	**3**	**106.0**	**28.1**

[1] - *SRW description as given in the White Book 2000 (Sivintsev et al., 2005).*
[2] - *Listed but with no additional information.*
[3] - *Listed but only with a stated volume of 242 m³.*
[4] - *Listed but only with a stated volume of 532 m³ and a Sr-90 equivalent activity of 12 TBq.*

1.4 Other sources of radioactive contamination to the Barents and Kara Seas

1.4.1 Nuclear weapon testing on Novaya Zemlya

In the period between 1955 to 1990, 130 nuclear weapon tests with a total of 265 megatons were conducted at Novaya Zemlya either in the atmosphere (high and low levels), underground, at sea or underwater. In a regional context (such as Stepovogo Fjord), sources of anthropogenic radionuclides from these tests should be considered as possible contributors to environmental concentrations. Atmospheric tests were mainly carried out over the southern part of the northern island, over both the Barents and Kara Sea coastlines, while 33 of a total of 39 underground tests were carried out at the northern tip of the southern island, approximately 90 km from Stepovogo Fjord. A total of 5 nuclear weapon tests (3 underwater, 1 above water and 1 surface) were carried out in Chernaya Fjord on the south western coastline Novaya Zemlya, while 1 above water test was conducted at Bashmachnaya Fjord further to the west. Subsequent studies in Chernaya Fjord have revealed Pu-239,240 sediment concentrations in excess of 15000 Bq/kg along with elevated levels of Cs-137 and Co-60 (Smith et al., 2000). It has been estimated that approximately 11 TBq of Pu-239,240 is present within sediments in Chernaya Fjord, with evidence from Pu-240/Pu-239 ratios of subsequent transport of this plutonium along the southern coastline of Novaya Zemlya (Smith et al., 2000).

1.4.2 Other sources

In addition to contamination arising from dumped nuclear waste and nuclear weapon testing on Novaya Zemlya, the following sources of radioactive contamination continue to contribute to levels of anthropogenic radioactivity in the Barents and Kara Seas:

- Global fallout from atmospheric nuclear weapons testing in the 1950s and 1960s
- Transport by the rivers Ob and Yenisey of radionuclides originating from global fallout and releases from nuclear installations situated within their catchment areas
- Long range oceanic transport of radionuclides discharged from European reprocessing plants at SeIlafield (UK) and Cap la Hague (France)
- Long range oceanic transport of Chernobyl fallout along the Norwegian coast from the Baltic Sea
- The re-entry of the SNAP-9A satellite in 1964

Other potentially significant sources of contamination to the Barents and Kara Seas include the sunken nuclear submarines K-159 in the Barents Sea and "Komsomolets" in the Norwegian Sea.

1.5 Previous investigations of dumped nuclear waste

1.5.1 Overview of earlier investigations

A series of joint Norwegian-Russian investigations into the status of dumped nuclear waste were held in the 1990s, with a cruise to Kara Sea in 1992 followed by cruises to Tsivolky and Stepovogo Fjords in 1993 and Stepovogo and Abrosimov Fjords in 1994 (JRNEG, 1996). The main objectives of these investigations were to locate and identify dumped objects as well as to identify any leakages from the waste via the collection of environmental samples. Numerous objects were localised in Tsivolky, Stepovogo and Abrosimov Fjords during these investigations, which in some cases were observed with considerable corrosion damage. These cruises were followed up in 2002 with a further investigation of Abrosimov Fjord (Niktin et al., 2005) supported by the International Science and Technology Centre (ISTC) and Norwegian-Russian organised cruises in 2003 and 2004 that re-investigated radionuclide contamination in sediments in Tsivolky, Stepovogo and Abrosimov Fjords as well as the Novaya Zemlya trench (Dahle et al., 2009). In addition, a series of Russian led investigations to different dumped waste sites have taken place in the intervening years up to and including 2012. Overall trends from these investigations have shown that seawater

activity concentrations of radionuclides in fjords with dumped waste were broadly similar to those observed in the open Kara Sea with higher activity concentrations observed in bottom waters compared to surface waters. In some cases, elevated activity concentrations in bottom water were attributed to leakages from dumped waste or resuspension of contaminated sediments. For sediments, activity concentrations in cores taken in fjords with dumped waste were similar to those in the open Kara Sea. However in some cases, gross surface sediment samples collected by a ROV close to dumped objects showed highly enriched activity concentrations of Cs-137, Co-60, Sr-90 and Pu-239,240.

***Table 1.10**. Activity concentrations in seawater (<1 μm) from previous investigations*

		Pechora Sea	Kara Sea		Tsivolky Fjord	Abrosimov Fjord		Novaya Zemlya trench
		1992	1992	1993	1993	1994	2002	1993
Cs-137	S	7.4	3.4 - 8.1	5.2 - 5.7	4.5 - 5.2	4.2 - 7.0	0.3 - 3.0	-
(Bq/m^3)	B	6.6	7.8 - 20.4	-	6.9 - 10.0	4.4 - 9.4	2.9 - 4.3	7.4 - 13.5
Cs-134	S	0.23	0.09 - 0.26	0.08 - 0.12	0.12 - 0.16	0.08 - 0.20		-
(Bq/m^3)	B	0.12	0.16 - 0.68	-	0.23 - 0.27	0.08 - 0.28		0.32
Sr-90	S	3.7	3.2 - 11.5	2.6 - 3.1	4.8 - 5.7	1.9 - 3.5	1.2 - 4.9	-
(Bq/m^3)	B	3.4	3.5 - 6.4	-	3.5 - 4.1	2.0 - 3.6	2.9 - 4.3	2.8 - 2.9
Tc-99	S	0.16	0.063 - 0.20	-	-	-	-	-
(Bq/m^3)	B	-	-	-	-	-	-	-
Pu-238	S	-	0.1 - 0.3	-	-	-	-	-
(mBq/m^3)	B	-	0.1 - 1.4	-	0.9	-	-	-
Pu-239,240	S	2.0	1.8 - 7.7	2.6 - 3.8	4.0 - 9.8	3.5 - 6.8	1.2 - 4.9	-
(mBq/m^3)	B	7.4	5.4 - 16	-	5.8 - 8.1	3.5 - 5.1	3.5 - 6.4	7.0 - 12
Am-241	S	-	0.5 - 1.5	-	-	12 - 74	-	-
(mBq/m^3)	B	0.2	0.5 - 1.8	-	-	10 - 28	-	-

S - surface water; B - bottom water.
All data; JRNEG (1996), Salbu et al. (1997) or JRNEG (unpublished), except 2002; Nikitin et al. (2005).

***Table 1.11.** Activity concentrations in surface sediments[1] from previous investigations*

	Pechora Sea	Kara Sea	Tsivolky Fjord		Abrosimov Fjord		Novaya Zemlya trench	
	1992	1992	1993	2004	1994[2]	2004	1993	2004
Cs-137 (Bq/kg d.w.)	10.3 - 10.6	2.2 - 27.4	3.1 - 30.9	1.4 - 11.5	23 - 31000	13.4 - 21.9	12.1 - 29.2	6.5 - 11.0
Co-60 (Bq/kg d.w.)	-	-	0.17 - 1.8	-	0.4 - 180	-	-	-
Sr-90 (Bq/kg d.w.)	-	-	0.4 - 1.0	-	0.3 - 8854	-	0.8	-
Pu-238 (Bq/kg d.w.)	0.089	0.021 - 0.080	0.029	<0.004 - 0.066	0.02 - 17.8	<0.018 - 0.11	0.048	<0.016
Pu-239,240 (Bq/kg d.w.)	2.1	0.3 - 1.4	0.03 - 0.5	0.06 - 0.5	0.02 - 18.0	0.4 - 1.0	0.32 - 0.94	0.44
Am-241 (Bq/kg d.w.)	0.04	-	0.06 - 0.13	-	0.02 - 18.3	-	0.32	-

1 - Includes data from gross surface sediment samples, 0 to 1 cm sediment samples and 0 to 2 cm sediment samples.
2 - Includes sediment samples taken close to dumped objects with ROV.

All data; JRNEG (1996), Salbu et al. (1997) or JRNEG (unpublished), except 2004; Dahle et al. (2009).

1.5.2. Previous investigations in Stepovogo Fjord

In 1993 and 1994, the nuclear submarine K-27 was located in the outer part of Stepovogo Fjord and despite poor visibility underwater, visual inspections of K-27 with an ROV identified an opening in the submarine hull. In 1994, numerous metal containers were detected and observed in the inner part of Stepovogo Fjord at a depth of around 50 m. Some of the metal containers were semi-buried in bottom sediments while others were observed with holes in their outer casings.

In the investigations in 1993 and 1994, similar activity concentrations in surface seawater were observed within Stepovogo Fjord as compared to the open Kara Sea in 1992 and 1993, but activity concentrations of Cs-137 and Sr-90 were elevated in bottom water from the inner part of Stepovogo Fjord. In general, activity concentrations in sediments from the outer part of Stepovogo Fjord were uniform and similar to those observed in the open Kara Sea. However, single sediment samples collected with a ROV close to the nuclear submarine K-27 in 1993 and 1994 showed Cs-137 activity concentrations more than two orders of magnitude higher than other samples. Activity concentrations of Cs-137, Sr-90, Cs-134 and Co-60 in surface sediments from the inner part of Stepovogo Fjord were highly variable but elevated in some samples taken close to dumped objects. Trends in vertical sediment profiles showed typically higher activity concentrations of Cs-137 and Pu-239,240 in the top 5 cm of sediment cores with decreasing activity concentrations down core. Based on Pb-210 measurements, the sedimentation rate at one station in the inner part of Stepovogo Fjord was estimated to be 1.3 mm/year. The presence of radioactive Co-60 particles in sediments from Stepovogo Fjord was confirmed by autoradiography and scanning electron microscopy (SEM).

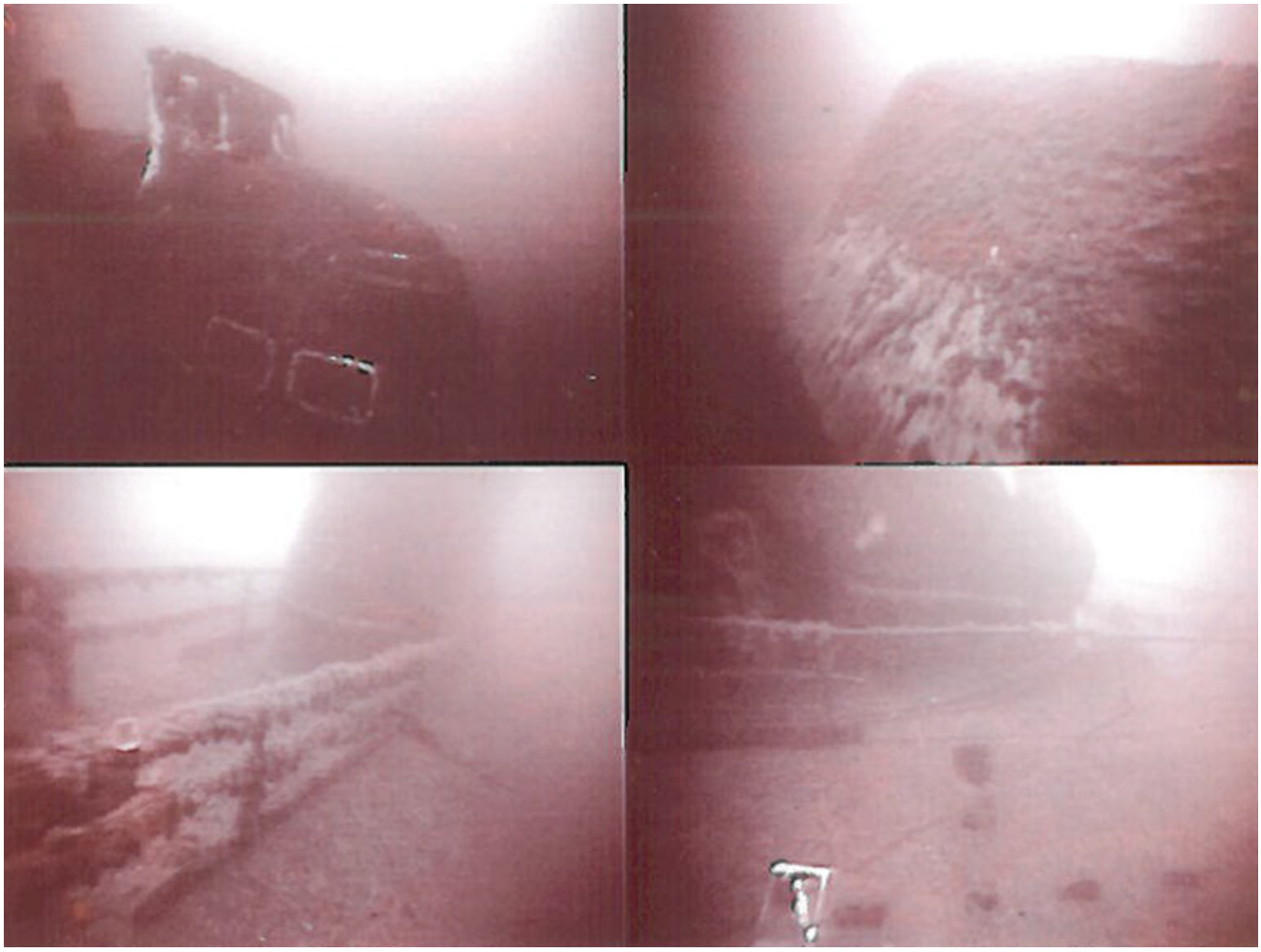


Figure 1.3. *The nuclear submarine K-27 in the outer part of the Stepovogo Fjord in 1990s (Photo JRNEG).*

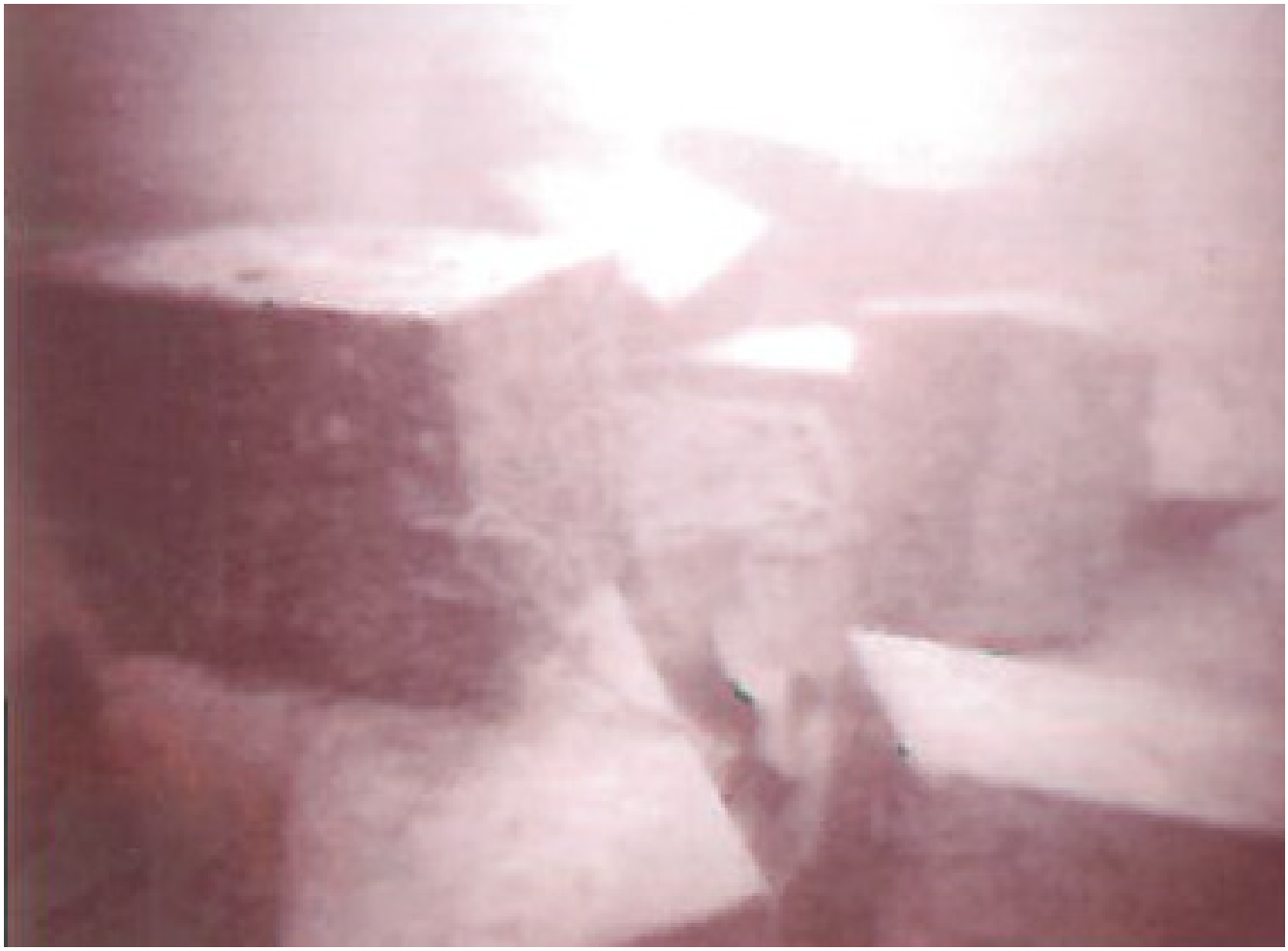


Figure 1.4. *Dumped containers in the inner part of Stepovogo Fjord in 1994 (Photo JRNEG).*

Table 1.12. *Activity concentrations in seawater (<1 μm) from previous investigations in Stepovogo Fjord*

		Entrance to Stepovogo Fjord	Outer part of Stepovogo Fjord		Inner part of Stepovogo Fjord	
		1993	1993	1994	1993	1994
Cs-137	S	5.1	5.2	8.6	5.8	7.4
(Bq/m^3)	B	6.9	8.2	9.5	26.5	16.7
Cs-134	S	0.12	0.14	0.07	0.13	0.06
(Bq/m^3)	B	0.16	0.21	0.16	0.21	0.09
Sr-90	S	5.5	4.9	2.9 - 3.1	5.2	2.4 - 2.9
(Bq/m^3)	B	4.2	4.0	4.2 - 4.7	24.3	3.9 - 5.0
Pu-239,240	S	2.9	2.3	3.6	1.9	4.7
(mBq/m^3)	B	18.0	2.3	4.5	6.0	5.9
Am-241	S	-	1.5	-	2.0	-
(mBq/m^3)	B	5.3	7.7	-	-	-

S - surface water; B - bottom water.
All data; JRNEG (1996), Salbu et al. (1997) or JRNEG (unpublished).

Table 1.13 *Activity concentrations in surface sediments[1] from previous investigations in Stepovogo Fjord*

	Outer part of Stepovogo Fjord			Inner part of Stepovogo Fjord		
	1993[2]	1994[2]	2004	1993[2]	1994[2]	2004
Cs-137 (Bq/kg d.w.)	6.7 - 203	2 - 1670	4.4 - 19.2	34.3 - 289	4 - 109000	19.5 - 268
Cs-134 (Bq/kg d.w.)	0.17 - 0.23	0.3 - 2.4	-	0.32 - 0.47	0.3 - 39.8	-
Co-60 (Bq/kg d.w.)	0.29 - 1.2	<0.1 - 6	-	1.6 - 15.2	<0.3 - 3150	-
Sr-90 (Bq/kg d.w.)	0.4	0.4 - 6	-	2 - 4	1 - 310	-
Pu-238 (Bq/kg d.w.)	-	0.04 - 0.09	<0.008 - 0.031	0.03 - 0.06	0.11 - 6.5	<0.008 - 0.35
Pu-239,240 (Bq/kg d.w.)	0.37	<0.1 - 6	0.27 - 0.62	0.73 - 0.79	0.64 - 28.4	0.29 - 1.1
Am-241 (Bq/kg d.w.)	0.07	-	-	-	1.1 - 2.0	-

1 - Includes data from gross surface sediment samples, 0 to 1 cm sediment samples and 0 to 2 cm sediment samples.
2 - Includes sediment samples taken close to dumped objects with ROV.

All data; JRNEG (1996), Salbu et al. (1997) or JRNEG (unpublished), except 2004; Dahle et al. (2009).

1.5.3 Conclusions from previous investigations

The Joint Norwegian-Russian Expert Group report (JRNEG, 1996) on the investigations in the 1990s concluded that elevated levels of radionuclides in sediments collected close to dumped objects demonstrated that leakages had occurred and that the highest activity concentrations of Cs-137, Co-60, Sr-90, and Pu-239,240 were observed in sediments collected close to dumped containers in Stepovogo and Abrosimov Fjords. Although indications of possible leakage from the reactors in K-27 were observed in outer part of Stepovogo Fjord in 1993, this was not confirmed by subsequent sampling in 1994.

The observed activity concentrations of anthopogenic radionuclides in seawater, sediments and biota in the open Kara Sea could be attributed to global fallout from the atmospheric nuclear weapons tests and marine transport of authorised discharges from European reprocessing plants, fallout from the Chernobyl accident and outflow from the Ob and Yenisey rivers. No additional contribution from dumped radioactive waste was observed, though the report recommended a regular monitoring programme due to the potential for leakage from the range of dumped waste in the future.

Follow up investigations in the period 2002 to 2004 found no evidence of additional leakages from dumped objects with SNF including the reactors in K-27, but noted that sediment activity concentrations in Stepovogo Fjord as well as other dump sites had decreased appreciably compared to the 1990s.

2. Sampling and analytical methods

All sampling work was conducted onboard the R.V. "Ivan Petrov" of the Federal Service for Hydrometeorology and Environmental Monitoring (Roshydromet) during August/September 2012. Samples were collected in the Pechora Sea close to the Kara Strait, in the Kara Sea near the entrance to Stepovogo Fjord and within the inner and outer parts of Stepovogo Fjord (Figure 2.1a and 2.1b).

Figure 2.1. *Sailing route and sampling locations during cruise onboard R.V. "Ivan Petrov"*

Figure 2.1a

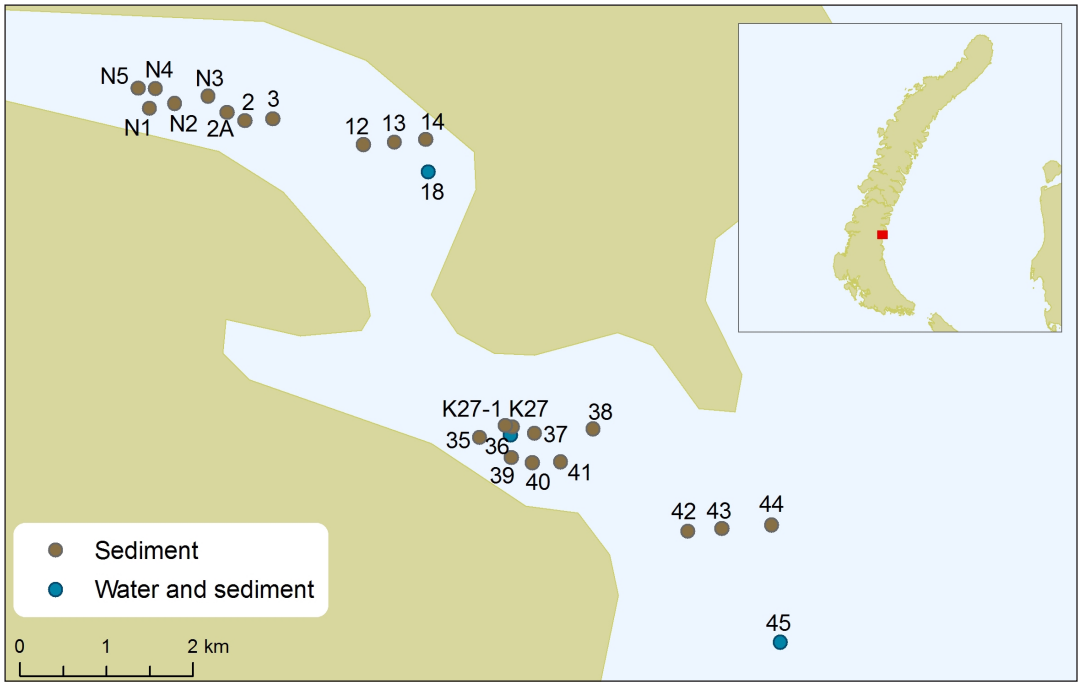


Figure 2.1b

2.1 Hydrographic, sonar and ROV surveys

Temperature and salinity profiles were recorded at all sampling stations (except STEP-NEW2) with a SAIV STD/CTD (model SD204) at a vertical speed of approximately 0.5 m/s. Prior to collection of samples within Stepovogo Fjord, sonar surveys were conducted with a 100 kHz Katran side scan sonar towed at a speed of 2-3 kn. In the outer part of Stepovogo Fjord, sonar surveys were carried out in the vicinity of K-27, while in the inner part of Stepovogo Fjord, sonar surveys were used to identify the location of dumped containers for subsequent investigations. Visual and spectrometric inspections of K-27 and a number of dumped containers were carried out with a RTM-500 Remote Operated Vehicle (ROV) equipped with a video camera, a REM-26 NaI gamma spectrometer and a simple sediment collecting device. Count times for in situ gamma measurements were 50 s to ensure a Cs-137 detection limit of approximately 20 Bq/kg in sediments and 1 Bq/l in seawater. Estimates of Cs-137 activity concentrations in sediments by in situ measurements are based on the assumption that all detected Cs-137 occurs in the sediment and through validation of Monte Carlo N-Particle code calibrations.

Figure 2.2. Katran side scan sonar and RTM-500 Remote Operated Vehicle (ROV) with attached REM-26 NaI gamma spectrometer (Photo NRPA).

2.2 Seawater sampling

Seawater samples were collected in the Pechora Sea (St. 1), at one station in the Kara Sea near the entrance of Stepovogo Fjord (St. 45) and at one station in both the inner (St. 36) and outer (St. 18) parts of Stepovogo Fjord. At each seawater sampling station, samples were taken from the surface and at specified depths close to the sea floor with the use of pumps. An additional intermediate depth sample was collected in the Pechora Sea (Table 2.1). Unless stated, all seawater samples were pre-filtered through 1 µm filters before further handling and all seawater samples were stored on deck. The temperature and salinity of collected seawater was checked with a portable temperature/salinity meter and exact volumes were determined with the use of flowmeters or graduated cans.

Table 2.1. *Sampling depths at seawater stations in the Pechora Sea and Stepovogo Fjord.*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer Part (St. 36)	Inner Part (St. 18)
Sampling	S	1	1	1	1
depths	M	25	-	-	-
(m)	B	80	30[a]/36[b]	30	43[a]/46[b]

S - surface water; M - mid-depth water; B - bottom water.
a - As recorded by Norway; b - As recorded by Russia.

2.2.1 Processing of seawater samples onboard by Norway

For Cs-137, a Cs-134 yield tracer was added to 300 l of pre-filtered seawater and allowed to mix thoroughly. The sample was continuously pumped through a 1 µm filter impregnated with cupric hexacyanoferrate at a flow rate of approximately 3.6 l/min for at least 2 hr. The filter was then allowed to drip dry, before drying at 60 ˚C.

For Pu-238, Pu-239,240 and Am-241, 300 l of pre-filtered seawater was acidified to pH 2 with concentrated HCl, followed by the addition of a combined Pu-242/Am-243 yield tracer, 2 g of $FeCl_3 \cdot 6H_2O$ and 100 g of $K_2S_2O_5$. The sample was then allowed to mix for at least 1 hr by aeration, before increasing the pH to 10 with the addition of 6 M NaOH. The sample was then allowed to stand for at least 12 hr to allow the precipitate to settle. The supernatant was carefully removed, to allow collection of the precipitate in a volume of less than 10 l in a plastic can. The processing tank was washed with a small volume of 0.1 M HCl and distilled water, with each washing added to the collected sample. Finally, the precipitate in each sample was re-dissolved by the addition of concentrated HCl at approximately 15 ml/l.

For Sr-90 (100 l) and U isotopes (25 l), pre-filtered seawater was collected in 25 l plastic cans and acidified to pH 2 with concentrated HCl.

For Tc-99, a standard rhenium solution yield tracer was added to 100 l of unfiltered seawater in four 25 l plastic cans and allowed to mix thoroughly. The sample was then pumped through Amberlite IRA-400 anion exchange resin in 60 ml syringes at a flow rate of 33 ml/min.

In parallel to the aforementioned sampling, size and charge fractionation techniques were carried out in order to obtain information on the speciation (physico-chemical forms) of anthropogenic (Pu, U-236 and, I-129) and naturally occurring radionuclides (e.g. Po-210, U-235 and U-238), trace metals (e.g. Fe and Cr) and metalloids (e.g. As). For 0.45 µm filtration, unfiltered seawater was used, while pre-filtered seawater (1 µm) was used prior to ultrafiltration. Through successive filtration (0.45 µm Millipore nitrocellulose filters) and ultrafiltration (large size Pall hollow fiber cartridges of 3 and 10 kDa nominal cut-off, respectively) interfaced with ion chromatography (Chelex-100 cation exchange resin and AG-1 anion exchange resin) the following fractions were obtained:

- Particles > 0.45 µm
- Pseudo-colloids 10 kDa - 0.45 µm
- Colloids 3 - 10 kDa
- Low molecular mass (LMM) forms < 10 kDa and < 3 kDa
- Positively charged LMM forms < 10 kDa as cations (Pu, trace elements)
- Negatively charged LMM forms < 10 kDa as anions (I-129, U, trace elements)

Seawater samples (total and fractions) for Pu and U analysis were reduced in volume from about 200 l to 10-25 l by chemical precipitation (as described previously) with Pu-242 used as a yield tracer. Samples for I-129/I-127 were collected in 1 l plastic bottles. Samples for stable I-127 were sampled in 100 ml bottles and stored at -20 ˚C to avoid loss of volatile iodine. Samples (50 ml) collected for U-235, U-238 and trace element analysis were acidified with 2.5 ml of concentrated ultrapure HNO_3.

2.2.2 Processing of seawater samples onboard by Russia

For Cs-137, large volumes of pre-filtered seawater (500 - 3200 l) were pumped through cartridges containing fibers (Mtilon-T) impregnated with cupric hexacyanoferrate at flow rates up to 500 l/h.

For Pu-238 and Pu-239,240, 100 l of pre-filtered seawater was acidified to pH 2 with concentrated HCl, followed by the addition of a Pu-242 yield tracer and 200 g of Na_2SO_3. The sample was stirred periodically for 12 hr prior to the addition of a solution of $FeCI_3$ (1 g Fe^{3+} per 100 l) and then stirred periodically for a further 2-3 hr. A NH_4OH-solution was then added stepwise until the pH of the sample reached 8.5-9. The sample was then left for 12-24 hr to allow the precipitate to settle. The supernatant was carefully removed, to allow collection of the precipitate in a volume of 1.0-1.5 l in a plastic bottle.

For Sr-90, 10 l samples of pre-filtered seawater were collected in plastic bottles and 120 g of Na_2CO_3 was added and stirred for 30 min. After 24-48 hr, the Sr-, Ca-, Mg-carbonate precipitate was then collected by vacuum filtration on a filter. Corresponding pre-filtered seawater samples (250 ml) were collected for determination of stable strontium.

For H-3, 1 l of pre-filtered seawater was collected in a plastic bottle.

2.2.3 Processing of seawater samples onboard by the IAEA

For Cs-137, Sr-90, Pu- and Am-isotopes, a single sample of 60-90 l of unfiltered seawater was filtered through a 0.45 µm membrane filter into 30 l plastic containers and acidified to pH 2 with concentrated HCl. Additionally unfiltered seawater was collected for determination of iodine (1 l) and tritium (1 l).

Figure 2.3. *Processing of seawater samples onboard (Photo NRPA).*

2.3 Sediment sampling

Where conditions allowed, sediment samples were collected using Smøgen box corers with an inner area of 30 x 30 cm. In some locations where the seafloor was covered with stones, sediment samples were obtained using a Petersen grab. In addition, surficial sediments were collected with the ROV close to K-27 and one dumped container.

For sediment samples collected using Smøgen box corers, 2 to 3 sediment cores per sample were obtained with plastic tubes with an inner diameter of 10 cm. Cores were cut either into 1 cm (0-10 cm) and 2 cm (the remaining core) slices or all as 2 cm slices. In addition, samples of surface sediments (upper 3 cm) were taken from all Smøgen box corers as well as from all samples obtained with a Petersen grab. Norwegian samples were stored frozen at -20 °C until analysed, while Russian samples were stored unfrozen.

Figure 2.4. *Deployment of box corer and cutting of sediment core (Photo NRPA).*

2.4 Biota sampling

Samples of biota were principally collected within the inner and outer parts of Stepovogo Fjord using fishing nets, eel traps and baited pots. Biota samples were also collected opportunistically directly from sediment samples and through the use of a three angled bottom dredge in areas adjacent to K-27 in the outer part of Stepovogo Fjord. In addition, a number of fish samples were collected using fishing lines at station 1 in the Pechora Sea. Biota samples collected included various fish species, seaweed, molluscs, crustaceans and other benthic biota, as well as one example of a ringed seal *(Pusa hispida)* that had become entangled in a fishing net.

The species, length, weight, sex and maturation stage of fish samples were recorded, before being dissected to remove the secondary gill arch, stomach, liver, kidney, gonads, bone and muscle samples according to procedures in the EMERGE protocol (Rosseland et al., 2001). Otoliths were removed for age determination and stomach contents were later characterised. In total, 6 cod *(Gadus morhua)*, 23 navaga *(Eleginus nawaga)*, 20 sculpin (Cottidae spp.), 4 long rough dab *(Hippoglossoides platessoides)* and 4 arctic eelpout *(Lycodes reticulatus)* were dissected. The ringed seal was dissected to remove the stomach, liver, kidneys and a muscle sample. Examples of benthic biota were rinsed in seawater to remove any sediment prior to storage. All biota samples were stored frozen at -20 ˚C until analysed.

Figure 2.5. *Retrieval of eel traps (Photo NRPA).*

3. Analytical methodologies

3.1 By Norway

3.1.1 Determination of gamma emitters

In the laboratory, sediment samples were freeze-dried and ground, while biota samples were either freeze-dried or dried at 105 ˚C until constant weight. Cs-137 filters were ashed at 450 °C in a muffle furnace. All samples were homogenized before being packed into standard plastic counting geometries and counted on high-resolution gamma spectrometers (HPGe). Spectra were collected for periods of between 1 and 4 days.

3.1.2 Determination of Pu isotopes and Am-241

Pu isotopes and Am-241 were determined from precipitated seawater samples, sediment and biota according to the procedure described by IAEA (1989), Clacher (1995) or Lusa et al. (2009). For sediment, 8-10 g of dried material was ashed overnight at 450-550 ˚C prior to chemical digestion and separation. For biota, 8-20 g of dried, ground and homogenized material was chemically digested before separation. Pu-242 and Am-243 were added as yield tracers to sediment and biota samples. Where uranium isotopes were determined in the same samples, U-233 was added as a yield tracer. After chemical separation, plutonium and americium fractions were electrodeposited onto stainless steel discs and their activity determined by alpha spectroscopy on semiconductor silicon detectors.

Pu atom ratios were determined by accelerator mass spectrometry (AMS) as described by e.g. Wendel et al. (2013). Following the addition of Pu-242 as yield tracer, ultraclave digestion with concentrated HNO_3 and separation by anion exchange (Clacher, 1995), samples were evaporated to dryness, dissolved in concentrated HNO_3 with 2 mg Fe as $Fe(NO_3)_3$ and baked at 500 °C overnight to give prepared samples for AMS analysis.

3.1.3 Determination of Tc-99

Tc-99 in seawater and biota was determined according to the procedures described by Harvey et al. (1992). In the laboratory, the anion exchange resins from seawater samples processed onboard were ashed at 450 °C. For Tc-99 in biota, 10 g of dried homogenized sample was ashed at 450 °C with $KReO_4$ as a yield tracer. In both cases, ashed material was then dissolved in 6M HCl in the presence of Fe^{3+} and H_2O_2. After iron hydroxide scavenging, Tc-99 and Re were extracted by anion-exchange and subsequent sulphide precipitation. Tc-99 and Re, as their tetraphenyl arsonium salts, were then filtered off onto pre-weighed hydrophobic-edged membrane filters. Filters with precipitates were dried overnight, weighed and mounted onto plastic discs and counted on low-background anti-coincidence GM beta counters. The yield of the rhenium tetraphenyl arsonium salt was determined gravimetrically.

3.1.4 Determination of Sr-90

Sr-90 in seawater was determined according to the standard fuming nitric acid method (Sutton & Kelly, 1968) through its daughter nuclide Y-90, with Sr-85 as a yield tracer. For biota, dried homogenised samples were first ashed at 450 °C, before Sr-85 was added as a yield tracer. Samples were then treated with Aqua Regia and Sr purified using Sr-resin (Eichrom). In all cases, recovery of the daughter nuclide Y-90 was determined by titration with EDTA according to the method of Varskog et al. (1997), followed by analysis of Y-90 on low background anti-coincidence GM beta counters.

3.1.5 Determination of H-3

Seawater samples were first distilled before 8 ml of the distillate was combined with ULTIMA GOLD LLT (PerkinElmer) prior to beta activity determination using liquid scintillation on a Quantulus liquid scintillation counter.

3.1.6 Determination of U isotopes

Uranium isotopes were determined in seawater samples according to a procedure adapted from Martin and Hancock (2004) with U-232 as a yield tracer. Uranium isotopes were separated by initial co-precipitation with $Fe(OH)_3$, before the precipitate was re-dissolved in 8 M HNO_3 and extracted into TBP (tri-butyl phosphate) together with thorium. The TBP solution was then diluted with xylene and thorium removed by repeated washing with 1.5 M HCl, before uranium isotopes were back extracted with distilled water. After chemical separation, uranium fractions (including yield tracers) were then electrodeposited onto stainless steel discs and their activity determined by alpha spectroscopy on semiconductor silicon detectors. Additionally, the concentration of U and U-235/U-238 atom ratios were determined by ICP-MS (See 3.19).

3.1.7 Determination of I-129, I-127 and iodine speciation

For speciation analysis of I-129 and I-127, iodate and iodide species were separated by anion exchange. Each sample (500 ml) was loaded onto an anion exchange column with around 2000 Bq of I-125 as a yield tracer. The iodate eluted fraction was collected and combined with wash solutions of 30 ml of 0.2 M $NaNO_3$ and 20 ml of ultrapure water. Iodide was eluted from the column with 50 ml of 5% NaClO and 30 ml of 3 M HNO_3. The elution yield was calculated by comparing I-125 counts in the eluate to the I-125 solution used to spike the sample.

I-127 was determined in 1 ml of the original seawater sample and 1 ml of the separated iodide and iodate fractions. All samples were diluted with 1% $NH_3{\cdot}H_2O$ to 10 ml, before cesium was added to a final concentration of 2 ng/g as an internal standard to monitor the ionization efficiency of iodine. I-127 in the diluted solutions was measured by ICP-MS (Thermo Fisher, X Series II).

Total I-129 was determined from a volume of between 100-200 ml of seawater. Following the addition of around 2000 Bq of I-125 as a yield tracer and 1.975 mg of I-127 as a carrier, $K_2S_2O_8$ was added to the sample to a final concentration of 30 mg/g and the sample heated at 60 °C for 20 hours to convert all organic iodine to an inorganic form. I-127 as a carrier (1.975 mg) was added to all iodate and iodide fractions prior to solvent extraction. Iodine in the decomposed seawater, iodate and iodide fractions were separated by extraction using $CHCl_3$ after reduction with 1 M $K_2S_2O_5$ and oxidation with 1M $NaNO_2$. Iodide in the back-extracted aqueous solution was precipitated as AgI using 0.5 M $AgNO_3$, dried at 70 °C and mixed with niobium powder and pressed into a copper holder. I-129 in the prepared AgI target was measured by AMS.

3.1.8 Digital autoradiography

In order to identify radioactive heterogeneities (particles) in sediment samples and benthic organisms, small aliquots of dried sediment core slices (0.2 to 2.1 g) or rinsed (4 x 100 ml distilled water), dried (60 °C) and coarsely ground soft tissues (0.2 g) of two mussel specimens *(Mya arenaria)* were subjected to digital autoradiography. Storage phosphor screens (Molecular Dynamics) were exposed to samples for 5 or 14 days in a (low activity) lead chamber, whereupon the plates were scanned within 1 hour with a Typhoon 8600 digital image scanner (resolution 200 µm).

3.1.9 Determination of trace elements

Trace elements (including U) in all samples were determined in the laboratory by ICP-MS (Agilent ICP-MS 8800 QQQ) from triplicate 50 ml fractionated seawater samples acidified with 5% ultrapure HNO_3 and in gills and livers of fish and in sediments by ultraclave digestion of freeze dried samples with 10% ultrapure HNO_3. The accuracy of the measurements was controlled using the standard reference materials; CLASS 5, to control seawater measurements; Dolt4, to control measurements of fish (gill and liver) samples; NCS DC 73324 and IAEA384, to control measurements of sediments.

3.2 By Russia

3.2.1 Determination of gamma emitters

In the laboratory, sediment samples were dried at 70 ˚C, while Cs-137 filters and fiber sorbents were ashed at 350 °C. All samples were homogenized before being packed into standard plastic counting geometries and counted on high-resolution gamma spectrometers (HPGe). Spectra were collected for periods of between 5 and 32 hr.

3.2.2 Determination of Pu isotopes

Separation of Pu isotopes from precipitated seawater samples was completed in the laboratory by ion-exchange (Procedure, 2004). Pu isotopes in sediment and suspended matter were determined by acid digestion and ion-exchange with Pu-242 as a yield tracer (Procedure, 2004). After chemical separation, plutonium fractions were electrodeposited onto stainless steel discs and their activity determined by alpha spectroscopy on semiconductor silicon detectors.

3.2.3 Determination of Sr-90

Separation of Sr-90 from precipitated seawater samples was completed in the laboratory by co-precipitation (Guidelines, 1986). The yield of the strontium was determined by atomic absorption spectroscopy. Determination of Sr-90 in sediment samples was performed by acid digestion and co-precipitation. The activity of Sr-90 in both cases was determined via Y-90 after ingrowth and separation with a stable yttrium carrier on low background anti-coincidence GM beta counters.

3.2.4 Determination of H-3

H-3 in seawater was determined by distillation and electrolytic enrichment of samples (Methods, 1995), prior to analysis on a low background liquid scintillation beta spectrometer.

3.3 By the IAEA

3.3.1 Determination of gamma emitters, Pu isotopes and Am-241 in sediment

In the laboratory, 0.5 g of fresh sediment from each sample was sieved at 300 μm and then subjected to particle size analysis. The remaining amount of each sediment sample was freeze-dried and then ground, before being packed into standard plastic counting geometries and counted on high-resolution gamma spectrometers (HPGe).

Pu isotopes and Am-241 were determined in sediment samples according to the procedure described by La Rosa et al. (2001). 7-10 g of dry ground sediment was ashed at 550 ˚C and then Pu-242 and Am-243 added as yield tracers. Soluble forms of Pu and Am isotopes were leached with concentrated HNO_3 and purified using ion-exchange, co-precipitation and liquid-extraction. After chemical separation, plutonium and americium fractions were electrodeposited onto stainless steel discs and their activity determined by alpha spectroscopy on semiconductor silicon detectors.

3.3.2 Determination of Sr-90, Cs-137, Pu isotopes and Am-241 in seawater

In the laboratory, each acidified seawater sample was spiked with Pu-242, Am-234, Cs-134 (with stable Cs as a carrier) and stable Sr as yield tracers. Sequential precipitation was used for pre-concentration of radionuclides with Pu and Am precipitated with MnO_2 (La Rosa et al., 2001), Cs isotopes adsorbed with AMP (Molero et al., 1993) and Sr isotopes precipitated as Ca/Sr oxalate (La Rosa et al., 2002). Pu and Am isotopes were purified using ion-exchange, co-precipitation and liquid-extraction (La Rosa et al., 2001). After chemical separation, plutonium and americium fractions were electrodeposited onto stainless steel discs and their activity determined by alpha spectroscopy on semiconductor silicon detectors.

Cs isotopes were purified by dissolution of AMP precipitates in 10 M NaOH and secondary scavenging with small amount of AMP (Molero et al. 1993, La Rosa et al. 2001). The final precipitate was re-dissolved in a few ml of 10 M NaOH, diluted with distilled water to the required geometry volume and then counted on high-resolution gamma spectrometers (HPGe).

Ca/Sr oxalate was ashed at 500 ˚C and Sr was purified by subsequent precipitation of Ca/Sr nitrate, Ba chromate and Fe/Y hydroxide. Y was re-precipitated after its ingrowth as Y oxalate and Y-90 decay data was obtained using gas-flow proportional beta-counters for Sr-90 determination (La Rosa et al., 2002).

3.3.3 Particle size analysis

Aliquots of approximately 0.5 g of fresh sediment were first sieved at 300 μm before added as a slurry to 500 ml of distilled water. These solutions were then pumped and circulated through a Malvern Mastersizer Micro v2.12 system and the size analysis of particles determined by laser light scattering.

3.4 Data handling and quality control

Unless otherwise stated, data is reported as individual values with associated uncertainties (2 sigma), as individual mean values of Norwegian, Russian and IAEA values (where available) for samples taken at the same sampling location with their propagated uncertainties (2 sigma) or as means of individual or mean values with associated standard deviations. Activity ratios based on individual measurements are reported with their propagated uncertainties (2 sigma) or with associated standard deviations where stated as mean values. Sources of data for individual measurements, mean values and activity ratios (i.e. Norway, Russia or IAEA) are indicated in all data tables and figures. Data for Pu-238 in seawater samples are not reported due to all results being below or close to detection limits. Data values reported for Pu-239,240 in seawater filtered through <1 µm are based on Norwegian data only due to the larger sampling volume employed. Russian data for Pu-239,240 in seawater filtered through <1 µm showed similar trends for all sampling stations as Norwegian data. Data for Cs-137 and Sr-90 in seawater filtered through either <1 µm or <0.45 µm filters are assumed to be comparable, but data for Pu-239,240 and Am-241 in seawater filtered through <0.45 µm filters are reported separately. Data reported as 'surface sediments' is based on gross surface samples (assumed to represent 0 to 2 cm), 0 to 2 cm layers of cores sliced into 2 cm layers and derived 0 to 2 cm values from the layers 0 to 1 cm and 1 to 2 cm sliced into 1 cm layers. In the case of Pu isotopes and Am-241, data from 0 to 1 cm layers of cores has been included in 'surface sediment' calculations where data for the 1 to 2 cm layer was not available.

Sediment distribution coefficients (K_d) and bioconcentration factors for biota (BCF) were derived using activity concentration data from seawater samples filtered through <0.45 µm filters with the exception of BCFs for Tc-99 where activity concentration data from unfiltered seawater was used. BCFs were derived from bottom water activity concentrations in all cases with the exception of seaweed where surface water concentrations were used. A dry weight fresh weight ratio of 0.2 was used for derivation of BCFs for all seaweed samples.

K_ds were derived by: K_d = Activity concentration in sediment (Bq/kg d.w.)/Activity concentration in seawater (Bq/l)
BCFs were derived by: BCF = Activity concentration in biota (Bq/kg f.w.)/Activity concentration in seawater (Bq/l)

A sediment sample from the inner part of Stepovogo Fjord (St. 18) was prepared for intercomparison purposes. The sample was mixed onboard and divided into samples for Norway, Russia and the IAEA. The Norwegian sample was mixed again (but not homogenized) in the laboratory before further subdivision into 4 samples for each institute. Results from the different laboratories for activity concentrations of Cs-137 show a range of activity concentrations between 34.3 and 45.6 Bq/kg (d.w.), with a standard deviation of 10% for the mean value of 39.1 Bq/kg (d.w.). Activity concentrations of Cs-137 in other surface sediments or cores from the same grab or box corer and analysed by more than one laboratory typically showed comparable results. Results for two laboratories for Sr-90 were similar with regard to the associated uncertainties on the measurements. Results from the different laboratories for activity concentrations of Pu-239,240 and Am-241 showed very good agreement with standard deviations of 3% and less than 1% respectively.

Table 3.1. *Intercomparison of analytical results from Norwegian, Russian and IAEA laboratories.*

	Activity concentrations in surface sediment from St. 18 (Bq/kg d.w.)				
	Cs-137	Sr-90	Pu-238	Pu-239,240	Am-241
NRPA	45.6 ±3.6	-	<0.008	0.34 ±0.05	0.26 ±0.04
IMR	41.3 ±1.4	-	-	-	-
IFE	34.3 ±2.1	0.45 ±0.13	0.018 ±0.010	0.31 ±0.05	0.26 ±0.06
NMBU	40.2 ±1.2	-	-		
RPA Typhoon	35.0 ±1.0	0.63 ±0.13	-	0.33 ±0.05	-
Krylov SRC	38.2 ±2.3	-	-	-	-
IAEA	39.2 ±3.9	-	0.018 ±0.005	0.38 ±0.02	0.27 ±0.02

4. Results of investigations in Stepovogo Fjord

4.1 Sedimentology and oceanography

Stepovogo Fjord is characterized by distinct outer and inner parts which are partially separated from the open Kara Sea by an underwater sill at the entrance to the fjord. The outer and inner parts of Stepovogo Fjord are divided by a second underwater sill of a depth around 20 m. The outer part reaches maximum depths of around 30 to 40 m, while the inner part reaches depths up to 60 m. Sediments sampled in the outer and inner parts of Stepovogo Fjord generally have a similar composition with a high mud content (89% to 95%) and would therefore be expected to have a higher adsorption capacity compared to the coarser and less uniform sediments from the entrance of Stepovogo Fjord (Table 4.1).

Table 4.1. *Particle size analysis of gross surface sediments from Stepovogo Fjord in 2012[1].*

	n	%Sand	%Silt	%Clay	%Mud
Entrance to Stepovogo Fjord	4	59 ±22	33 ±16	8 ±7	41 ±22
Outer part of Stepovogo Fjord	3	11 ±2	58 ±1	30 ±1	89 ±2
Inner part of Stepovogo Fjord	11	5 ±2	65 ±5	29 ±6	95 ±2

Sand 300 µm to 63µm; silt 63 µm to 3.9 µm; clay <3.9 µm; mud - clay and silt fractions combined. n - number of sampling stations.

1 - IAEA data (Mean ±SD)

With regard to the physical oceanography of Stepovogo jord, the most distinct features are the clear layering of the water masses (Figure 4.1). At the entrance to Stepovogo Fjord and in the outer part there was a pycnocline present at 10 to 15 m due mainly to a distinct halocline as well as a decrease in water temperature. The upper 10 m in both the outer and inner parts of Stepovogo Fjord were stratified, although the water temperature is homogeneous in these surface layers, with the upper mixed layer better defined in the inner part compared to the outer part. In the inner part of Stepovogo Fjord, there was a second distinct pycnocline at between 30 to 40 m, below which the water masses were more or less well mixed. This second deeper pycnocline was not present in the outer part of Stepovogo Fjord, although a steady increase in salinity accompanied by a steady decrease in temperature with depth resulted in the density increasing more or less monotonically towards the bottom. There was no sign of a well-mixed bottom layer in the outer part of Stepovogo Fjord. Lower salinity seawater in the surface layers of Stepovogo Fjord is likely to derive in the main from the Kara Sea, while higher salinity bottom water is likely to result from brine release during formation of sea-ice during the winter. This was particularly obvious in the inner part of Stepovogo Fjord where the underwater sill between the inner and outer parts of Stepovogo Fjord prevents the flushing of the bottom water. As a consequence, the bottom water in the inner part of the fjord is probably only renewed through winter convection. The different structures to the vertical temperature and salinity gradients may indicate that diffusive processes dominate over turbulence in terms of mixing. However, as temperature-salinity plots (Figure 4.2) indicated mostly diapycnal mixing at all stations, internal waves at the interfaces between different water mass layers may contribute to mixing processes.

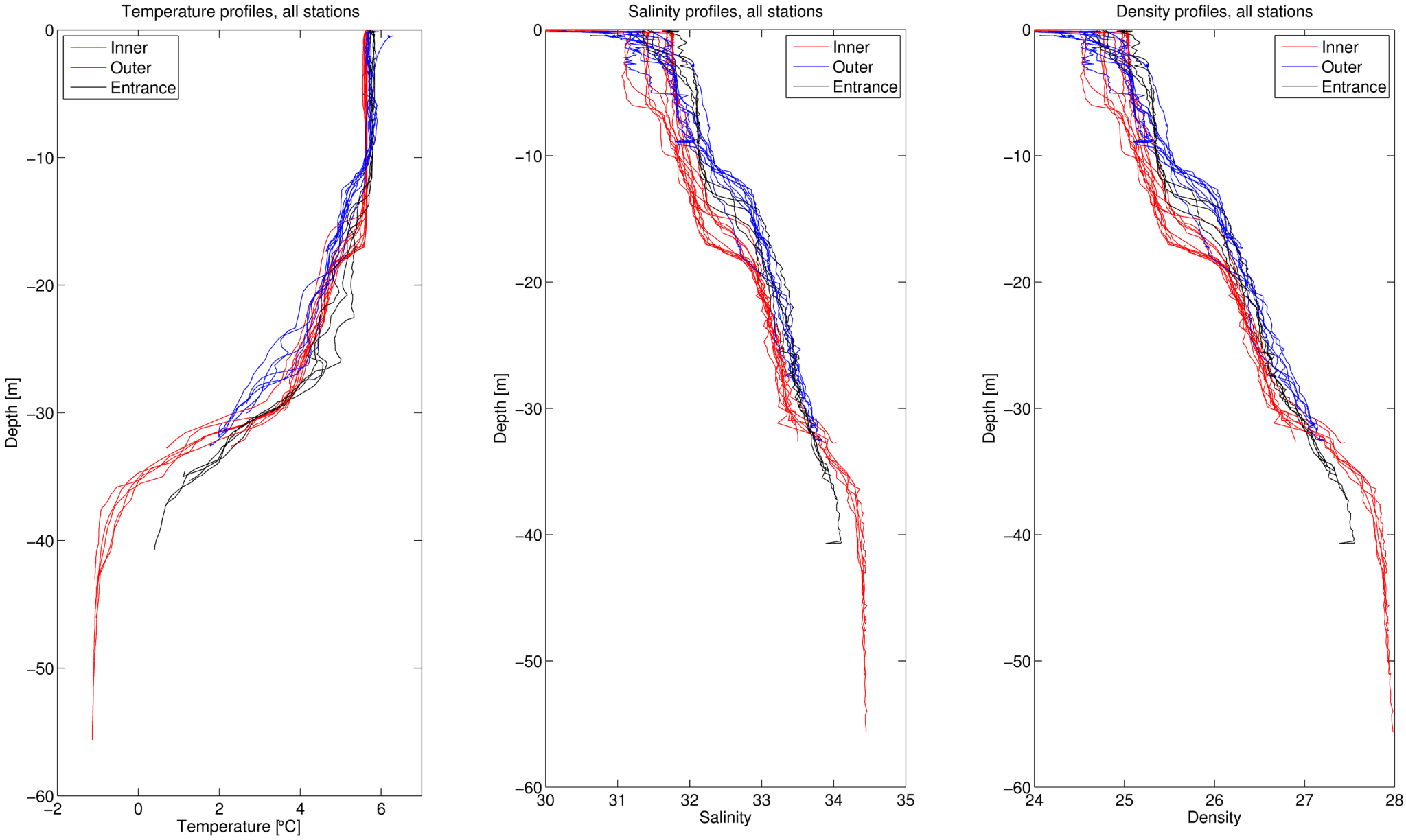


Figure 4.1. *Temperature, salinity and density plots for stations at the entrance (black), outer part (blue) and inner part (red) of Stepovogo Fjord.*

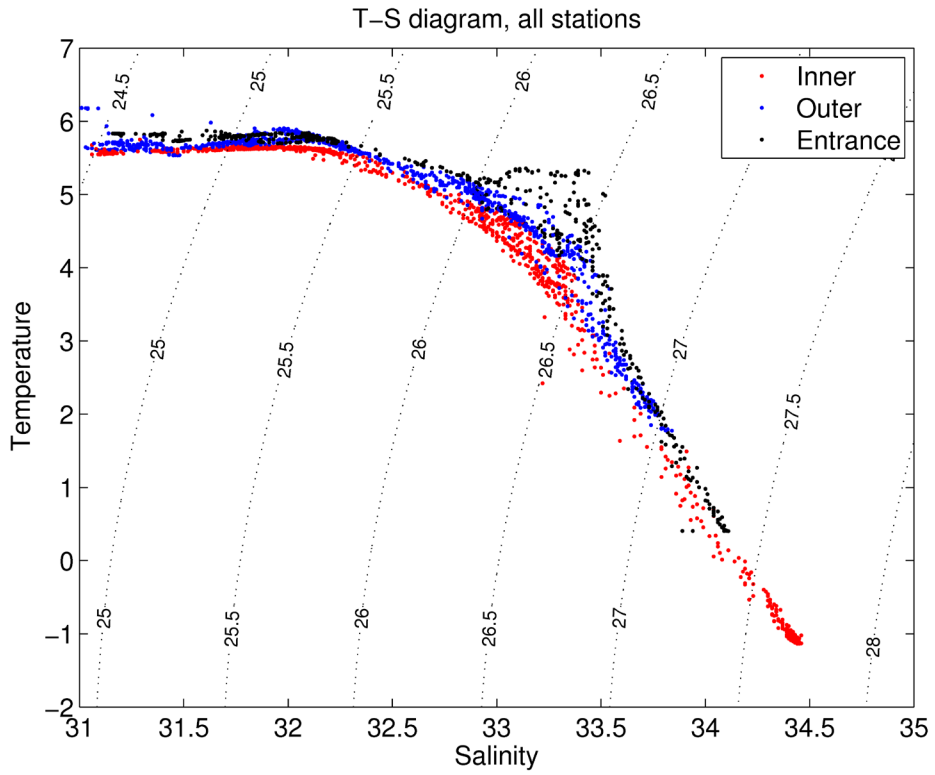


Figure 4.2. *Temperature-salinity plot of all stations at the entrance (black), outer part (blue) and inner part (red) of Stepovogo Fjord.*

4.2 Sonar surveys, visual inspection and in situ gamma measurements of dumped objects

The position of the nuclear submarine K-27 in the outer part of Stepovogo Fjord was confirmed by sonar survey, whilst dumped containers were detected at several positions along the length of the inner part of Stepovogo Fjord (Figure 4.3).

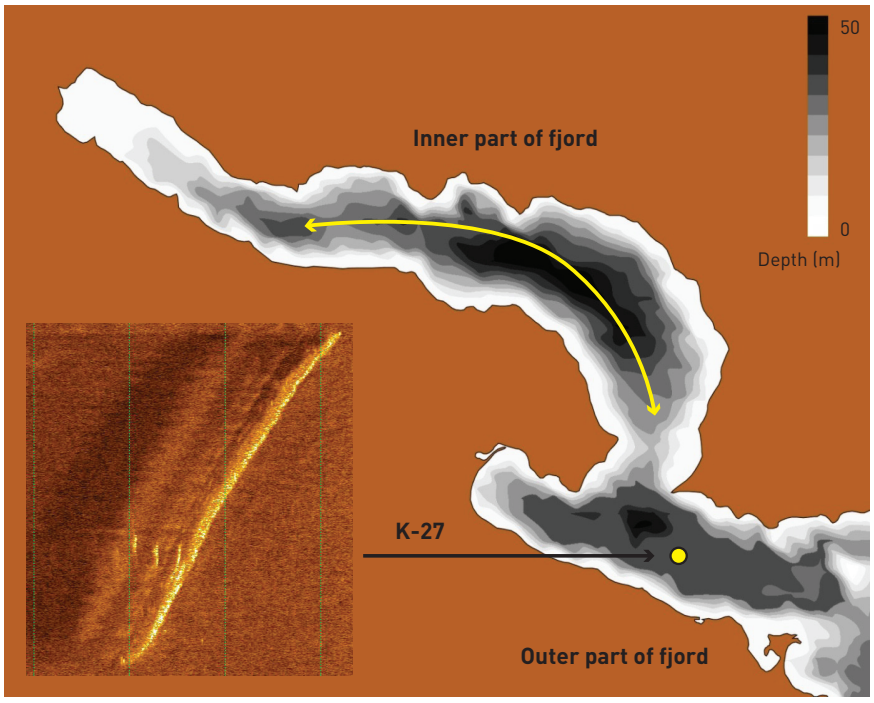


Figure 4.3. *Location of nuclear submarine K-27 (sonar image inset) in the outer part of Stepovogo Fjord and the region of the inner part of Stepovogo Fjord where dumped objects were detected by sonar survey (yellow arrow) in 2012.*

The nuclear submarine K-27 was observed with the ROV lying upright and clear of bottom sediments at a depth of around 30 m in the outer part of Stepovogo Fjord (Figure 4.4). No obvious corrosion damage of the outer hull was observed, with markings still discernible on the bow and forward and aft anchors deployed. A number of hatches in the outer hull were missing, but it is unknown whether these were removed before the K-27 was sunk or whether they have subsequently disappeared. It was not possible to inspect the status of the interior pressure hull. The deck of K-27 has been covered with a 3 to 5 cm layer of sediment which has been colonized by bivalve molluscs and other benthic organisms.

Bow of K-27

Stern of K-27

Tower (front view)

Tower (back view) and rails

Stern hatch

Bow anchor

Figure 4.4. *The nuclear submarine K-27 in the outer part of the Stepovogo Fjord.*

Due to technical problems, only a limited number of dives with the ROV were possible in the inner part of Stepovogo Fjord to investigate the status of dumped containers. A few containers were observed with the ROV at depths of up to 50 m embedded in soft sediments (Figure 4.5). The containers that were examined appeared to be intact and were observed with various benthic organisms attached to the sides of the containers.

Figure 4.5. *Dumped containers on the seafloor in the inner part of Stepovogo Fjord*

The series of in situ gamma measurements performed around the nuclear submarine K-27 showed that gamma-radiation levels in the near environment had not changed since previous measurements made during a Russian survey in 2006. A clear peak for Cs-137 in collected spectra was only visible when in situ measurements were taken directly over the deck of the reactor compartment of K-27 (Figure 4.6).

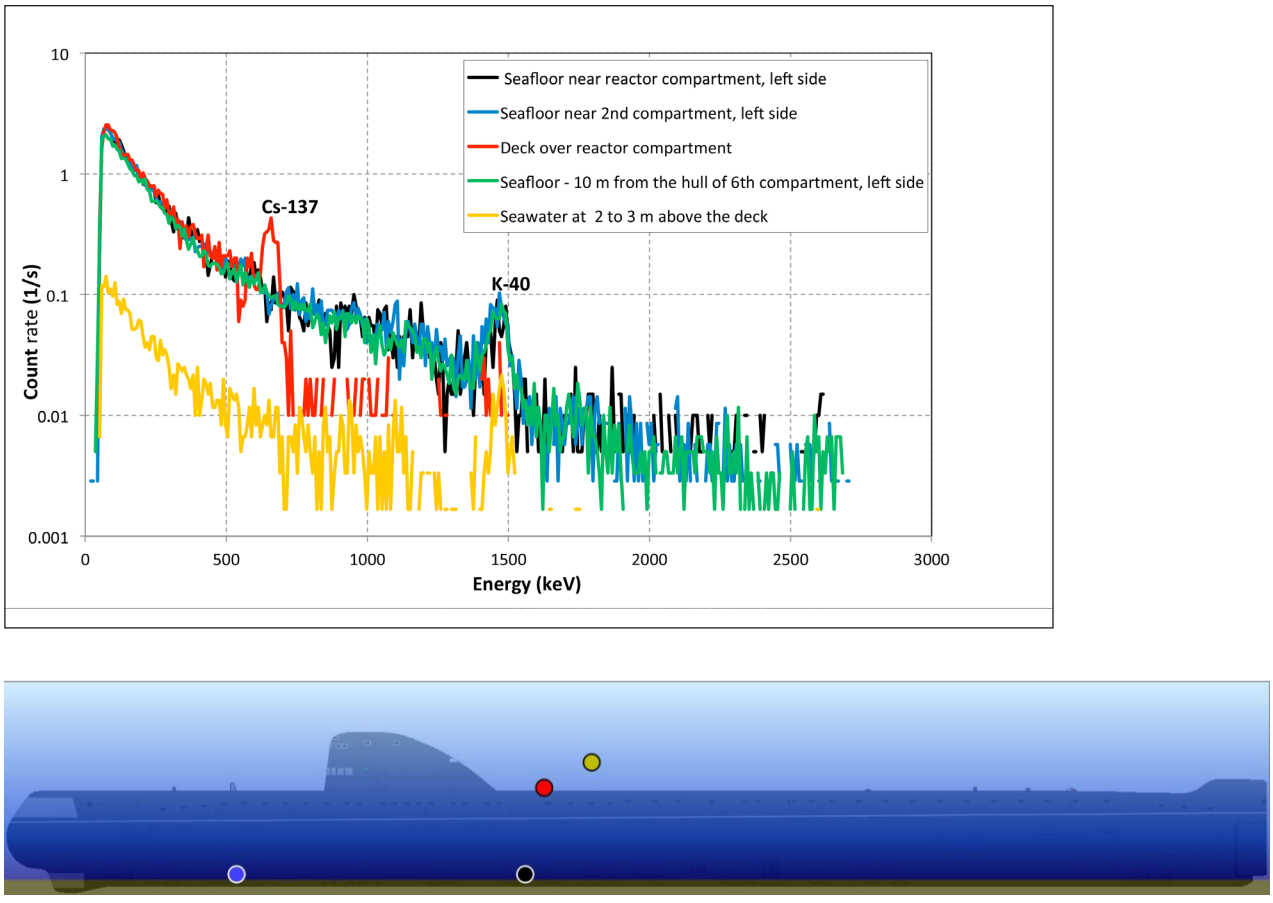


Figure 4.6. *In situ gamma spectra obtained at different locations around the nuclear submarine K-27 in the outer part of Stepovogo Fjord. Locations where spectra were taken close to K-27 are shown above.*

From the series of in situ gamma measurements that were performed around dumped containers at St. 18 in the inner part of Stepovogo Fjord, Cs-137 and Co-60 could be detected with measurements made close to 3 dumped containers, but not at a distance of 10 m from one of the containers (Figure 4.7). The maximum detected Cs-137 activity concentration in the top 5 cm of sediment close to the dumped containers was approximately 400 Bq/kg (f.w.) which would equate to a dose rate of approximately 1 µSv/h. In comparison, the maximum detected activity concentration of Co-60 in the same spectra was 10 fold lower at approximately 40 Bq/kg (f.w.). As no Cs-137 was detected in the top 5 cm of sediments at a distance of 10 m from one of the containers, the Cs-137 activity concentrations were probably less than the detection limit of 20 Bq/kg (f.w.).

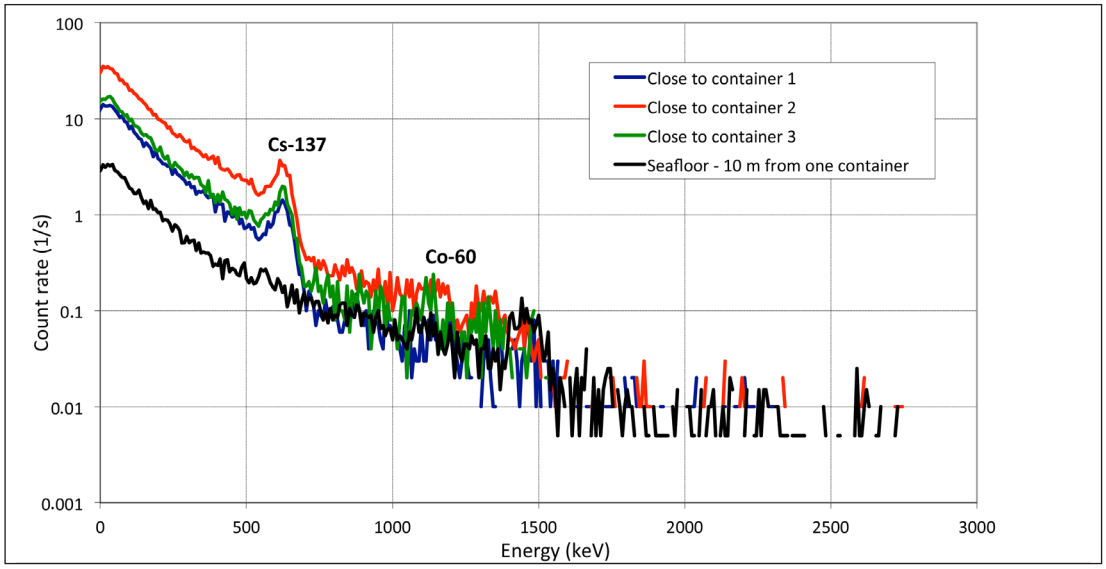


Figure 4.7. *In situ gamma spectra obtained close to dumped containers in inner part of Stepovogo Fjord.*

4.3 Radionuclides and trace elements in seawater

4.3.1 Cs-137

The mean activity concentrations of Cs-137 in filtered seawater from stations in Stepovogo Fjord were similar to those observed in the Pechora Sea (St. 1) at between 1.5 and 1.8 Bq/m^3, with the exception of bottom water from the inner part of Stepovogo Fjord. The mean activity concentration of Cs-137 in bottom water from the inner part of Stepovogo Fjord (St. 18) was 15.4 ±1.2 Bq/m^3, an order of magnitude higher than that observed in surface water at the same station (Table 4.2). A similar trend was reported for Cs-137 in seawater from the previous investigations in 1993 and 1994 in Stepovogo Fjord (JRNEG, 1996). However, observed activity concentrations of Cs-137 at the entrance (St. 45), outer part (St. 36) and inner part of Stepovogo Fjord in 2012 were all lower than those observed in 1993 and 1994 (JRNEG, 1996) at similar sampling stations (Figure 4.8). Similarly, mean Cs-137 activity concentrations in surface and bottom water from the Pechora Sea in 2012 were lower than values reported from the previous investigation in 1992 (JRNEG, 1996).

The elevated activity concentrations of Cs-137 observed in bottom water from the inner part of Stepovogo Fjord in previous investigations and in 2012 are likely the result of leakages from dumped containers and the resuspension of Cs-137 from contaminated sediments. As Cs-137 activity concentrations in bottom water from the inner part of Stepovogo Fjord remain elevated compared to those in surface water, vertical mixing between surface and bottom water masses and the flushing of bottom water with inflowing Kara Sea water would seem to be inhibited.

The mean Cs-137 activity concentrations in surface and bottom water sampled close to the nuclear submarine K-27 in the outer part of Stepovogo Fjord were similar to those observed at the entrance to Stepovogo Fjord and the Pechora Sea. This suggests that there have not been any significant leakages from the reactors of K-27. Additionally these observations give further weight to the limited exchange of bottom water from the inner part of Stepovogo Fjord with the outer part of the fjord.

The mean Cs-137 activity concentrations in bottom water from the inner part of Stepovogo Fjord in 2012 were similar to those observed in 2011 in the open Irish Sea (RPII. 2012), but lower than values reported for UK coastal waters in the Irish Sea (RIFE, 2012) and for waters in the Baltic Sea region (HELCOM, 2013). Mean Cs-137 activity concentrations for all other stations in Stepovogo Fjord in 2012 were similar to those in the Barents Sea in 2012, but lower than reported values for the Norwegian Sea and North Sea in 2011 (BfS/BMU, 2014). This trend suggests that the main source of Cs-137 in inflowing Kara Sea water to Stepovogo Fjord is derived from long range ocean transport of Cs-137 from sources further afield (Figure 4.10).

4.3.2 Sr-90

The mean activity concentrations of Sr-90 at all stations in Stepovogo Fjord were higher than those observed in the Pechora Sea. Mean activity concentrations of Sr-90 were higher in surface water than bottom water at the entrance and outer part of Stepovogo Fjord and vice versa for the inner part of Stepovogo Fjord. The mean Sr-90 activity concentration in bottom water from the inner part of Stepovogo Fjord was 5.8 ±1.5 Bq/m^3 which was similar to that observed in surface water at the entrance to Stepovogo Fjord (Table 4.2).

A similar set of observations were reported for Sr-90 in seawater from the previous investigations in 1993 and 1994 in Stepovogo Fjord (JRNEG, 1996). However, unlike the situation with Cs-137, observed activity concentrations of Sr-90 in entrance, outer part and inner part of Stepovogo Fjord in 2012 were in general comparable to those observed in 1993 and 1994 (JRNEG, 1996) at similar sampling stations (Figure 4.9). The investigation in 1993 did report an elevated Sr-90 activity concentration of 24 Bq/m^3 in bottom water from the inner part of Stepovogo Fjord, but the 1994 investigation reported a value similar to that observed in 2012 (JRNEG, 1996). In the case of the Pechora Sea, mean Sr-90 activity concentrations in surface and bottom water in 2012 were lower than values reported from the previous investigation in 1992 (JRNEG, 1996).

The possibility cannot be excluded that the Sr-90 activity concentrations observed in 2012 in bottom water from the inner part of Stepovogo Fjord continue to be influenced by leakages from dumped containers and or the remobilisation of Sr-90 from contaminated sediments.

As the mean Sr-90 activity concentrations in bottom water sampled close to the nuclear submarine K-27 in the outer part of Stepovogo Fjord were lower than those observed in surface water at the same sampling point, this would suggest that there have not been any significant leakages from the reactors of K-27.

The mean Sr-90 activity concentrations in surface and bottom water at the entrance, outer part and inner part of Stepovogo Fjord in 2012 (Figure 4.11) were lower than those observed in 2011 in UK coastal waters in the Irish Sea (RIFE, 2012) and for waters in the Baltic Sea region (HELCOM, 2013), but higher than values in the Barents Sea, Norwegian Sea and North Sea in 2011 (BfS/BMU, 2014; NRPA, 2014). This trend suggests that the main source of Sr-90 in inflowing Kara Sea water to Stepovogo Fjord is derived from sources within the Kara Sea region and most likely as a result of riverine discharges from the Ob and Yenisey rivers (e.g. Gao et al., 2009).

The different likely sources of Cs-137 and Sr-90 to water masses in Stepovogo Fjord, either from inflowing Kara Sea water or in situ sources are supported by derived Cs-137/Sr-90 activity ratios for surface and bottom water at the different sampling stations (Table 4.3). For comparison purposes, activity ratios of Cs-137 and Sr-90 in the North Sea in 2011 ranged from 1.6 to 3.4 (BfS/BMU, 2014).

Table 4.2. *Mean Cs-137 and Sr-90 activity concentrations (Bq/m^3) in filtered seawater from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Cs-137	S	1.7 ±0.3	1.6 ±0.2	1.6 ±0.3	1.5 ±0.3
(Bq/m^3)	M	1.6 ±0.2	-	-	-
	B	1.5 ±0.3	1.7 ±0.3	1.8 ±0.3	15.4 ±1.2
Sr-90	S	1.5 ±0.7	5.2 ±1.8	3.9 ±1.3	4.3 ±1.3
(Bq/m^3)	M	1.5 ±0.7	-	-	-
	B	1.3 ±0.6	3.4 ±1.6	2.5 ±1.1	5.8 ±1.5

S - surface water; M - mid-depth water; B - bottom water.
1 - Norwegian, Russian and IAEA data (St. 45 - Norwegian and Russian data only).

Table 4.3. *Cs-137/Sr-90 activity ratios in filtered seawater from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Cs-137/	S	1.1 ±0.5	0.3 ±0.1	0.4 ±0.2	0.3 ±0.1
Sr-90	M	1.1 ±0.5	-	-	-
	B	1.1 ±0.6	0.5 ±0.3	0.7 ±0.4	2.7 ±0.7

S - surface water; M - mid-depth water; B - bottom water.
1 - Norwegian, Russian and IAEA data (St. 45 - Norwegian and Russian data only).

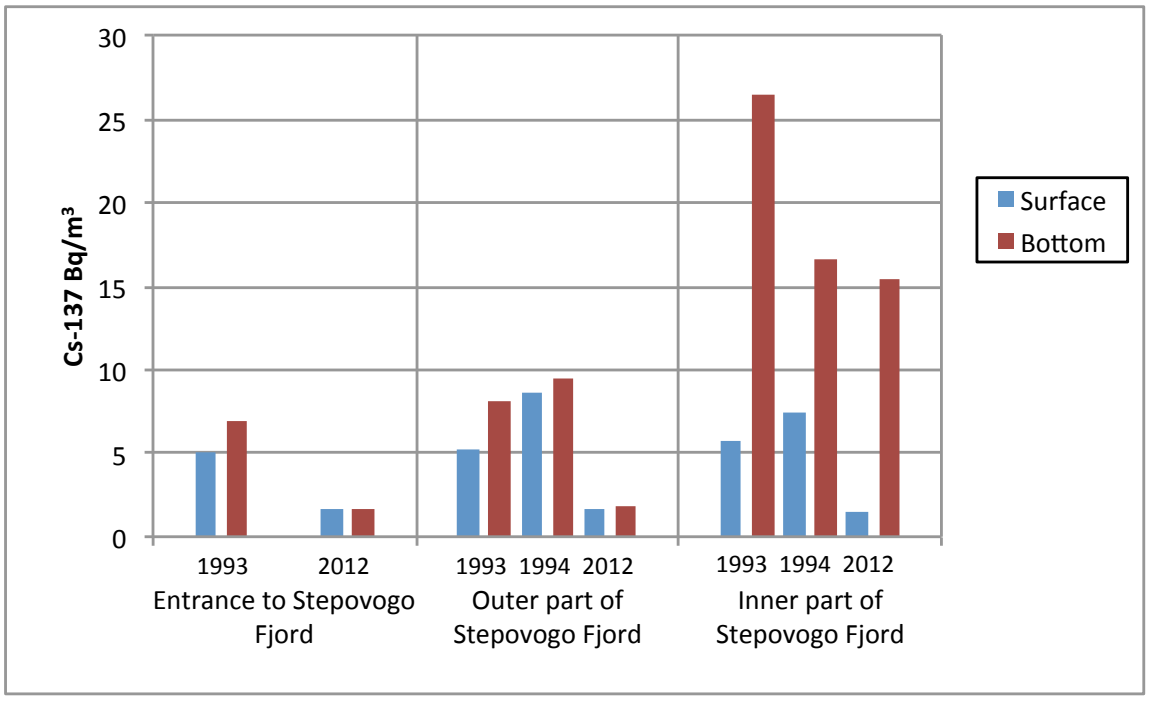


Figure 4.8. *Comparison of mean Cs-137 activity concentrations (Bq/m³) in filtered seawater (Norwegian, Russian and IAEA data) from Stepovogo Fjord in 2012 with results from previous investigations in 1993 and 1994 (JRNEG, 1996).*

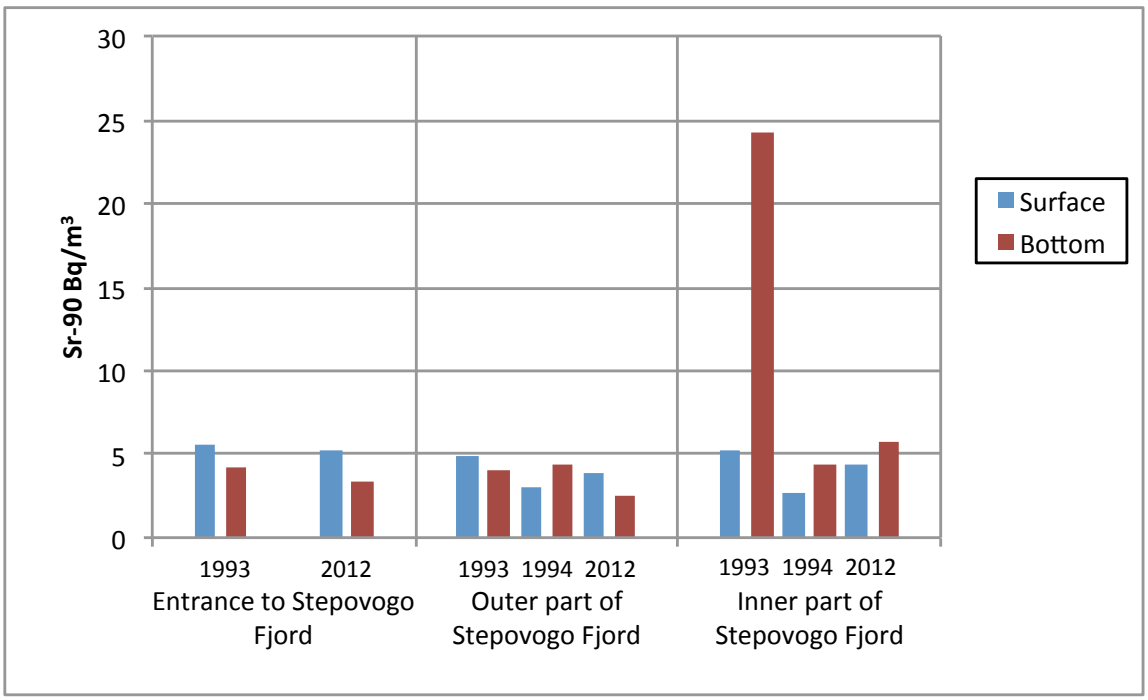


Figure 4.9. *Comparison of mean Sr-90 activity concentrations (Bq/m³) in filtered seawater (Norwegian, Russian and IAEA data) from Stepovogo Fjord in 2012 with results from previous investigations in 1993 and 1994 (JRNEG, 1996).*

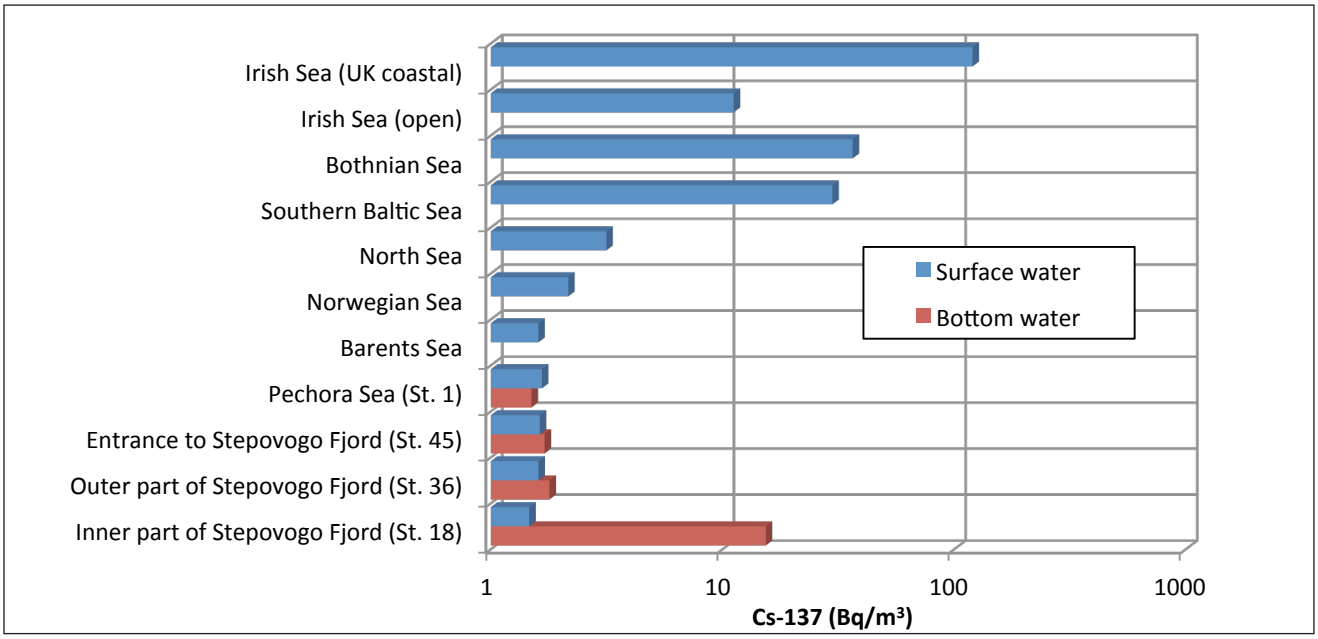


Figure 4.10. *Comparison of mean seawater Cs-137 activity concentrations (Bq/m³) from this study (Norwegian, Russian and IAEA data) with values for other seas in 2011 and 2012 (Note use of logarithmic scale for activity concentrations).*

Irish Sea, UK coastal (2011) - maximum mean activity concentration for coastal stations (RIFE, 2012).
Irish Sea, open (2011) - mean activity concentration for open sea stations (RPII, 2012)
Bothnian Sea and Southern Baltic Sea (2011) - mean activity concentration (HELCOM, 2013)
North Sea (2011) - mean activity concentration (BfS/BMU, 2014)
Norwegian Sea (2011) - mean activity concentrations (NRPA, 2014
Barents Sea (2012) - mean activity concentrations (NRPA, unpublished)

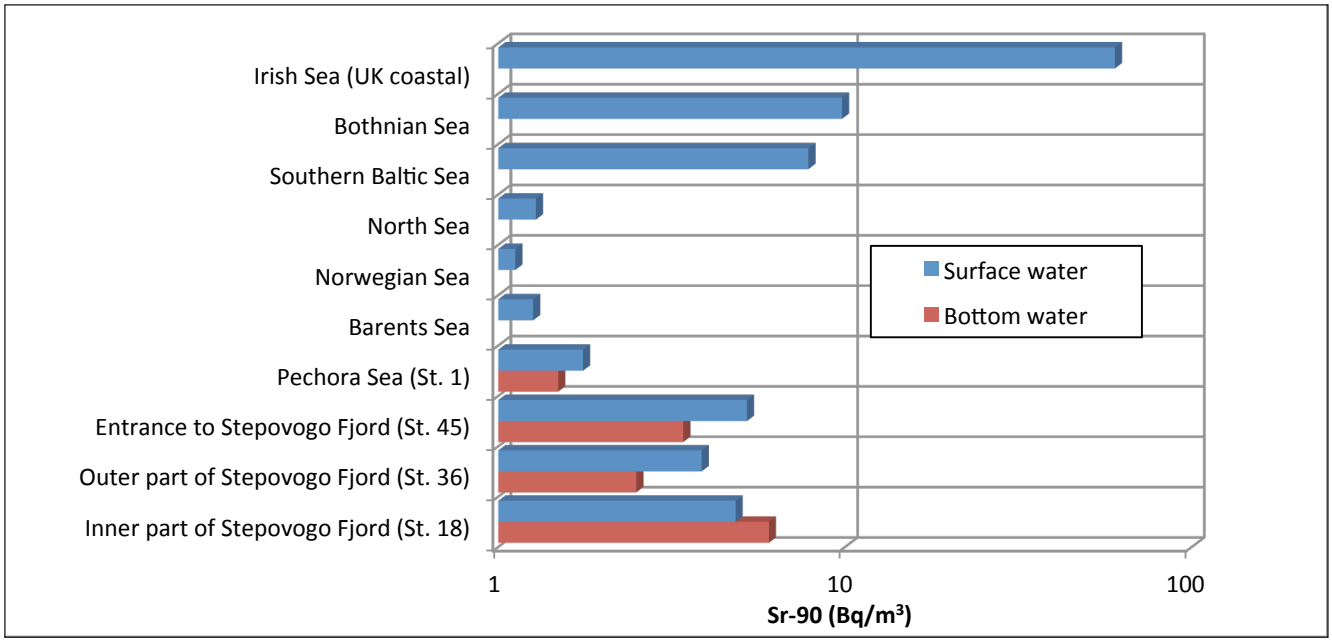


Figure 4.11. *Comparison of mean seawater Sr-90 activity concentrations (Bq/m³) from this study (Norwegian, Russian and IAEA data) with values for other seas in 2011 (Note use of logarithmic scale for activity concentrations).*

Irish Sea (UK coastal) - maximum mean activity concentration for coastal stations (RIFE, 2012).
Bothnian Sea and Southern Baltic Sea - mean activity concentration (HELCOM, 2013)
North Sea - mean activity concentration (BfS/BMU, 2014)
Norwegian Sea and Barents Sea - mean activity concentrations (NRPA, 2014)

4.3.3 Pu isotopes and Am-241

The activity concentrations of Pu-239,240 in surface water in Stepovogo Fjord were similar to that observed in surface water in the Pechora Sea at between 2.2 and 2.4 mBq/m^3. Pu-239,240 activity concentrations in bottom water from Stepovogo Fjord showed an increasing trend from the entrance (2.4 ±0.7 mBq/m^3), the outer part (3.2 ±0.7 mBq/m^3), and the inner part (4.8 ±0.8 mBq/m^3) of the fjord, while Pu-239,240 activity concentrations in the Pechora Sea increased in bottom water to 5.2 ±0.9 mBq/m^3 (Table 4.4). The activity concentrations for Pu-238 in all seawater samples were below or close to detection limits. Pu-239,240 activity concentrations in surface and bottom water from the inner and outer parts of Stepovogo Fjord in 2012 showed a slight decrease compared to reported values from previous investigations in 1993 and 1994 (JRNEG, 1996) at similar sampling stations. A distinct decrease in Pu-239,240 activity concentrations (Figure 4.12) was observed in bottom water from the entrance to Stepovogo Fjord in 2012 compared to 1993 (JRNEG, 1996). Pu-239,240 activity concentrations in bottom water from the Pechora Sea were slightly lower in 2012 than was observed in 1992 (JRNEG, 1996).

The activity concentrations of Pu-239,240 in bottom water from the entrance, outer part and inner part of Stepovogo Fjord in 2012 (Figure 4.13) were similar to those observed in surface seawater in 2011 in the Barents Sea and Norwegian Sea (NRPA, 2014), but lower than values reported for the North Sea (BfS/BMU, 2014) and for waters in the Baltic Sea region (HELCOM, 2013).

The activity concentrations of Pu-239,240 in suspended matter showed similar trends in all samples as was observed for seawater samples (Table 4.5). As a relative fraction (%) of the total activity concentration, Pu-239,240 in suspended matter accounted for 2% to 6%, with a higher fraction associated with suspended matter in bottom water than in surface water from Stepovogo Fjord. Based on size fractionation of selected samples (Table 4.6), Pu-239,240 was predominately associated with the low molecular mass (LMM) fraction (69% to 100%). However, a relative large fraction of Pu in surface water from the inner part of Stepovogo Fjord was associated with colloids (30%).

The activity concentrations of Am-241 in surface and bottom water in Stepovogo Fjord (Table 4.4) were similar at between 1.1 and 1.8 Bq/m^3 and somewhat lower than values for surface and bottom from the Pechora Sea. The observed Am-241 activity concentrations in bottom water from the entrance and outer part of Stepovogo Fjord in 2012 were lower than values reported from the previous investigations in 1993 for similar sampling stations (JRNEG, 1996). The activity concentrations of Am-241 in seawater filtered through 0.45 µm filters (Table 4.6) were similar to those in seawater samples filtered through 1 µm based on their respective uncertainties.

Am-241 activity concentrations in surface and bottom water from the entrance, outer part and inner part of Stepovogo Fjord in 2012 (Figure 4.14) were lower than those observed in 2011 in the Norwegian Sea (NRPA, 2014) and North Sea (BfS/BMU, 2014).

Am-241/Pu-239,240 activity ratios (Table 4.7) in seawater from Stepovogo Fjord and the Pechora Sea ranged between 0.31 to 1.07 and 0.40 to 1.17, respectively, but with typically large associated uncertainties that precludes any detailed interpretation. For comparison purposes, Am-241/Pu-239,240 activity ratios have been reported as being between 0.10 to 0.79 in the North Sea (BfS/BMU, 2014) and 0.17 to 0.47 in the Norwegian Sea (NRPA, 2014) in 2011. Pu-240/Pu-239 atom ratios in selected filtered (<0.45 µm) and ultrafiltered (<10 kDa) seawater samples ranged between 0.16 to 0.22 but due to the associated uncertainties involved these values likely reflect only global fallout as a source (Table 4.8).

Table 4.4. *Pu-239,240 and Am-241 activity concentrations (mBq/m^3) in filtered seawater (<1 µm) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Pu-239,240	S	2.4 ±0.8	1.4 ±0.7	2.3 ±0.6	2.2 ±0.5
(mBq/m^3)	M	4.5 ±0.9	-	-	-
	B	5.2 ±0.9	2.4 ±0.7	3.2 ±0.7	4.8 ±0.8
Am-241	S	2.8 ±0.9	1.5 ±0.6	1.3 ±0.5	1.8 ±0.5
(mBq/m^3)	M	1.8 ±0.6	-	-	-
	B	2.9 ±0.7	1.8 ±0.6	1.1 ±0.4	1.5 ±0.5

S - surface water; M - mid-depth water; B - bottom water.
1 - Norwegian data (Only Norwegian data used due to larger sampling volume. Russian data followed the same trends as Norwegian data).

Table 4.5. *Pu-239,240 activity concentrations (mBq/m³) in suspended matter (>1 μm) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Pu-239,240	S	0.10 ±0.02	0.20 ±0.07	0.17 ±0.04	0.25 ±0.06
(mBq/m³)	M	0.41 ±0.14	-	-	-
	B	0.53 ±0.18	0.43 ±0.16	0.31 ±0.10	0.38 ±0.15
% of total	S	2.1%	3.6%	3.7%	4.2%
Pu-239,240	M	6.3%	-	-	-
	B	4.3%	6.0%	5.4%	5.3%

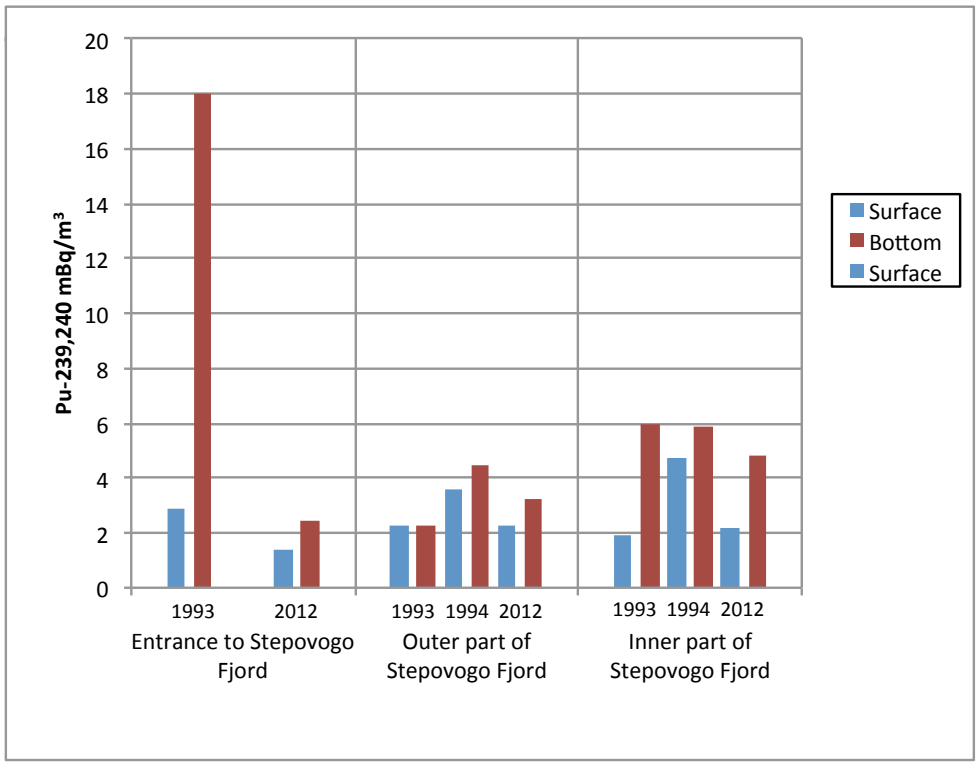


Figure 4.12. *Comparison of Pu-239,240 activity concentrations (mBq/m³) in filtered seawater (Norwegian data) from Stepovogo Fjord in 2012 with results from previous investigations in 1993 and 1994 (JRNEG, 1996).*

Table 4.6. *Pu-239,240 and Am-241 activity concentrations (mBq/m³) in filtered seawater (<0.45 μm) and ultrafiltered seawater (<10 kDa) from the Pechora Sea and Stepovogo Fjord in 2012.*

		Pechora Sea	Stepovogo Fjord	
		(St. 1)	Outer part (St. 36)	Inner part (St. 18)
Pu-239,240[1]	S	2.1 ±0.6[2]	1.8 ±0.6	2.5 ±0.9
(mBq/m³) <0.45 μm	B	3.3 ±1.2	2.2 ±0.6[2]	3.5 ±1.0
Pu-239,240[1]	S	3.2 ±0.5	1.8 ±1.8	1.7 ±0.3
(mBq/m³) <10 kDa	B	3.5 ±0.5	2.6 ±0.7	3.3 ±0.5
Am-241[2]	S	1.6 ±0.4	0.5 ±0.3	1.2 ±0.5
(mBq/m³) <0.45 μm	B	0.8 ±0.4	1.0 ±0.4	2.6 ±0.8

S - surface water; B - bottom water.
1 - Norwegian data
2 - IAEA data.

Table 4.7. *Am-241/Pu-239,240 activity ratios in filtered seawater (<1 µm) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Am-241/	S	1.17 ±0.54	1.07 ±0.69	0.57 ±0.26	0.82 ±0.29
Pu-239,240	M	0.40 ±0.16	-	-	-
	B	0.56 ±0.17	0.75 ±0.33	0.34 ±0.15	0.31 ±0.12

S - surface water; M - mid-depth water; B - bottom water.
1 - Norwegian data.

Table 4.8. *Pu-240/Pu-239 atom ratios in filtered seawater (<0.45 µm) and ultrafiltered seawater (<10 kDa) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord	
		(St. 1)	Outer part (St. 36)	Inner part (St. 18)
Pu-240/Pu-239	S	-	0.21 ±0.14	0.19 ±0.15
<0.45 µm	B	0.22 ±0.16	-	0.19 ±0.11
Pu-240/Pu-239	S	0.18 ±0.06	-	0.16 ±0.08
<10 kDa	B	0.19 ±0.06	0.17 ±0.10	0.16 ±0.05

S - surface water; B - bottom water.
1 - Norwegian data only.

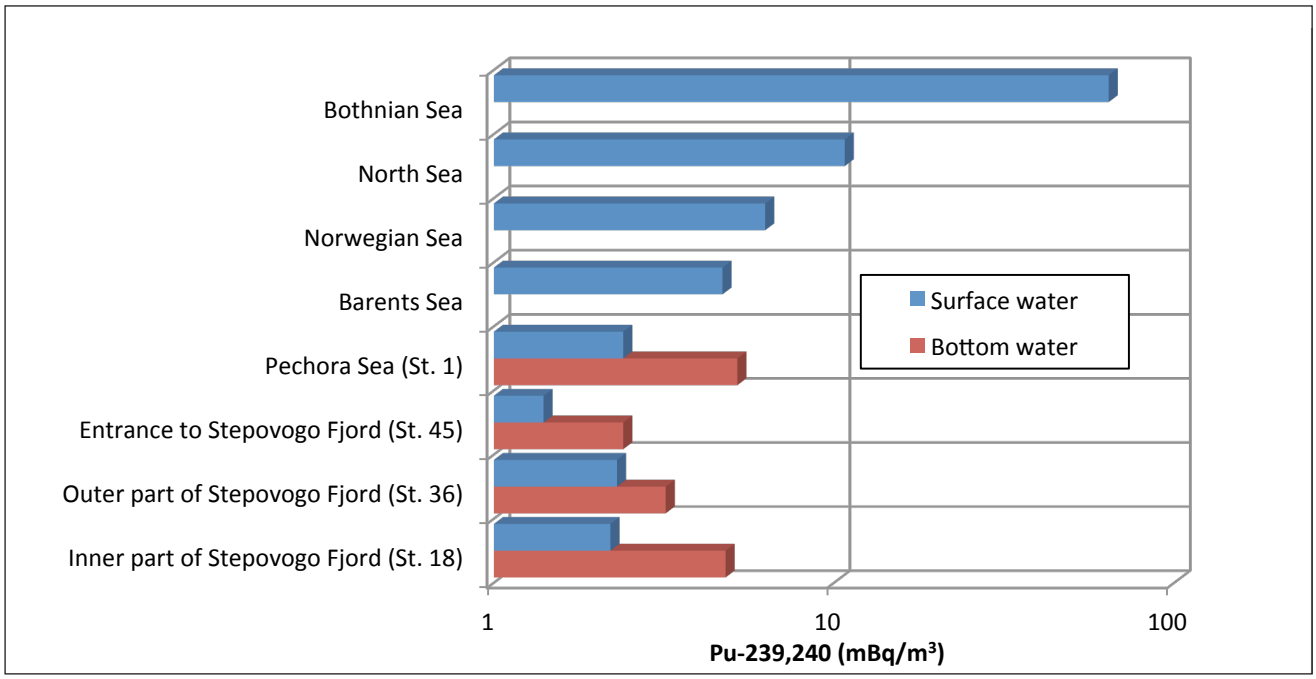


Figure 4.13. *Comparison of Pu-239,240 activity concentrations in seawater (mBq/m³) from this study (Norwegian data) with values for other seas in 2011 (Note use of logarithmic scale for activity concentrations).*

Bothnian Sea - single reported activity concentration (HELCOM, 2013)
North Sea - mean activity concentration (BfS/BMU, 2014)
Norwegian Sea - mean activity concentrations (NRPA, 2014)
Barents Sea - single reported activity concentration (NRPA, 2014)

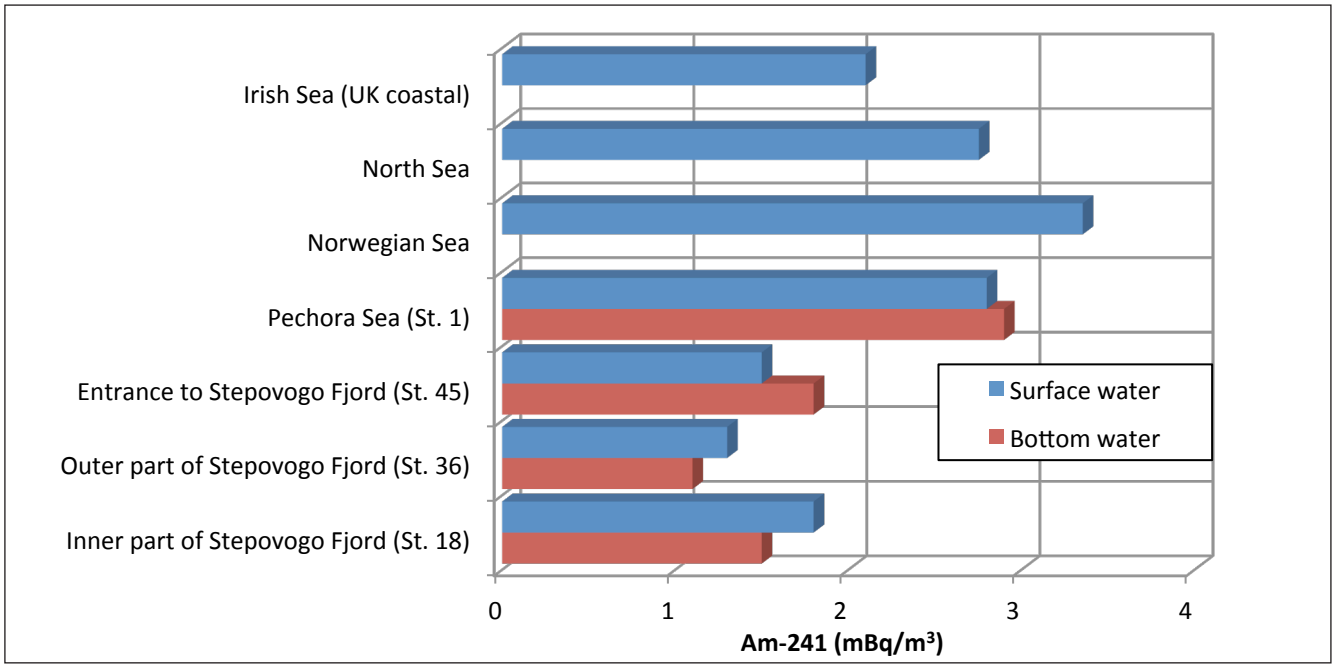


Figure 4.14. *Comparison of mean seawater Am-241 activity concentrations (mBq/m³) from this study (Norwegian data) with values for other seas in 2011.*

Irish Sea, UK Coastal - mean activity concentration (RIFE, 2012)
North Sea - mean activity concentration (BfS/BMU, 2014)
Norwegian Sea - mean activity concentrations (NRPA, 2014)

4.3.4 U isotopes

The activity concentrations of U isotopes showed little variation between the different stations sampled when taking into account their associated uncertainties (Table 4.9). Assuming a uranium seawater concentration of 3.3 µg/l (e.g. Ku et al., 1977) and typical isotopic abundance ratios, activity concentrations of U isotopes in seawater could be expected to be 40.0 Bq/m³ (U-238), 1.85 Bq/m³ (U-235) and 40.9 Bq/m³ (U-234). Activity concentrations of U isotopes in seawater samples from Stepovogo Fjord and the Pechora Sea showed some variation compared to these values. However, activity ratios of U-234/U-238 from all sampling stations (Table 4.10) were similar and in good agreement with the typical values of 1.15 reported for Atlantic water (e.g. Koide and Goldberg, 1963; Ku et al., 1977). The activity ratios of U-235/U-238 in all seawater samples (Table 4.10) were somewhat lower than the assumed natural environmental activity ratio of 0.046 (e.g. Martin and Hancock, 2004). Uranium concentrations were predominantly present as low molecular mass (LMM) species, while atom ratios of U-235/U-238 in all samples and fractions were similar at 0.007 ±0.001. The total concentration of U in surface and bottom seawater (<0.45 µm) from Stepovogo Fjord ranged between 3.2 and 3.7 µg/l (Table 4.13). U-236 measurements by AMS, which is a much more sensitive method, will be performed on seawater samples at a later date to check that no release of enriched uranium has occurred in Stepovogo Fjord.

Table 4.9. *U-234, U-235 and U-238 activity concentrations (Bq/m³) in filtered seawater (<1 µm) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
U-234	S	48.6 ±4.2	46.4 ±3.3	39.9 ±3.3	44.9 ±3.2
(Bq/m³)	B	-	46.5 ±3.3	48.8 ±3.5	47.0 ±3.3
U-235	S	1.6 ±0.3	1.6 ±0.2	1.2 ±0.3	1.4 ±0.2
(Bq/m³)	B	-	1.7 ±0.2	1.7 ±0.2	1.6 ±0.2
U-238	S	42.3 ±3.7	39.7 ±2.8	36.2 ±3.0	39.2 ±2.8
(Bq/m³)	B	-	41.4 ±3.0	43.4 ±3.1	40.9 ±2.9

S - surface water; B - bottom water
1 - Norwegian data.

Table 4.10. *Uranium activity ratios in filtered seawater (<1 µm) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
U-234/U-238	S	1.15 ±0.14	1.17 ±0.12	1.10 ±0.13	1.15 ±0.12
	B	-	1.12 ±0.11	1.12 ±0.11	1.15 ±0.11
U-235/U-238	S	0.038 ±0.008	0.040 ±0.006	0.033 ±0.009	0.036 ±0.006
	B	-	0.041 ±0.006	0.039 ±0.005	0.039 ±0.006

S - surface water; B - bottom water.
1 - Norwegian data.

4.3.5 Tc-99

The activity concentrations of Tc-99 (Table 4.11) in surface and bottom water from all sampling stations in Stepovogo Fjord were similar and comparable with surface and bottom water from the Pechora Sea. These observations are somewhat higher than the range of activity concentrations (0.05 to 0.08 Bq/m^3) reported for the western part of the Barents Sea in 2012 (NRPA, 2014), but comparable to reported values in 2009 (Gwynn et al., 2012). Following the reduction in discharges of Tc-99 from the nuclear reprocessing facility at Sellafield in 2004/2005, a slow decline in seawater activity concentrations has been observed in the Barents Sea (Gwynn et al., 2012). The higher activity concentrations of Tc-99 observed in Stepovogo Fjord in 2012 compared with the Barents Sea probably reflect the time difference for seawater with declining levels of Tc-99 from Sellafield to be transported to the Kara Sea from the Barents Sea. Additionally, contributions of Tc-99 may derive from releases from the nuclear facility at Mayak to the Techa river (Trapeznikov et al., 2000), a tributary of the Ob river that discharges into Kara Sea.

Table 4.11. *Tc-99 activity concentrations (Bq/m^3) in unfiltered seawater from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Tc-99	S	0.14 ±0.07	0.15 ±0.03	0.15 ±0.05	0.22 ±0.07
(Bq/m^3)	M	0.36 ±0.26	-	-	-
	B	0.16 ±0.10	0.21 ±0.04	0.20 ±0.06	0.27 ±0.07

S - surface water; M - mid-depth water; B - bottom water.
1 - Norwegian data.

4.3.6 H-3

The activity concentrations of H-3 in surface water from all sampling stations in Stepovogo Fjord were similar and higher than values for bottom water in Stepovogo Fjord and the Pechora Sea (Table 4.12). These values are in good agreement with previously reported activity concentrations of H-3 in the region from 2000 and 2003 (Soyfer et el., 2011). The H-3 values observed in surface water in Stepovogo Fjord probably reflects levels of H-3 in inflowing Kara Sea water. Sources of H-3 to the Kara Sea may derive from the Ob and Yenisey rivers (Yi et al., 2012) or possibly from other dumped radioactive waste (Soyfer et al., 2011). H-3 activity concentrations in surface water from Stepovogo Fjord in 2012 were comparable to 2011 values from areas in the North Sea influenced by freshwater input and discharges from Cap la Hague (BfS/BMU, 2014). H-3 activity concentrations in the Pechora Sea were comparable to open sea values in the North Sea in 2011 (BfS/BMU, 2014).

Table 4.12. *Activity concentrations of H-3 (kBq/m^3) in filtered seawater (<1 µm) from the Pechora Sea and Stepovogo Fjord in 2012[1].*

		Pechora Sea	Stepovogo Fjord		
		(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
H-3	S	0.58 ±0.10	3.50 ±0.20	3.20 ±0.20	3.30 ±0.20
(kBq/m^3)	M	1.20 ±0.20	-	-	-
	B	0.20 ±0.08	1.80 ±0.20	1.60 ±0.20	1.30 ±0.20

S - surface water; M - mid-depth water; B - bottom water.
1 - Russian data.

4.3.7 I-129

The concentrations of I-129 in seawater at all sampling stations in Stepovogo Fjord and the Pechora Sea varied between 73.0 x 10^8 and 102.8 x 10^8 atoms/l. These concentrations are significantly higher than the range (9 to 25 x 10^8 atoms/l) reported for surface waters from the Barents and Kara Seas in the 1990s (Raisbeck et al., 1993). The higher concentrations observed in 2012 can be accounted for by the long range transport of increased discharges of this radionuclide in recent years from the nuclear reprocessing facilities at Sellafield and in particular Cap la Hague (Hou et al., 2009). The concentrations of I-129 and the atom ratios of I-129/I-127 in size and charge fractionated water samples were somewhat higher in bottom water from the outer and inner parts of Stepovogo Fjord compared to the entrance to the fjord and the Pechora Sea (Figure 4.15). Both I-129 and I-127 were predominately (90% to 100%) present as low molecular mass (LMM) species. However, in bottom water from the outer part of Stepovogo Fjord, 10% of I-129 was present in colloidal form, whereas virtually all I-127 was present as LMMs. As has been observed in other studies (e.g. Hou et al., 2003, 2007), the distribution between iodate and iodide forms for I-129 (0.6 and 0.7) and I-127 (0.2 and 0.3) in bottom water from the outer and inner parts of Stepovogo Fjord were significantly different showing that the isotopes were not in chemical equilibrium.

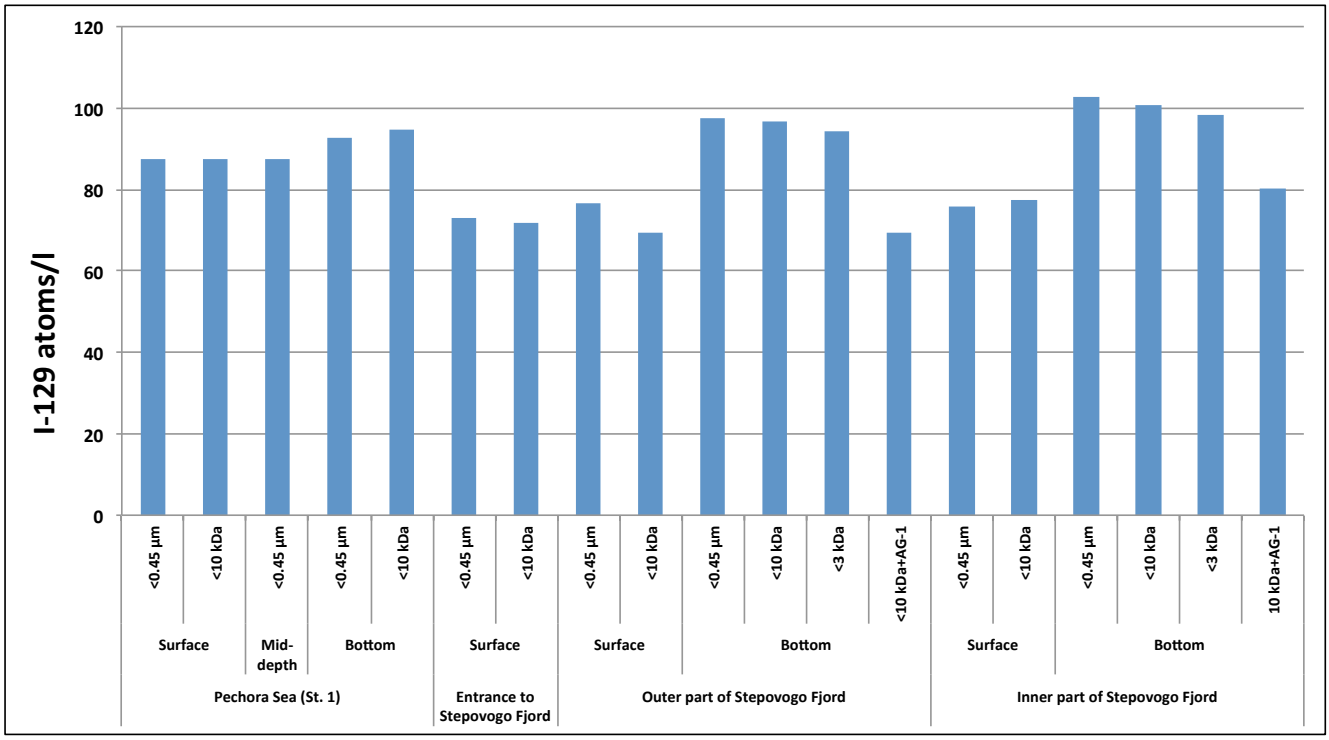


Figure 4.15. *Concentrations of I-129 (atoms/l) in size and charge fractionated seawater from the Pechora Sea and Stepovogo Fjord in 2012 (Norwegian data).*

4.3.8 Trace elements

Trace element data can provide useful site specific insights into the behaviour and occurrence of their radioactive counterparts allowing for more accurate predictions of the consequences of any hypothetical releases in the Stepovogo Fjord. Furthermore, an understanding of concentrations of potentially toxic elements is of importance in a multiple stressor context.

The relatively high concentration of Al, Fe, Mn and Co in surface waters from Stepovogo Fjord compared to the Pechora Sea and the increase in concentration from the entrance to the outer part and to the inner part of the fjord probably reflects a terrestrial signature in freshwater runoff to the fjord (Table 4.13). The concentration of Cu and Sr increased with depth for stations in Stepovogo Fjord in contrast to the trend observed for Al, Fe, Mn and Co. No difference was observed in U concentrations between surface and bottom water for any sampling station. For Al and Fe, there was an increase in the colloidal fraction in surface water from the outer part of Stepovogo Fjord compared to the inner part, where the particulate fraction was dominant indicating aggregation and removal of these trace elements from seawater within the fjord. The processes controlling the speciation of these trace elements in Stepovogo Fjord might also affect the speciation of radionuclides in the fjord.

Table 4.13. *Mean (±SD) concentration of selected trace elements in seawater from the Pechora Sea and Stepovogo Fjord in 2012[1].*

			Pechora Sea	Stepovogo Fjord		
			(St. 1)	Entrance (St. 45)	Outer part (St. 36)	Inner part (St. 18)
Al	S	Unfiltered	3.4 ±0.3	19.9 ±1.9	30.7 ±3.8	44.8 ±6.2
(µg/l)		<0.45 µm	4.9 ±3.4	9.3 ±2.4	27.2 ±8.1	10.8 ±1.2
		<10 kDa	1.0 ±0.3	9.2 ±1.5	12.2 ±4.6	8.8 ±0.5
	B	Unfiltered	11.8 ±0.8	4.6 ±1.4	11.7 ±5	8.1 ±2.3
		<0.45 µm	7.14	-	5.0 ±5.2	2.85
		<10 kDa	2.5 ±0.7	4.2 ±1.5	5.5 ±4.5	1.5 ±1.8
Mn	S	Unfiltered	1.7 ±0.1	7.7 ±0.2	7.0 ±0.2	14.8 ±0.3
(µg/l)		<0.45 µm	1.3 ±0.4	4.5 ±0.3	5.6 ±0.4	10.3 ±0.3
		<10 kDa	1.7 ±0.1	6.6 ±3.1	5.8 ±1.0	10.1 ±0.1
	B	Unfiltered	1.1 ±0.0	3.0 ±0.4	5.0 ±0.6	11.8 ±0.1
		<0.45 µm	0.8	-	4.2 ±0.3	1.68
		<10 kDa	1.4 ±0.1	2.4 ±0.3	4.3 ±0.5	2.2 ±0.1
Fe	S	Unfiltered	3.9 ±0.7	23.8 ±1.3	40.1 ±1.4	54.1 ±3.5
(µg/l)		<0.45 µm	4.3 ±0.7	1.5 ±0.5	31.8 ±13.3	2.1 ±0.5
		<10 kDa	4.5 ±3.0	1.9 ±0.6	6.9 ±2.0	2.2 ±0.2
	B	Unfiltered	17.3 ±0.4	31.3 ±1.4	21.6 ±8.8	43.9 ±3.0
		<0.45 µm	6.0	-	8.0 ±4.3	29.44
		<10 kDa	3.6 ±0.3	3.1 ±0.2	3.8 ±0.9	13.8 ±1.4
Co	S	Unfiltered	0.04 ±0.01	0.20 ±0.02	0.18 ±0.00	0.34 ±0.01
(µg/l)		<0.45 µm	0.03 ±0.01	0.19 ±0.01	0.17 ±0.01	0.29 ±0.03
		<10 kDa	0.05 ±0.01	0.23 ±0.04	0.12 ±0.03	0.27 ±0.05
	B	Unfiltered	0.04 ±0.00	0.08 ±0.02	0.05 ±0.02	0.07 ±0.02
		<0.45 µm	-	-	0.06 ±0.01	0.03
		<10 kDa	0.05 ±0.02	0.05 ±0.00	0.10 ±0.06	0.03 ±0.00
Cu	S	Unfiltered	0.3 ±0.0	0.4 ±0.1	0.6 ±0.0	0.5 ±0.0
(µg/l)		<0.45 µm	0.5 ±0.0	0.4 ±0.1	-	0.5 ±0.1
		<10 kDa	0.3 ±0.0	0.6 ±0.2	-	0.4 ±0.1
	B	Unfiltered	0.3 ±0.0	2.7 ±0.1	1.2 ±0.1	1.9 ±0.1
		<0.45 µm	0.5	-	1.2 ±0.4	1.45
		<10 kDa	0.6 ±0.0	1.1 ±0.0	0.9 ±0.2	1.0 ±0.1
Sr	S	Unfiltered	7.8 ±0.3	7.3 ±0.5	7.1 ±0.2	7.2 ±0.4
(mg/l)		<0.45 µm	7.8 ±0.1	7.0 ±0.4	7.0 ±0.2	7.4 ±0.1
		<10 kDa	7.5 ±0.3	7.2 ±0.1	7.2 ±0.8	7.2 ±0.5
	B	Unfiltered	7.8 ±0.1	7.9 ±0.4	7.8 ±0.1	8.0 ±0.1
		<0.45 µm	8.0	-	7.8 ±0.3	8.02
		<10 kDa	7.8 ±0.2	7.6 ±0.1	7.7 ±0.9	8.1 ±0.1
U	S	<0.45 µm	3.5 ±0.1	3.3 ±0.1	3.2 ±0.1	3.3 ±0.0
(µg/l)		<10 kDa	3.5 ±0.0	3.3 ±0.0	3.3 ±0.2	3.3 ±0.0
	B	<0.45 µm	3.5 ±0.0	3.5 ±0.0	3.5 ±0.0	3.66
		<10 kDa	3.54	-	3.4 ±0.2	3.5 ±0.1

S - surface water; B - bottom water

1 - Norwegian data - n=3 in each case except where no SD is stated (n=1); where SD is given as 0.0 and 0.00, SD was less than 0.05 and 0.005, respectively. Uncertainties on individual measurements were typically between 2% and 5%

4.4 Radionuclides and trace elements in sediments

4.4.1 Cs-137

The overall mean activity concentrations of Cs-137 in surface sediments collected with sediment grab or box corer showed an increasing trend from the entrance (4.6 ±3.3 Bq/kg d.w.), to the outer part (9.1 ±2.0 Bq/kg d.w.) and inner part (35.6 ±14.3 Bq/kg d.w.) of Stepovogo Fjord (Table 4.14). In comparison, the mean activity concentration of Cs-137 in surface sediments from the Pechora Sea in 2012 was 5.1 ±1.3 Bq/kg (d.w.), lower than those observed in the previous investigation in 1992 (JRNEG, 1996).

The range of mean activity concentrations of Cs-137 in surface sediments in the outer part of Stepovogo Fjord at a distance from the nuclear submarine K-27 in 2012 was similar to that observed in previous investigations in 1993, 1994 and 2004 (JRNEG, 1996; Dahle et al., 2009). The mean activity concentration of Cs-137 in a single surface sediment sample collected from the deck of the nuclear submarine K-27 with the ROV was higher than other surface sediments collected in the outer part of Stepovogo Fjord at 24.3 ±6.4 Bq/kg (d.w.). This value, however, is lower than observations of elevated activity concentrations of Cs-137 in surface sediments around K-27 from previous investigations in 1993 of 203 Bq/kg (d.w.) and 1670 Bq/kg (d.w.) in 1994 (JRNEG, 1996).

The range of mean activity concentrations of Cs-137 in surface sediments in the inner part of Stepovogo Fjord in 2012 showed less variation than in previous investigations in 1993, 1994 and 2004 (JRNEG, 1996; Dahle et al., 2009). The maximum mean activity concentration of Cs-137 in surface sediments collected with sediment grab or box corer in the inner part of Stepovogo Fjord in 2012 was up to 2 orders of magnitude lower than similar samples taken in 1993, 1994 and 2004 (JRNEG, 1996; Dahle et al., 2009). The mean activity concentration of Cs-137 in a single surface sediment sample collected close to a dumped container with the ROV was higher than other surface sediments collected in the inner part of Stepovogo Fjord at 67.2 ±7.4 Bq/kg (d.w.). This value, however, is lower than observations of elevated activity concentrations of Cs-137 in surface sediments close to dumped containers from previous investigations in 1994 of up to 109000 Bq/kg (d.w.) (JRNEG, 1996).

The mean Cs-137 activity concentrations in surface sediments from the inner part of Stepovogo Fjord in 2012 (Figure 4.16) were similar to those observed in 2011 in the open Irish Sea (RPII, 2012), but lower than values reported for UK coastal waters in the Irish Sea (RIFE, 2012) and for waters in the Baltic Sea region (HELCOM, 2013). Mean Cs-137 activity concentrations in surface sediments from the outer part of Stepovogo Fjord in 2012 were higher than those reported for the Norwegian Sea in 2011 and the Barents Sea in 2012 (NRPA, 2014).

The activity concentrations of Cs-137 in sediment cores from the outer part of Stepovogo Fjord showed relatively little variation in the top 10 cm, with peak activity concentrations typically occurring in the top 5 cm (Figure 4.17). A similar trend was observed in sediment cores from the outer part of Stepovogo Fjord in the previous investigation in 1993 (JRNEG, 1996). In contrast, activity concentrations of Cs-137 in sediment cores from the inner part of Stepovogo Fjord (Figure 4.18) typically showed elevated activity concentrations in the top 8 cm, decreasing sharply at deeper sediment depths to below 1 Bq/kg (d.w.). Profiles of Cs-137 activity concentrations in sediment cores from the entrance to Stepovogo Fjord and the Pechora Sea were similar, but with peak Cs-137 activity concentrations occurring at 4 to 6 cm depth (Figure 4.19). Peak Cs-137 activity concentrations in cores from the inner part of Stepovogo Fjord occurred in general at 3 to 4 cm, whereas peak Cs-137 activity concentrations in cores from the previous investigation in 1993 and 1994 (JRNEG, 1996) were typically observed in the top 2 cm (Figure 4.20).

Inventories of Cs-137 in sediment cores from Stepovogo Fjord in 2012 showed the same pattern as for surface sediments with an increasing trend from the entrance (386 ±125 Bq/m^2 d.w.), to the outer part (850 ±276 Bq/m^2 d.w.) and inner part (3302 ±560 Bq/m^2 d.w.) of Stepovogo Fjord (Table 4.15). In comparison, the inventory of Cs-137 in a single sediment core from the Pechora Sea was 581 ±25 Bq/m^2 (d.w.).

The observed profiles and inventories of Cs-137 in sediments in the inner and outer parts of Stepovogo Fjord are likely due to the different sedimentation rates and fluxes of Cs-137. The bathymetry of the inner part of Stepovogo Fjord and the presence of the underwater sill between the inner and outer parts of the fjord are likely to result in higher sedimentation rates in the inner part of Stepovogo Fjord compared to the outer part. The profiles of Cs-137 in sediments from the inner part of Stepovogo Fjord suggest that previous peak activity concentrations of Cs-137 in surface layers that resulted from leakages from dumped waste are slowly being buried by further input of sediment. However, that surface activity concentrations of

Cs-137 in the inner part of Stepovogo Fjord are still elevated compared to the outer part may reflect a combination of the high adsorption capacity of the sediments, the remobilisation of Cs-137 from earlier contaminated sediments, sediment mixing through bioturbation and water currents or more recent leakages from dumped containers. Although the maximum activity concentration of Cs-137 observed in any sediment sample from the inner part of Stepovogo Fjord in 2012 was 67.2 Bq/kg (d.w.), this does not preclude the possibility of sediment concentrations (either at the surface or sub-surface) with higher levels of contamination as was observed in sediment samples from the previous investigations in 1993 and 1994 (JRNEG, 1996). Sediment profiles and inventories of Cs-137 in the outer part of Stepovogo Fjord suggest a lower, but steady, flux of Cs-137 that is influenced by the slow exchange of contaminated bottom water from the inner part of Stepovogo Fjord as well as inflowing Kara Sea water.

4.4.2 Sr-90

The mean activity concentrations of Sr-90 in Stepovogo Fjord were all below 1 Bq/kg (d.w.) and as was the case for Cs-137, showed an increasing trend from the entrance (0.12 ±0.05 Bq/kg d.w.), to the outer part (0.43 ±0.28 Bq/kg d.w.) and inner part (0.72 ±0.36 Bq/kg d.w.) of Stepovogo Fjord (Table 4.14). All activity concentrations in surface sediments from Stepovogo Fjord were lower than was observed in the Pechora Sea and typically lower than observations of Sr-90 in surface sediments from previous investigations in 1993 and 1994 (JRNEG, 1996). The mean activity concentration of Sr-90 in a single surface sediment sample collected from the deck of the nuclear submarine K-27 with the ROV was higher than other surface sediments collected in the outer part of Stepovogo Fjord at 24.1 ±2.4 Bq/kg (d.w.). This value is higher than the reported range of Sr-90 activity concentrations (0.4 to 6 Bq/kg d.w.) in surface sediments around K-27 from the previous investigations in 1994 (JRNEG, 1996). It is worth noting that the sediment sample collected with the ROV from the deck of the nuclear submarine K-27 had a Cs-137/Sr-90 activity ratio up to two orders of magnitude lower than any other surface sediment sample collected in Stepovogo Fjord.

The activity concentration of Sr-90 in a single surface sediment sample collected close to a dumped container with the ROV was higher than other surface sediments collected in the outer part of Stepovogo Fjord at 3.5 ±0.7 Bq/kg (d.w.). This value however is lower than observations of elevated activity concentrations of Sr-90 in surface sediments close to dumped containers from previous investigations in 1994 of up to 310 Bq/kg (d.w.) (JRNEG, 1996).

The activity concentrations of Sr-90 in 2 sediment cores taken close to the nuclear submarine K-27 in the outer part of Stepovogo Fjord showed profiles of decreasing activity concentrations with sediment depth (Figure 4.21). Data available for a single core from the inner part of Stepovogo Fjord would suggest a sub-surface peak for Sr-90 activity concentrations at 7 to 8 cm, but not all slices from this core have been analysed. This compares to a peak activity concentration of Sr-90 at 3 to 4 cm in a single core from the inner part of Stepovogo Fjord taken in the previous investigation in 1994 (JRNEG, 1996). Inventories of Sr-90 for the two cores taken close to the nuclear submarine K-27 in the outer part of Stepovogo Fjord were 44 ±6 and 80 ±10 Bq/m^2 to a depth of 6 cm.

As was the case for Cs-137, the observed profiles of S-90 in sediments in the inner and outer parts of Stepovogo Fjord are likely due to the different sedimentation rates and fluxes of Sr-90. In the outer part of Stepovogo Fjord, the flux of Sr-90 to sediments is likely to be dominated by the continual supply of inflowing Kara Sea water against a background of low sedimentation rates. In the inner part of Stepovogo Fjord, Sr-90 arising from leakages from dumped containers is likely to be the dominant source against a background of higher sedimentation rates.

Although the maximum activity concentration of Sr-90 observed in any sediment sample from the inner part of Stepovogo Fjord in 2012 was 3.5 Bq/kg (d.w.), this does not preclude the possibility of sediment concentrations (either at the surface or sub-surface) with higher levels of contamination as was observed in sediment samples from the previous investigations in 1993 and 1994 (JRNEG, 1996). For comparison purposes, activity concentrations of Sr-90 observed in sediments from Stepovogo Fjord in 2012 were lower than maximum mean values reported for UK coastal sediments in 2011 from the Irish Sea (RIFE, 2012).

4.4.3 Co-60

Activity concentrations of Co-60 were observed (i.e. above detection limits) in only a few sediment samples from the inner part of Stepovogo Fjord, including in the ROV sediment sample collected close to a dumped container. All activity concentrations of Co-60 in 2012 were below 0.5 Bq/kg (d.w.), far lower than reported values of up to 3150 Bq/kg (d.w.) for

sediments from the inner part of Stepovogo Fjord from the previous investigation in 1994 (JRNEG, 1996). No other gamma emitting anthropogenic radionuclides were detected in any sediment sample.

Table 4.14. *Cs-137 and Sr-90 activity concentrations (Bq/kg d.w.) in surface sediments from the Pechora Sea and Stepovogo Fjord in 2012.*

Sediment	Cs-137[1] (Bq/kg d.w.)			Sr-90[2] (Bq/kg d.w.)		
	n	Mean (±SD)	Range	n	Mean (±SD)	Range
Pechora Sea (St. 1)	1	5.1 ±1.3[3]	-	1	1.9 ±0.6[4]	-
Entrance to Stepovogo Fjord	5	4.6 ±3.3	2.1 - 10.2	5	0.12 ±0.05	0.05 - 0.17
Outer part of Stepovogo Fjord	10	9.1 ±2.0	4.5 - 11.3	8	0.43 ±0.28	0.28 - 0.87
ROV sample from deck of K-27	1	24.3 ±6.4[3]	-	1	24.1 ±2.4[4]	-
Inner part of Stepovogo Fjord	13	35.7 ±14.3	17.2 - 58.9	12	0.72 ±0.36	0.36 - 1.60
ROV sample close to dumped container	1	67.2 ±7.4[3]	-	1	3.5 ±0.7[4]	-

Uncertainties on all individual Cs-137 and Sr-90 measurements were typically less than 10% and 30% respectively. n - number of sampling stations. Ranges for Cs-137 are the range of mean values for each sampling station, while ranges for Sr-90 are the range of individual measurements for each sampling station.

1 - Norwegian, Russian and IAEA data.
2 - Russian data.
3 - Mean (± SD) of measurements of two samples from the same station (Norwegian and Russian data).
4 - Based on a single sample and stated with associated uncertainty.

Table 4.15. *Inventories for Cs-137 (Bq/m² d.w.) in sediment cores from the Pechora Sea and Stepovogo Fjord in 2012[1].*

	n	Core depth (cm)	Cs-137 (Bq/m² d.w.)	
			Mean (±SD)	Range
Pechora Sea (St. 1)	1	12	581 ±25[2]	-
Entrance to Stepovogo Fjord	3	6-8	386 ±125	247 - 487
Outer part of Stepovogo Fjord	8	6-10	850 ±276	307 - 1148
Inner part of Stepovogo Fjord	11	8-16	3302 ±560	2556 - 4407

n - number of sampling stations.
1 - Norwegian data.
2 - Stated with propagated uncertainty

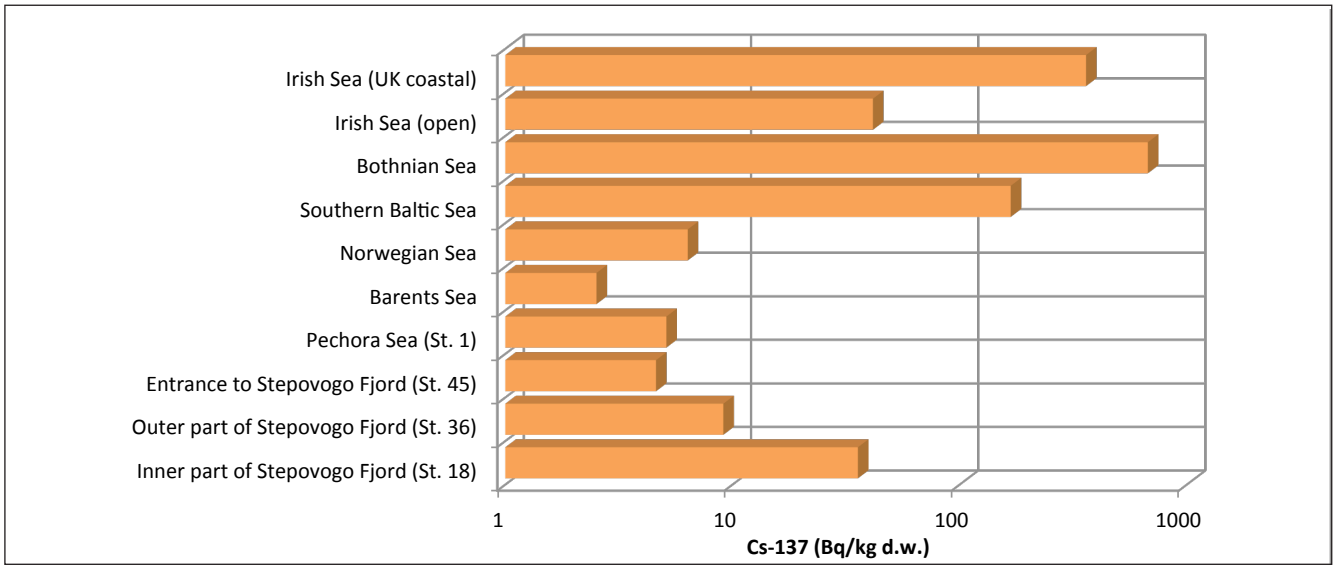


Figure 4.16. *Comparison of surface sediment Cs-137 activity concentrations (Bq/kg d.w.) from this study (Norwegian, Russian and IAEA data) with values for other seas in 2011 and 2012 (Note use of logarithmic scale for activity concentrations). Note, data from samples taken close to dumped objects with ROV in 2012 are not included.*

Irish Sea, UK coastal (2011) - maximum activity concentration for coastal stations (RIFE, 2012).
Irish Sea, open (2011) - mean activity concentration for open sea stations (RPII, 2012)
Bothnian Sea and Southern Baltic Sea (2011) - mean activity concentration (HELCOM, 2013)
Barents Sea (2012) - mean activity concentrations (NRPA, 2014)
Norwegian Sea (2011) - mean activity concentrations (NRPA, 2014)

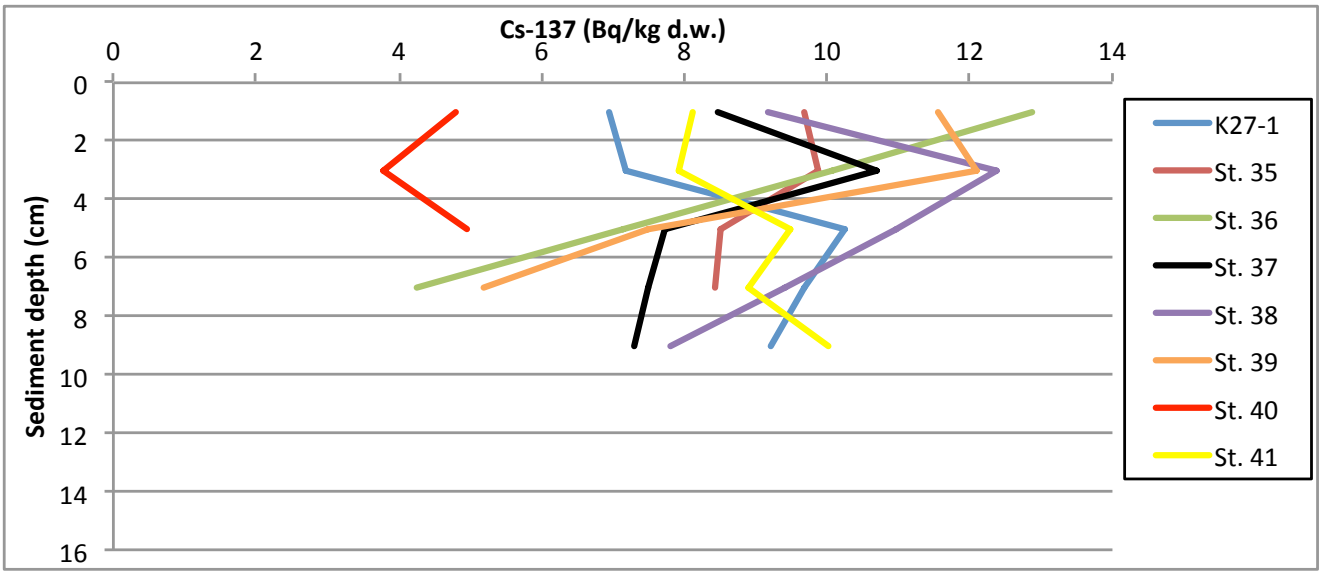


Figure 4.17. *Cs-137 activity concentrations (Bq/kg d.w.) in sediment cores (2 cm slices) from around the nuclear submarine K-27 in the outer part of Stepovogo Fjord in 2012 (Norwegian data only). Uncertainties on individual measurements were typically less than 10%.*

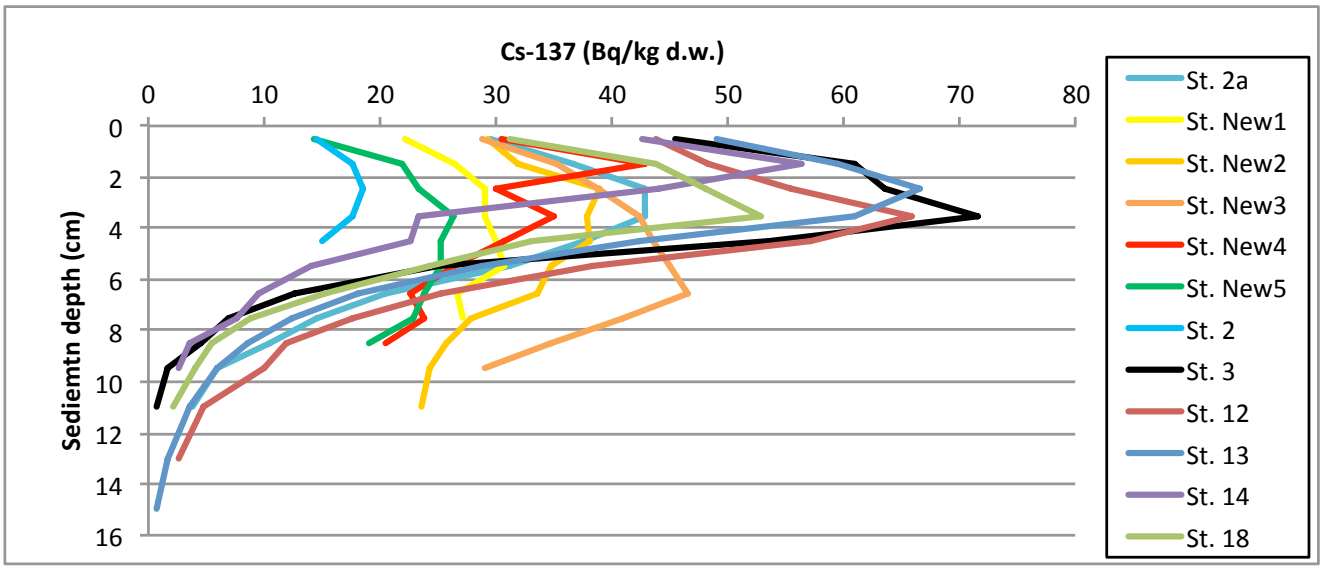


Figure 4.18*. Cs-137 activity concentrations (Bq/kg d.w.) in sediment cores (1 cm slices) from the inner part of Stepovogo Fjord in 2012 (Norwegian data only). Uncertainties on individual measurements were typically less than 10%.*

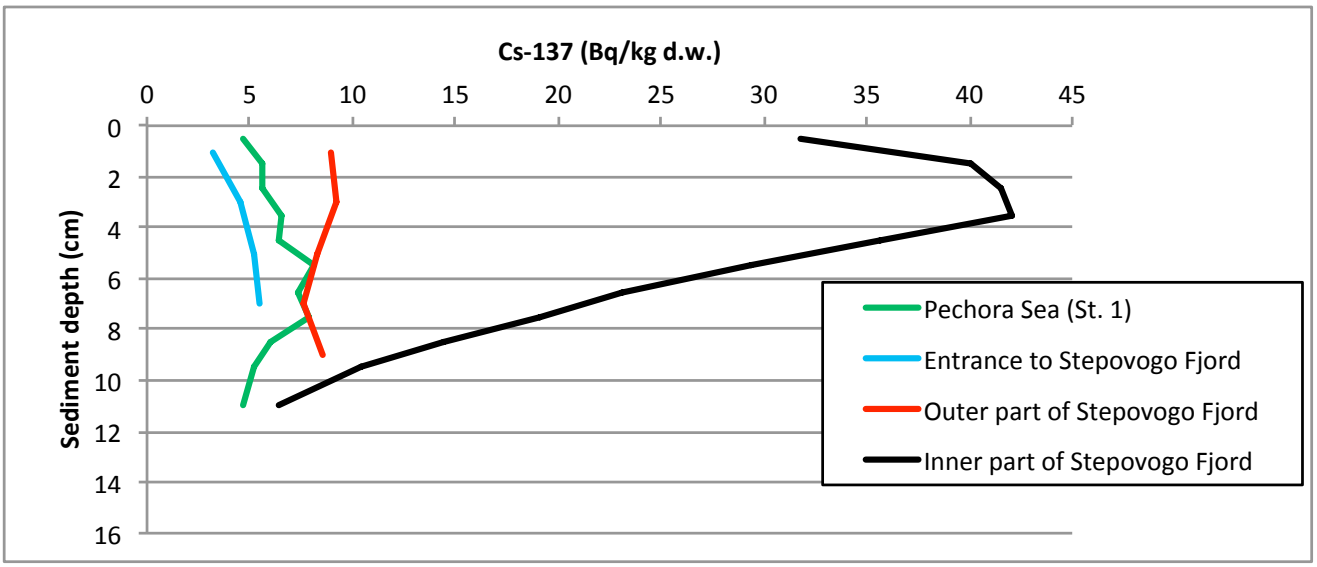


Figure 4.19. *Comparison of mean Cs-137 activity concentrations (Bq/kg d.w.) in sediment cores from the Pechora Sea (Norwegian and Russian data) and Stepovogo Fjord (Norwegian data only). Uncertainties on individual measurements were typically less than 10%.*

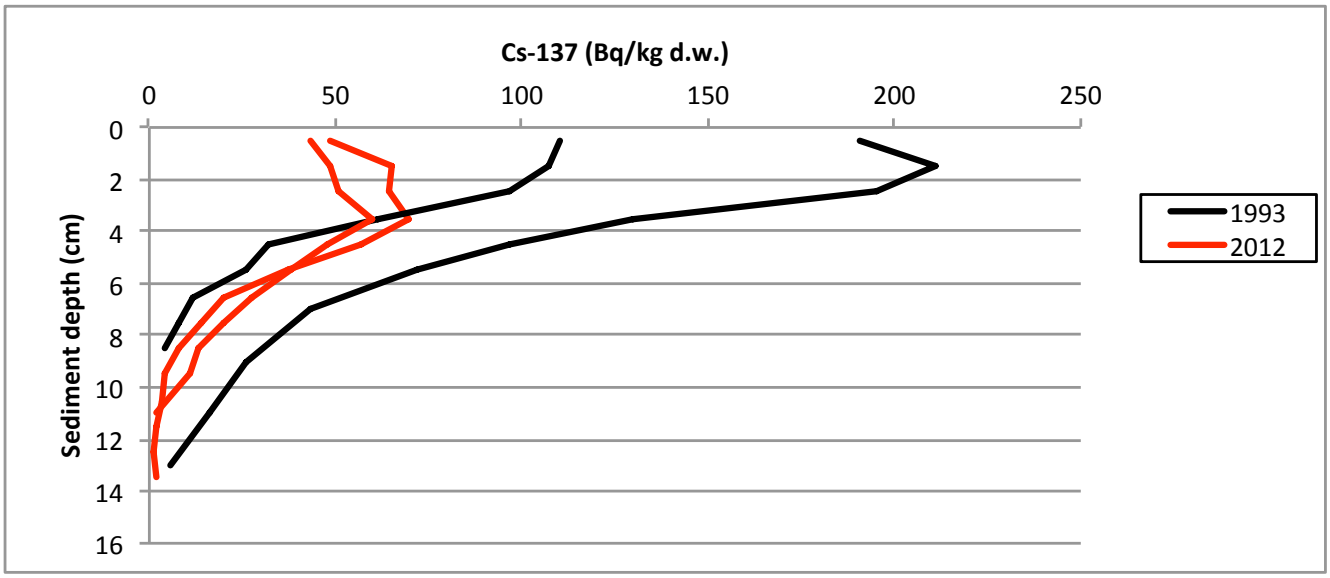


Figure 4.20. *Comparison of Cs-137 activity concentration (Bq/kg d.w.) profiles in sediment cores taken in the inner part of Stepovogo Fjord in 1993 (JRNEG, 1996) and 2012 (Norwegian, Russian and IAEA data). Note peak activity concentrations in 2012 were between 3 to 4 cm compared with the top 2 cm in 1993. Uncertainties on individual measurements were typically less than 10%.*

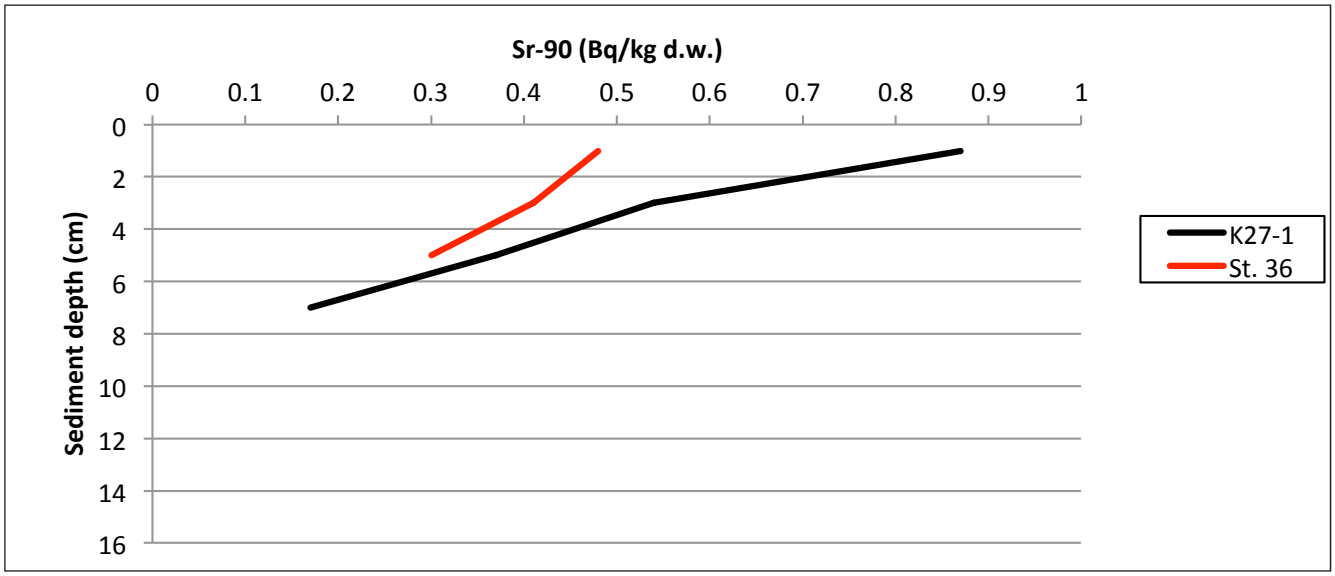


Figure 4.21. Sr-90 activity concentrations (Bq/kg d.w.) in two sediment cores close to the nuclear submarine K-27 in the outer part Stepovogo Fjord (Russian data). Uncertainties on individual measurements were typically less than 10%.

4.4.4 Pu isotopes and Am-241

In general, activity concentrations of Pu-238, Pu-239,240 and Am-241 in surface sediments collected with sediment grab or box corer were similar in the outer and inner parts of Stepovogo Fjord with means based on typical values somewhat higher than for the entrance of Stepovogo Fjord (Table 4.16). However, single sediment samples from stations at the entrance (St. 44), the inner part (St. 39) and outer part (St. New4) of Stepovogo Fjord showed higher activity concentrations than typical values (Table 4.16). Surface sediments from the Pechora Sea in 2012 showed higher activity concentrations of Pu-238 and Pu-239,240 than typical values for Stepovogo Fjord, but similar values for Am-241. The activity concentrations of Pu-238, Pu-239,240 and Am-241 in surface sediments collected in Stepovogo Fjord and in the Pechora Sea in 2012 fell within the range of values observed for these radionuclides in the previous investigations in the 1990s (JRNEG, 1996) and 2004 (Dahle et al., 2009).

The mean activity concentration of Pu-239,240 in a single surface sediment collected from the deck of the nuclear submarine K-27 with the ROV was 1.2 ±1.3 Bq/kg (d.w.) which was within the range of values (<0.1 to 6 Bq/kg d.w.) reported for surface sediments around K-27 from previous investigations in the 1990s (JRNEG, 1996). The activity concentrations of Pu-238, Pu-239,240 and Am-241 in the sediment sample collected close to a dumped container with the ROV were comparable to typical values for these radionuclides in surface sediments from other stations in the inner part of Stepovogo Fjord in 2012 (Table 4.16).

The activity concentrations of Pu-238, Pu-239,240 and Am-241 in surface sediments from Stepovogo Fjord and the Pechora Sea in 2012 were far lower than maximum mean values reported for sediments from UK coastal waters in the Irish Sea in 2011 (RIFE, 2012) of 110, 600 and 1300 Bq/kg (d.w.) respectively.

The profiles of Pu-238, Pu-239,240 and Am-241 activity concentrations in sediment cores showed similar trends within each core but some differences between sampling stations (Figure 4.22). In the Pechora Sea, the activity concentrations of Pu-238, Pu-239,240 and Am-241 showed some evidence of sub-surface maxima within the top 10 cm of sediment, whereas the activity concentrations of Pu-239,240 and Am-241 in the top 10 cm of a core (K27-1) taken close to the nuclear submarine K-27 in the outer part of Stepovogo Fjord were similar. Of 3 cores taken in the inner part of Stepovogo Fjord, one (St. New4) showed similar activity concentrations of Pu-238, Pu-239,240 and Am-241 down core, while cores from St. 12 and St. 18 showed sub-surface maxima at 3 to 4 cm with marked decreases in activity concentrations of Pu-238, Pu-239,240 and Am-241 at lower sediment depths.

Inventories of Pu-238, Pu-239,240 and Am-241 from sediment cores in Stepovogo Fjord were in general lower than those for a single sediment core from the Pechora Sea (Table 4.17). However, the inventory of Pu-238 (9.9 ±0.9 Bq/m^2 d.w.) from St. New4 in the inner part of Stepovogo Fjord was more than 2 fold higher than that observed at St. 1 and 9 fold higher than from two other cores from the inner part of Stepovogo Fjord. Additionally, the inventories of Pu-239,240 (77 ±4 Bq/m^2 d.w.) and Am-241 (50 ±3 Bq/m^2 d.w.) at St. New4 were higher than for all other sediment cores from either the inner or outer parts of Stepovogo Fjord.

In general, activity ratios of Pu-238/Pu-239,240 in all surface sediments from Stepovogo Fjord were similar with mean activity ratios based on typical activity concentrations of 0.039 ±0.021 for the entrance, 0.044 ±0.016 for the outer part and 0.045 ±0.011 for the inner part (Table 4.18). These values were similar to a Pu-238/Pu-239,240 activity ratio (0.046 ±0.031) derived for a surface sediment sample from the Pechora Sea. However, the activity ratio of Pu-238/Pu-239,240 in a surface sediment sample from the entrance (St. 44) to Stepovogo Fjord was 0.95 ±0.09, whereas surface sediment samples from two stations in the inner part of Stepovogo Fjord showed activity ratios of 0.18 ±0.06 (St. New4) and 0.10 ±0.02 (St. 12). Furthermore, the Pu-238/Pu-239,240 activity ratio in the top 1 cm of a sediment core from Station New4 in the inner part of Stepovogo Fjord was 0.23 ±0.05, with ratios greater than 0.10 observed in all 1 cm slices in the top 7 cm (Figure 4.23). Activity ratios comparable to typical surface sediments in were observed in all slices from another core from the inner part of Stepovogo Fjord (St. 18) as was the case for a core from the Pechora Sea (Figure 4.23). The overall range of Pu-238/Pu-239,240 activity ranges observed for sediments from the inner part of Stepovogo Fjord in 2012 was in good agreement with a previously reported range of 0.045 to 0.18 for sediments collected in the same area during the previous investigations in the 1990s (Oughton et al., 2004).

Based on earlier work by Hardy et al. (1973) and allowing for the decay of Pu-238, expected 2012 global fallout activity ratios of Pu-238/Pu-239,240 between 70 and 80˚N would be 0.018 ±0.003. Due to the associated uncertainties on activity ratios for the majority of samples, it is difficult to draw any firm conclusions as to the source(s) of Pu isotopes observed in sediments in Stepovogo Fjord in 2012. Other possible sources of Pu isotopes include long range transport of discharges from European reprocessing plants, local fallout from nuclear weapon tests on Novaya Zemlya and dumped waste within Stepovogo Fjord and the wider Kara Sea region. Additional contributions of Pu isotopes from any of these sources would result in higher Pu-238/Pu-239,240 activity ratios than would be expected from global fallout alone.

In the case of the two sampling stations in the inner part of Stepovogo Fjord (St. New 4 and St. 12) where Pu-238/Pu-239,240 activity ratios greater than 0.1 were observed in surface and deeper sediments, there is evidence to suggest the possible influence of sources of Pu isotopes within Stepovogo Fjord. That such activity ratios were only observed at two sampling stations may indicate strongly localised phenomena such as leakages from individual dumped containers.

Pu-240/Pu-239 atom ratios were measured in only a few sediment samples with ratios of 0.196 ±0.038 and 0.149 ±0.040 for surface sediments from the outer part (St. 36) and inner part (St. 18) of Stepovogo Fjord respectively. The Pu-240/Pu-239 atom ratio in a single surface sediment collected from the deck of the nuclear submarine K-27 with the ROV was 0.179 ±0.070. Pu-240/Pu-239 atom ratios in a core from the inner part of Stepovogo Fjord (St. 18) ranged between 0.113 ±0.032 to 0.183 ±0.040 in the top 7 cm (Figure 4.24). Observed Pu-240/Pu-239 atom ratios were predominately in good agreement with a reported mean global fallout Pu-240/Pu-239 atom ratio of 0.180 ±0.014 for the Northern hemisphere between 30 and 71˚N (Kelley et al., 1999). However, the variations in Pu-240/Pu-239 atom ratios in the upper sediments of the core from St. 18 would indicate an influence of a low Pu-240/Pu-239 atom ratio signal. In contrast, Oughton et al. (2004) reported Pu-240/Pu-239 atom ratios of 0.17 to 0.28 for surface sediments from the inner part of Stepovogo Fjord collected during the previous investigations in the 1990s indicating the influence of a high Pu-240/Pu-239 atom ratio signal. The combination of Pu-238/Pu-239,240 activity ratios and Pu-240/Pu-239 atom ratios observed in this study and those previously reported would suggest several sources of Pu isotopes to sediments in Stepovogo Fjord including the possibility of waste from different nuclear sources.

The activity ratios of Am-241/Pu-239,240 in all surface sediments from Stepovogo Fjord were broadly similar with mean activity ratios based on typical activity concentrations of 0.54 ±0.09 for the entrance, 0.53 ±0.09 for the outer part and 0.65 ±0.12 for the inner part (Table 4.18). These values were higher than the Am-241/Pu-239,240 activity ratio (0.29 ±0.12) derived for a surface sediment sample from the Pechora Sea. Am-241/Pu-239,240 activity ratios generally increased with sediment depth in cores from the Pechora Sea and close to the nuclear submarine K-27 in the outer part of Stepovogo Fjord (K27-1), while

cores from the inner part of Stepovogo Fjord (St. New4 and St. 12) showed variable Am-241/Pu-239,240 activity ratios (Figure 4.23). Differences in Am-241/Pu-239,240 activity ratios between sampling stations would appear to be primarily influenced by variations in Pu-239,240 activity concentrations.

Table 4.16. *Pu-238, Pu-239,240 and Am-241 activity concentrations (Bq/kg d.w.) in surface sediments from the Pechora Sea and Stepovogo Fjord in 2012.*

Sediment	Pu-238[1] (Bq/kg d.w.)			Pu-239,240[2] (Bq/kg d.w.)			Am-241[1] (Bq/kg d.w.)		
	n	Mean (±SD)	Range	n	Mean (±SD)	Range	n	Mean (±SD)	Range
Pechora Sea (St. 1)	1	0.053 ±0.033[3]	-	1	0.94 ±0.27[4]	-	1	0.33 ±0.12[3]	-
Entrance to Stepovogo Fjord	3	0.007 ±0.003	0.005 - 0.010	5	0.29 ±0.06	0.22 - 0.37	5	0.12 ±0.04	0.08 - 0.19
St. 44	1	1.6 ±0.1[3]	-	1	1.7 ±0.1[3]	-	-	-	-
Outer part of Stepovogo Fjord	10	0.019 ±0.007	0.010 - 0.030	10	0.49 ±0.11	0.31 - 0.69	10	0.23 ±0.05	0.14 - 0.31
St. 39	-	-	-	1	1.3 ±0.2[3]	-	-	-	-
ROV sample from deck of K-27	1	<0.023[3]	-	1	1.2 ±0.4[4]	-	1	0.14 ±0.03[3]	-
Inner part of Stepovogo Fjord	12	0.022 ±0.008	0.008 - 0.040	13	0.49 ±0.11	0.36 - 0.76	13	0.30 ±0.06	0.14 - 0.36
St. New4	1	0.13 ±0.03[3]	-	1	0.71 ±0.12[3]	-	1	0.58 ±0.17[3]	-
ROV sample close to dumped container	1	0.038 ±0.017[3]	-	1	0.46 ±0.09[4]	-	1	0.32 ±0.04[3]	-

Stated activity concentrations based on all available data for each sampling station. Uncertainties on all individual measurements were typically less than 70% (Pu-238), 20% (Pu-239,240) and 30% (Am-241). n - number of sampling stations.

1 - Norwegian and IAEA data.
2 - Norwegian, Russian and IAEA data.
3 - Based on a single sample and stated with associated uncertainty.
4 - Mean of two measurements of a single sample and stated with propagated uncertainty.

Table 4.17. *Inventories of Pu-238, Pu-239,240 and Am-241 (Bq/m² d.w.) in sediment cores from the Pechora Sea and Stepovogo Fjord in 2012.*

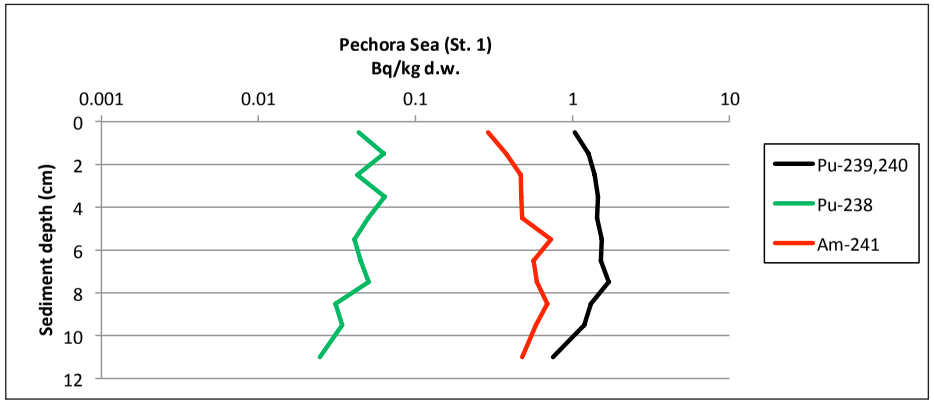
	Core depth (cm)	Pu-238 (Bq/m² d.w.)	Pu-239,240 (Bq/m² d.w.)	Am-241 (Bq/m² d.w.)
Pechora Sea (St. 1)	12	4.3 ±0.5	130 ±0.5	54 ±0.4
Outer part of Stepovogo Fjord				
K27-1	10	-	42 ±5	19 ±2
Inner part of Stepovogo Fjord				
St. New4	9	9.9 ±0.9	77 ±4	50 ±3
St. 12	14	0.98 ±0.36	21 ±2	14 ±1
St. 18	11	1.01 ±0.09	22.7 ±0.4	-

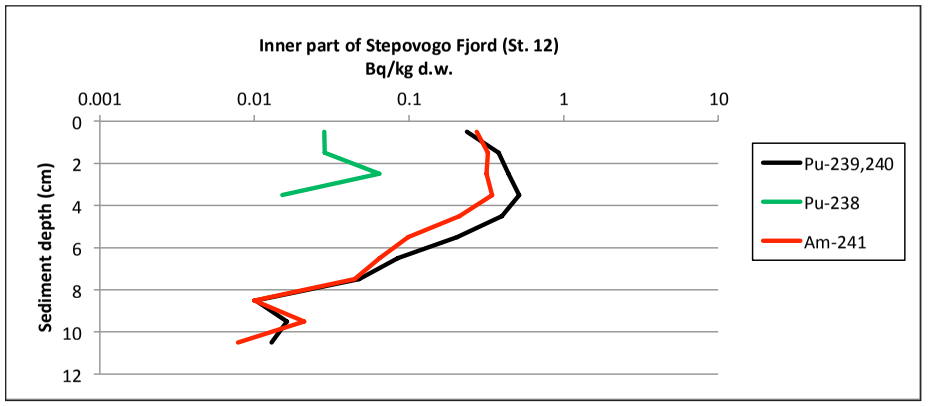
Inventories stated with propagated uncertainties. All Norwegian data, except for St. 18 (IAEA)

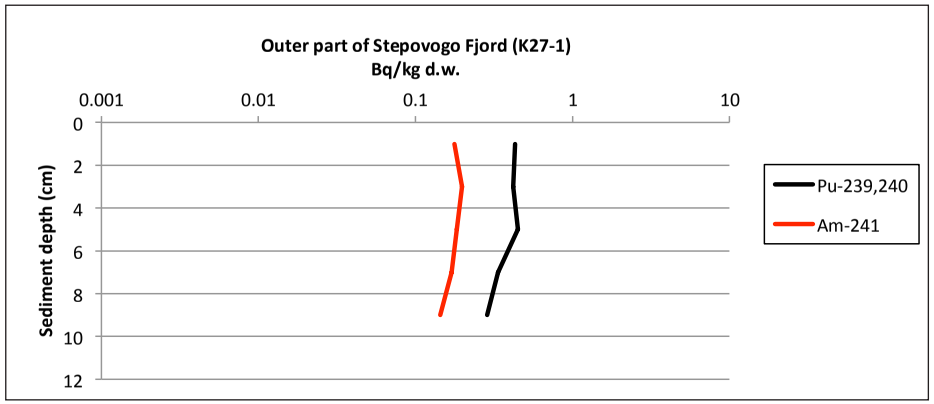
Table 4.18. *Pu-238/Pu-239,240 and Am-241/Pu-239,240 activity ratios in surface sediments from the Pechora Sea and Stepovogo Fjord in 2012.*

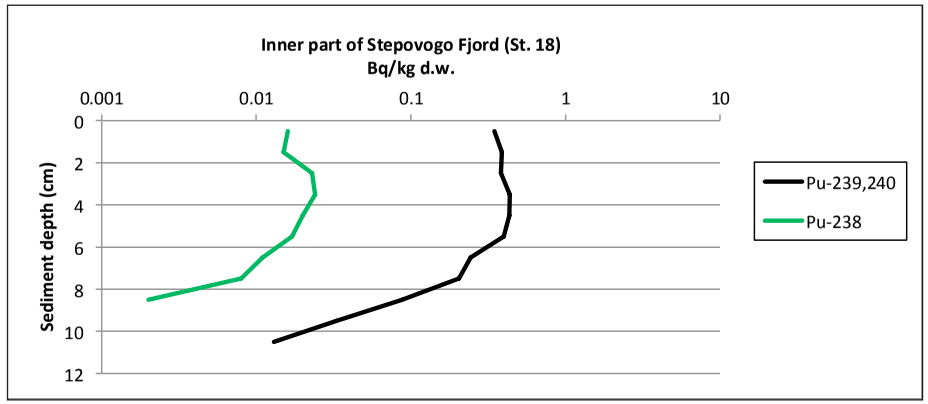
Sediment	Pu-238/Pu-239,240[1]			Am-241/Pu-239,240[2]		
	n	Mean (±SD)	Range	n	Mean (±SD)	Range
Pechora Sea (St. 1)	1	0.046 ±0.031	-	1	0.29 ±0.12	-
Entrance to Stepovogo Fjord	3	0.039 ±0.021	0.023 - 0.063	5	0.54 ±0.09	0.43 - 0.66
St. 44	1	0.95 ±0.09	-	-	-	-
Outer part of Stepovogo Fjord	10	0.044 ±0.016	0.029 - 0.074	10	0.53 ±0.09	0.35 - 0.66
ROV sample from deck of K-27	-	-	-	1	0.47 ±0.15	-
Inner part of Stepovogo Fjord	10	0.045 ±0.011	0.029 - 0.062	10	0.65 ±0.12	0.49 - 0.84
St. New4	1	0.18 ±0.06	-	1	0.82 ±0.27	-
St. 12	1	0.12 ±0.04	-	1	0.97 ±0.43	-
St. 13	-	-	-	1	1.11 ±0.24	-
ROV sample close to dumped container	1	0.066 ±0.031	-	1	0.56 ±0.10	-

Stated activity ratios based on data from one sediment sample per sampling station. Propagated uncertainties on all individual Pu-238/Pu-239-240 and Am-241/Pu-239,240 activity ratios were typically less than 70% and 30% respectively. n - number of sampling stations. Where n=1, activity ratio stated with propagated uncertainty.
1 - Norwegian and IAEA data.
2 - Norwegian data.









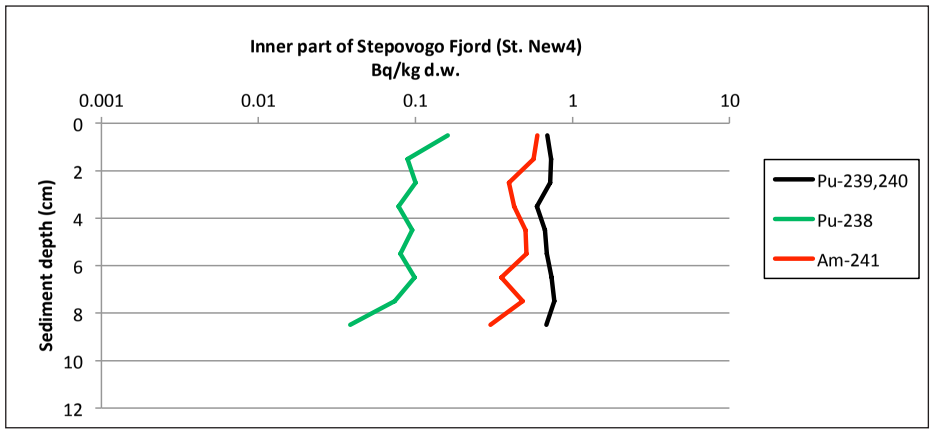


Figure 4.22. *Pu-238, Pu-239,240 and Am-241 activity concentrations (Bq/kg d.w.) in sediment cores (0-1 cm or 0-2 cm slices) from station 1 in the Pechora Sea compared to K27-1 from the outer part of Stepovogo Fjord and St. New4, St. 12 and St. 18 from the inner part of Stepovogo Fjord in 2012. Uncertainties on all individual measurements were typically less than 40% (Pu-238), 20% (Pu-239,240) and 30% (Am-241). All data Norwegian except for St. 18 (IAEA). Note use of logarithmic scale for activity concentrations.*

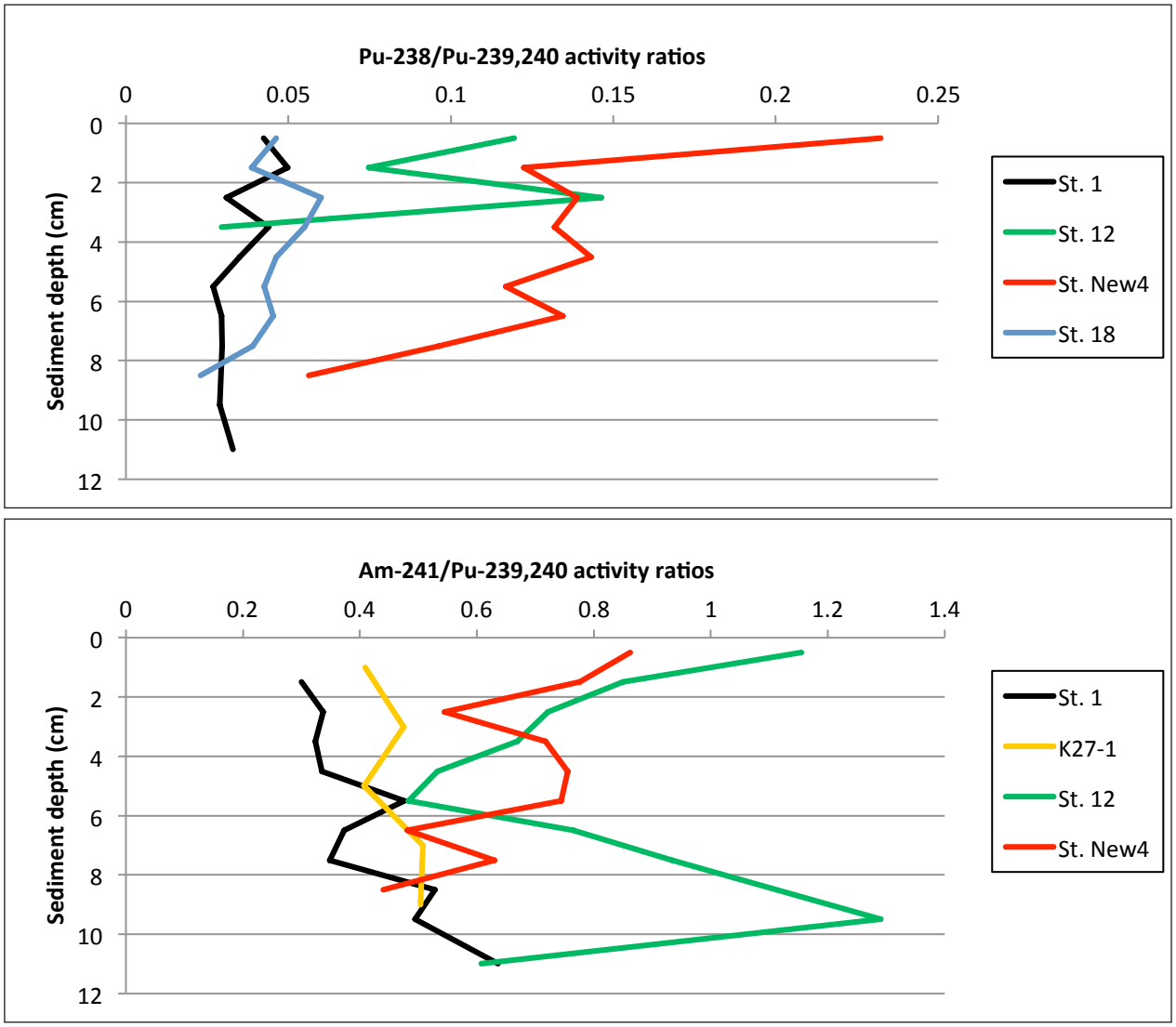


Figure 4.23. *Pu-238/Pu-239,240 and Am-241/Pu-239,240 activity ratios in sediment cores (0-1 cm or 0-2 cm slices) from St. 1 in the Pechora Sea compared to K27-1 from the outer part of Stepovogo Fjord and St. New4, St. 12 and St. 18 from the inner part of Stepovogo Fjord in 2012. Propagated uncertainties on all individual Pu-238/Pu-239-240 and Am-241/Pu-239,240 activity ratios were typically less than 40% and 30% respectively. All data Norwegian except for St. 18 (IAEA).*

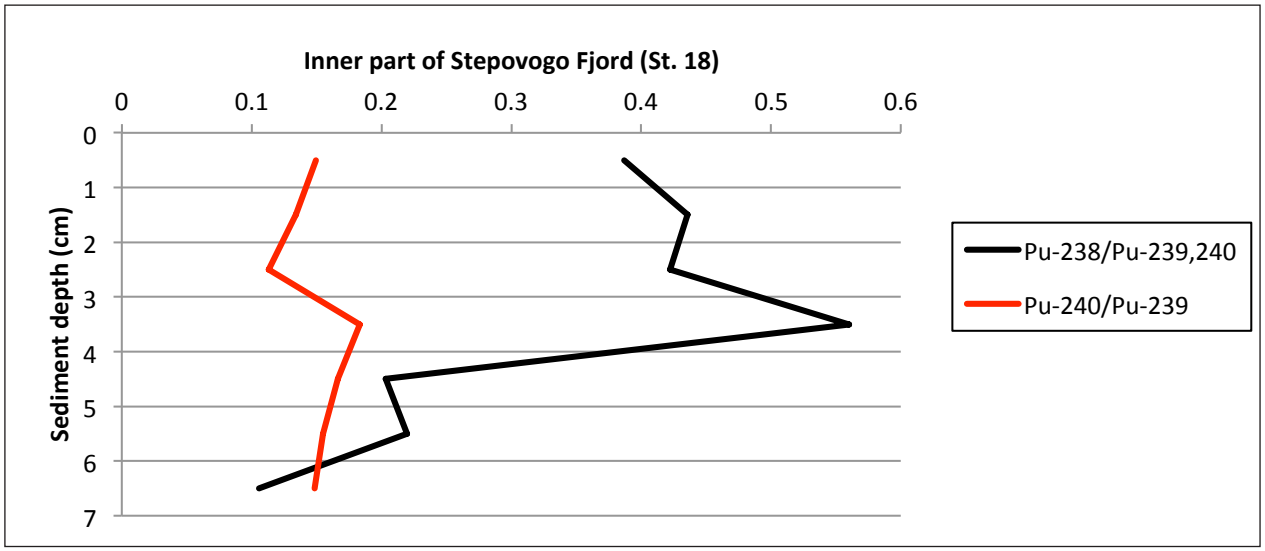
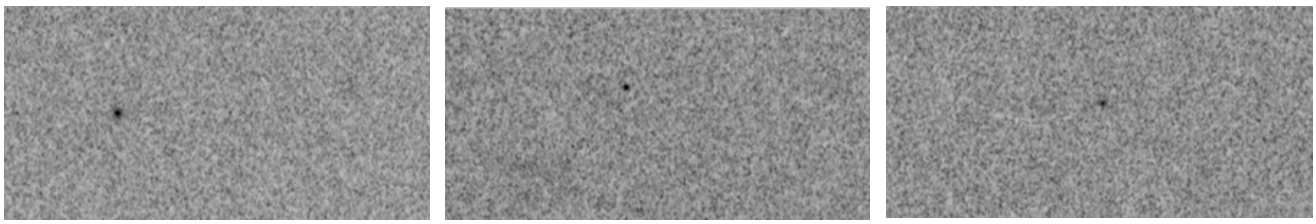


Figure 4.24. *Pu-238/Pu-239,240 activity ratios and Pu-240/Pu-239 atom ratios for St. 18 from the inner part of Stepovogo Fjord in 2012 (Norwegian data). Propagated uncertainties on individual Pu-238/Pu-239-240 activity ratios and Pu-240/Pu-239 atom ratios were typically less than 40% and 20% respectively.*

4.4.5 Autoradiography of sediments

Fourteen sediment samples were chosen for autoradiography based on results for Cs-137. Of these 14 samples, 3 samples revealed hot spots indicating the presence of radioactive particles (Figure 4.25). These particles will be isolated and characterised in order to identify the radionuclide and elemental composition and distribution. Radioactive particles containing Co-60 were identified in sediment samples taken close to dumped containers in the inner part of Stepovogo Fjord in the previous investigations in 1994 (JRNEG, 1996).



St. 14: 1 to 2 cm *St. 18: 2 to 3 cm* *St. 18: 3 to 4 cm*

Figure 4.25. *Digital autoradiograms of sediment samples from the inner part of Stepovogo Fjord in 2012 showing hot spots indicating radioactive particles (Norwegian data).*

4.4.6 Trace elements

The concentrations of trace elements were typically higher in surface sediments from Stepovogo Fjord than was observed in a single sediment sample from the Pechora Sea, with the exception of As, Sr and Th (Table 4.19). Within Stepovogo Fjord, all trace elements showed a similar trend of varying degrees of increasing concentrations in surface sediments from the entrance, to the outer part and inner part of Stepovogo Fjord with the exception of Cr which showed the reverse trend. In the case of Mn and Mo, concentrations in surface sediments from the inner part of Stepovogo Fjord were markedly higher than values for the outer part and entrance to the fjord. Assuming that Mn in surface sediments in Stepovogo Fjord is primarily derived from terrestrial run off, it is interesting to note that concentrations of Co, Ni, Zn, Cu, As, Sr, Mo, Cd and Pb in surface sediments from the 7 sampling stations in inner part of Stepovogo Fjord are positively related to Mn (R^2 values of 0.54 to 0.92), while concentrations of Bi, Th and U were inversely related (R^2 values of 0.53 to 0.89). Sorption to and co-precipitation with Mn oxides has been shown to be the main control on other trace elements in mixed freshwater and marine environments (e.g. Turner, 2000).

Profiles of U in sediment cores from the inner part of Stepovogo Fjord showed overall trends of decreasing concentrations with depth whereas the profile of U from a single core in the Pechora Sea showed increasing concentrations with depth (Figure 4.26). However, atom ratios of U-235/U-238 in all sediment samples were similar and in agreement with observed seawater values at 0.0073 ±0.0001.

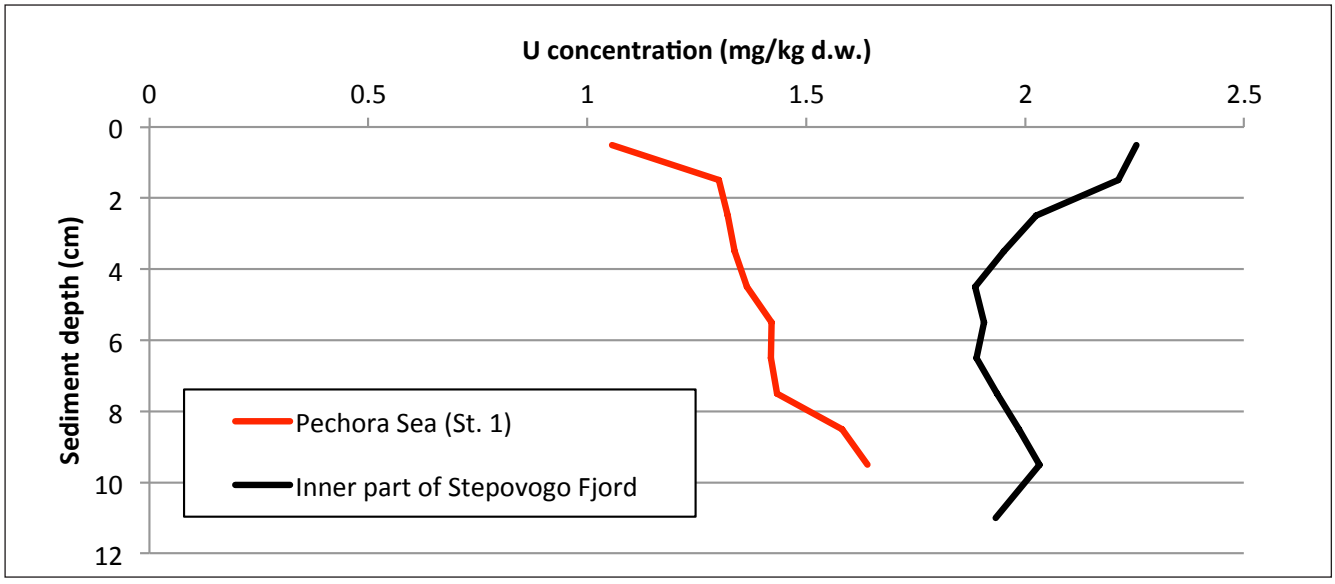


Figure 4.26. *Comparison of U concentration profiles in a core from the Pechora Sea (St. 1) and mean values for cores (n=7) from the inner part of Stepovogo Fjord in 2012 (Norwegian data).*

Table 4.19. *Mean (±SD) concentration of trace elements in surface sediments (0 to 1 cm) from the Pechora Sea and Stepovogo fjord in 2012[1].*

	Pechora Sea	Stepovogo Fjord		
	(St. 1)	Entrance	Outer part	Inner part
	n=1	n=2	n=2	n=7
As (mg/kg)	35	34 ±40	36 ±1	76 ±69
Bi (mg/kg)	0.12	0.11 ±0.03	0.21 ±0.00	0.21 ±0.02
Cd (mg/kg)	0.1	0.1 ±0.0	0.2 ±0.0	0.5 ±0.3
Co (mg/kg)	8.8	25 ±15	26 ±0	58 ±41
Cr (mg/kg)	68	136 ±14	122 ±8	104 ±19
Cu (mg/kg)	11	29 ±9	51 ±1	66 ±10
Mn (g/kg)	0.3	1.9 ±1.8	3.5 ±0.9	29 ±18
Mo (mg/kg)	0	1.2 ±0.8	2.4 ±0.6	36 ±22
Ni (mg/kg)	26	87 ±6	98 ±4	118 ±23
Pb (mg/kg)	12	9.8 ±5.4	20 ±0	22 ±2
Sr (mg/kg)	58	45 ±21	60 ±3	135 ±93
Th (mg/kg)	5.7	3.8 ±0.2	5 ±0	5.2 ±0.5
U (mg/kg)	1.1	1.1 ±0.2	1.7 ±0.0	2.3 ±0.4
Zn (mg/kg)	54	89 ±15	124 ±0	157 ±15

n - number of sampling stations. All results expressed as d.w.
1 - Norwegian data - where SD is given as 0, 0.0 and 0.00, SD was less than 0.5, 0.05 and 0.005, respectively. Uncertainties on individual measurements were typically between 2% and 5%.

4.5 Radionuclides and trace elements in biota

4.5.1 Cs-137

The activity concentrations of Cs-137 in all biota sampled were low and in many cases below detection limits. Direct comparison of seaweed and benthic invertebrates from the inner and outer parts of Stepovogo Fjord is therefore difficult, although the activity concentrations of Cs-137 in the sea urchin *Strongylocentrotus droebachiensis* and the brown algae *Desmarestia aculeda* were somewhat higher in samples from the inner part of Stepovogo Fjord (Tables 4.20 and 4.21). The activity concentrations of Cs-137 in individual seaweed samples from the outer part of Stepovogo Fjord in 2012 were below detection limits whereas two seaweed species were reported with Cs-137 activity concentrations of 3.4 ±0.2 and 2.5 ±0.4 Bq/kg (d.w.) in the previous investigation in 1993 (JRNEG, 1996). The activity concentrations of Cs-137 in all fish sampled from Stepovogo Fjord were low and similar to those observed in cod *(Gadus morhua)* from the Pechora Sea (Table 4.22). The observed activity concentrations of Cs-137 in all seaweed, benthic invertebrates and fish sampled in Stepovogo Fjord in 2012 were comparable to that reported for similar biota from the Norwegian Sea and Barents Sea in 2012 (NRPA, 2014). The activity concentrations of Cs-137 in muscle, liver, kidney, and stomach from the ringed seal *(Pusa hispida)* caught in the outer part of Stepovogo Fjord, ranged between 0.12 ±0.02 and 0.18 ±0.04 Bq/kg (f.w.). These values are lower than reported activity concentrations of Cs-137 in ringed seals sampled in Svalbard in 2003 (Gwynn et al., 2005).

4.5.2 Tc-99

The activity concentrations of Tc-99 in seaweed sampled in Stepovogo Fjord (Table 4.20) varied between 1.5 and 52.4 Bq/kg (d.w.) which was comparable with the range of Tc-99 activity concentrations reported for different seaweeds from the Norwegian Sea in 2012 (NRPA, 2014). All activity concentrations of Tc-99 in selected benthic invertebrates were low with similar values in samples of sea urchin *(Strongylocentrotus droebachiensis)* from the outer (0.32 ±0.03 Bq/kg f.w.) and inner parts (0.33 ±0.03 Bq/kg f.w.) of Stepovogo Fjord. In other benthic invertebrates from the outer part of Stepovogo Fjord, the activity concentration of Tc-99 in one bulk sample of brittlestars and seastars (Ophiuroidea and Asteroidea spp.) was 0.17 ±0.05 Bq/kg (f.w.), while for samples of the molluscs *Neptunea antiqua* and *N. despecta*, activity concentrations of Tc-99 were 0.59 ±0.05 and 0.58 ±0.05 Bq/kg (f.w.) respectively.

Table 4.20. *Cs-137 and Tc-99 activity concentrations (Bq/kg d.w.) in bulk samples of seaweed from Stepovogo Fjord[1].*

Species	Cs-137 (Bq/kg d.w.)	Tc-99 (Bq/kg d.w.)
Outer part of Stepovogo Fjord		
Fucus evanescens	<0.2	52.4 ±2.7
Desmarestia aculeda	<8.3	-
Laminaria sp.	<3.1	-
Inner part of Stepovogo Fjord		
Fucus evanescens	<0.40	5.5 ±0.3
Desmarestia aculeda	4.1 ±0.3	20.9 ±1.1
Saccharina latissima	0.3 ±0.2	1.5 ±0.2

1 - Norwegian data.

Table 4.21. *Cs-137 activity concentrations (Bq/kg f.w.) in benthic invertebrates from Stepovogo Fjord[1].*

Species	Sample type	Tissue	n	Cs-137 (Bq/kg f.w.)
Outer part of Stepovogo Fjord				
Axionice maculata	Polychaete	Worms and tubes	Bulk	0.71 ±0.11
Neptunea antiqua	Mollusc	Soft parts	119	<0.08
Neptunea despecta	Mollusc	Soft parts	50	<0.08
Mya arenaria	Mollusc	Soft parts	74	<0.51
Polyplacophora sp.	Mollusc	Whole animal	64	<0.64
Ophiuroidea and Asteroidea spp.	Echinoderm	Whole animal	Bulk	<0.11
Strongylocentrotus droebachiensis	Echinoderm	Soft parts	36	0.08 ±0.05
Strongylocentrotus droebachiensis	Echinoderm	Shell[2]	36	<0.10
Bryozoa	Bryozoan	Whole animal	Bulk	<0.65
Inner part of Stepovogo Fjord				
Strongylocentrotus droebachiensis	Echinoderm	Soft parts	21	0.60 ±0.08
Strongylocentrotus droebachiensis	Echinoderm	Shell	21	0.14 ±0.132

1 - Norwegian data. n - number of samples.
2 - Results expressed as Bq/kg (d.w.).

Table 4.22. *Cs-137 activity concentrations (Bq/kg f.w.) in fish from the Pechora Sea and Stepovogo Fjord[1].*

Species	Common name	Tissue	n	Cs-137 (Bq/kg f.w.)
Pechora Sea				
Gadus morhua	Cod	Muscle	5	0.13 ±0.02
Outer part of Stepovogo Fjord				
Gadus morhua	Cod	Muscle	1	<0.28
Eleginus nawaga	Navaga	Muscle	6	<0.20
Hippoglossoides platessoides	Long rough dab	Muscle	2	<0.22
Lycodes reticulatus	Arctic eelpout	Muscle	4	0.15 ±0.13
Cottidae sp.	Sculpin	Muscle	17	0.13 ±0.10
Myoxocephalus scorpius	Shortcorn sculpin	Muscle	5	<0.10
Eleginus nawaga	Navaga	Whole fish	48	0.08 ±0.02
Myoxocephalus scorpius	Shortcorn sculpin	Whole fish	15	0.06 ±0.03
Inner part of Stepovogo Fjord				
Eleginus nawaga	Navaga	Whole fish	156	0.11 ±0.03
Myoxocephalus scorpius	Shortcorn sculpin	Muscle	9	<0.22
Hippoglossoides platessoides	Long rough dab	Muscle	2	<0.15

1 - Norwegian data. n - number of samples.

4.5.3 Sr-90

Activity concentrations of Sr-90 in selected biota from Stepovogo Fjord (Table 4.23) were low and ranged from 0.05 ±0.04 Bq/kg (f.w.) in muscle from Shortcorn sculpin *(Myoxocephalus scorpius)* to 1.40 ±0.62 Bq/kg (d.w.) in sea urchin shells *(S. droebachiensis)*. These values were within the range of Sr-90 activity concentrations reported in similar biota from the Irish Sea in 2011 (RIFE, 2012).

4.5.4 Pu-239,240 and Am-241

Activity concentrations of Pu-239,240 and Am-241 in selected biota from Stepovogo Fjord (Table 4.23) were low and ranged from 0.001 to 0.37 Bq/kg (f.w.) and 0.004 to 0.111 Bq/kg (f.w.), respectively, with the highest values observed in the seaweed Fucus evanescens. The activity concentrations of Pu-239,240 and Am-241 in muscle, liver, kidney, and stomach from the ringed seal *(P. hispida)* caught in the outer part of Stepovogo Fjord were all below detection limits. The observed values were typically far lower than Pu-239,240 and Am-241 activity concentrations reported in similar biota from the Irish Sea in 2011 (RIFE, 2012).

4.5.5 Autoradiography of biota

Autoradiography was performed on dried soft tissues of two mussels *(Mya arenaria)* that were collected together with a single surface sediment from the deck of the nuclear submarine K-27 with the ROV. Hotspots indicating the presence of radioactive particles were observed in the mussel tissue (Figure 4.27). These particles will be isolated and characterised in order to identify the radionuclide and elemental composition and distribution.

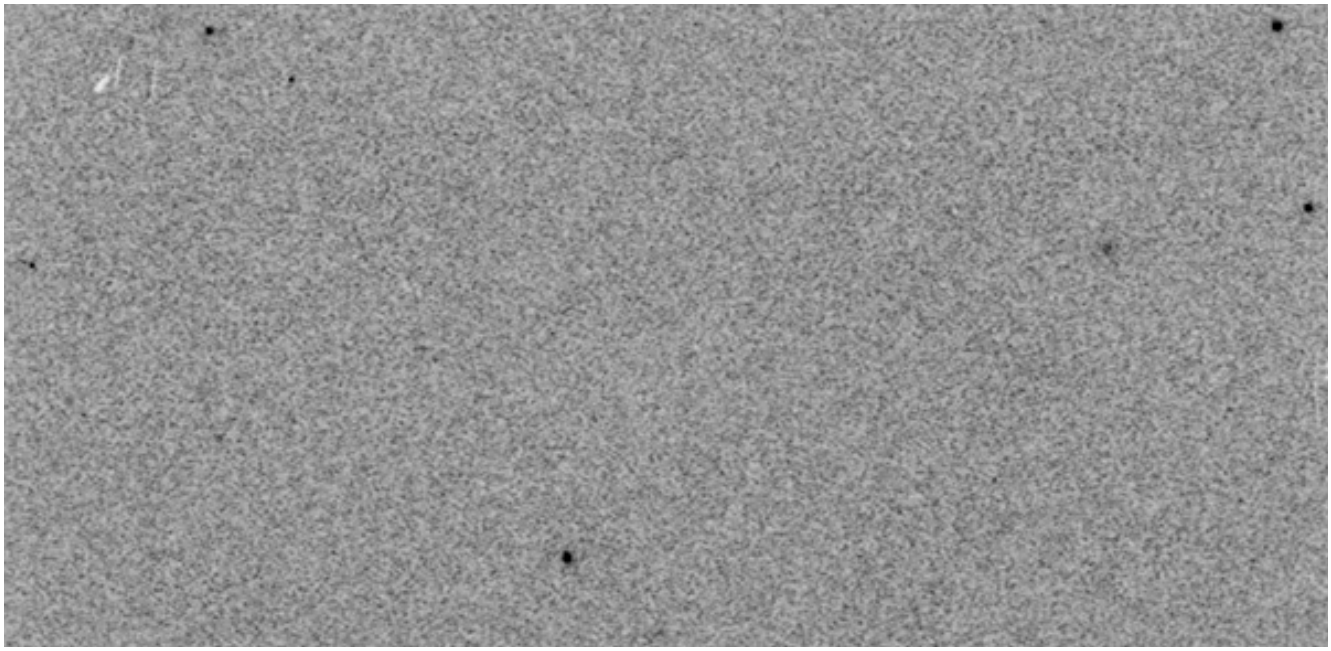


Figure 4.27. *Digital autoradiogram of soft tissues of two mussels (Mya arenaria) collected from the deck of the nuclear submarine K-27 with the ROV in 2012 showing hotspots indicating the presence of radioactive particles (Norwegian data). Exposure time was 7 days.*

Table 4.23. *Sr-90, Pu-239,240 and Am-241 activity concentrations (Bq/kg d.w. or f.w.) in biota from Stepovogo Fjord in 2012[1].*

Species	Sample type	Tissue	n	Sr-90 (Bq/kg)	Pu-239,240 (Bq/kg)	Am-241 (Bq/kg)
Outer part of Stepovogo Fjord						
Neptunea antiqua	Mollusc	Soft parts[2]	119	0.10 ±0.05	0.003 ±0.002	0.008 ±0.003
Neptunea despecta	Mollusc	Soft parts[2]	50	0.06 ±0.04	0.002 ±0.002	0.004 ±0.003
Ophiuroidea and Asteroidea spp.	Echinoderm	Whole animal[2]	Bulk	-	0.036 ±0.008	0.033 ±0.008
Fucus evanescens	Seaweed	Whole plant[3]	Bulk	0.74 ±0.20	-	-
Myoxocephalus scorpius	Fish	Muscle[2]	5	0.05 ±0.04	0.001 ±0.001	0.007 ±0.002
Eleginus nawaga	Fish	Muscle[2]	48	-	0.002 ±0.002	0.009 ±0.003
Myoxocephalus scorpius	Fish	Bone[3]	5	0.60 ±0.37	-	-
Inner part of Stepovogo Fjord						
Strongylocentrotus droebachiensis	Echinoderm	Soft parts[2]	21	-	0.028 ±0.004	0.019 ±0.003
Strongylocentrotus droebachiensis	Echinoderm	Shell[3]	21	1.40 ±0.62	0.024 ±0.009	0.040 ±0.015
Fucus evanescens	Seaweed	Whole plant[3]	Bulk	0.37 ±0.19	0.117 ±0.020	0.029 ±0.013
Saccharina latissima	Seaweed	Whole plant[3]	Bulk	0.25 ±0.17	0.017 ±0.008	0.045 ±0.014
Desmarestia aculeda	Seaweed	Whole plant[3]	Bulk	0.68 ±0.29	0.370 ±0.045	0.111 ±0.019

1 - Norwegian data. n - number of samples.
2 - Results expressed as Bq/kg (f.w.).
3 - Results expressed as Bq/kg (d.w.).

4.5.6 Trace elements

The concentrations of trace elements were determined in gills and liver of two fish species sculpin (Cottidae sp.) and navaga *(Eleginus nawaga)* collected in the outer and inner parts of Stepovogo Fjord (Table 4.24). A degree of variability in concentrations between samples for each species and location was observed, but in general there was little difference in mean concentrations of trace elements in samples of either fish species between the outer and inner parts of the fjord. Where variation between species was observed, mean concentrations of As, Co, Cu, and Fe in gills and/or liver were typically higher in sculpin than navaga, whereas the opposite was true for mean concentrations of Mn in gills. In the case of Cd and Zn, mean concentrations were higher in gills of navaga but higher in livers of sculpin.

The mean concentrations of U in gills of both fish species were similar whether from the outer or inner parts of Stepovogo Fjord. The concentrations of U in all liver samples were low and less than the quantification limit (0.001 μg/g). The atom ratios of U-235/U-238 in all gill samples were similar and in agreement with observed seawater and sediment values at 0.007 ±0.003.

Table 4.24. *Mean (±SD) concentration of trace elements (μg/g f.w.) in gills and liver of two fish species from the outer and inner parts Stepovogo Fjord in 2012[1].*

	Tissue	Outer part of Stepovogo Fjord		Inner part of Stepovogo Fjord	
		Sculpin (n=15)	Navaga (n=6)	Sculpin (n=7)	Navaga (n=16)
Al (μg/g f.w.)	Gill	4.7 ±4.5	3.4 ±2.3	10.9 ±15.1	6.6 ±5.2
As (μg/g f.w.)	Gill	6.8 ±3.5	1.3 ±0.4	3.8 ±2.7	1.2 ±0.3
	Liver	43.1 ±29.3	9.0 ±2.9	8.8 ±2.7	6.1 ±1.6
Cd (μg/g f.w.)	Gill	0.008 ±0.002	0.018 ±0.014	0.006 ±0.002	0.014 ±0.004
	Liver	0.16 ±0.09	0.05 ±0.03	0.12 ±0.03	0.03 ±0.02
Co (μg/g f.w.)	Gill	0.15 ±0.06	0.086 ±0.06	0.13 ±0.04	0.032 ±0.014
	Liver	0.11 ±0.04	0.06 ±0.02	0.14 ±0.09	0.05 ±0.02
Cr (μg/g f.w.)	Gill	<0.14	0.32 ±0.56	<0.14	<0.14
	Liver	0.02 ±0.04	0.01 ±0.01	0.01 ±0.01	0.01 ±0.01
Cu (μg/g f.w.)	Gill	1.52 ±0.4	0.56 ±0.12	1.6 ±0.52	0.5 ±0.08
	Liver	3.7 ±2.7	2.5 ±1.1	5.7 ±5.6	2.8 ±1.7
Fe (μg/g f.w.)	Gill	35.8 ±11.4	23.8 ±6.6	50.4 ±13.8	19.4 ±6.6
	Liver	38.9 ±25.8	34.6 ±11.2	42.3 ±17.9	22.5 ±6
Mn (μg/g f.w.)	Gill	4.1 ±1.3	13.3 ±5.6	5.2 ±1.7	13.4 ±2.3
	Liver	2.0 ±0.9	1.6 ±0.5	1.6 ±1.0	1.5 ±0.4
Ni (μg/g f.w.)	Gill	<0.12	0.4 ±0.62	<0.12	<0.12
	Liver	0.02 ±0.01	0.13 ±0.06	0.07 ±0.04	0.07 ±0.03
Pb (μg/g f.w.)	Gill	<0.06	<0.06	<0.06	<0.06
	Liver	0.091 ±0.234	0.02 ±0.009	0.004 ±0.001	0.007 ±0.005
Sr (μg/g f.w.)	Gill	59.8 ±13.4	62.4 ±16	45.8 ±10	49 ±9.6
	Liver	0.76 ±0.69	0.73 ±0.56	0.43 ±0.23	0.62 ±0.32
Th (μg/g f.w.)	Gill	0.0006 ±0.0004	0.0006 ±0.0012	0.002 ±0.001	0.0016 ±0.0014
U (μg/g f.w.)	Gill	0.0072 ±0.0024	0.0058 ±0.0022	0.0052 ±0.0024	0.0049 ±0.0014
	Liver	<0.001	<0.001	<0.001	<0.001
Zn (μg/g f.w.)	Gill	12.5 ±1.5	18.7 ±1.0	13.0 ±2.2	19.7 ±2.7
	Liver	31.1 ±9.4	16.6 ±1.6	31.4 ±6.8	15.6 ±4.2

Sculpin - Cottidae sp.; Navaga - Eleginus nawaga. n - number of fish.
1 - Norwegian data. Uncertainties on individual measurements were typically between 2% and 5%.

4.6 Derived parameters

4.6.1 Sedimentation distribution coefficients (K_d)

Sedimentation distribution coefficients (K_d) can be a useful indicator of the potential mobility of radionuclides and trace elements from contaminated sediments. Radionuclides and trace elements from a particular source may be present in the environment in different physico-chemical forms (e.g. low molecular mass species (LMM), colloids, particles) that can influence their bioavailability and ultimately determine the degree of doses, toxicity and effects that might be incurred by organisms. LMM species and colloids are generally considered to be mobile forms, while particles more readily retained in sediments. If mobile species of radionuclides and trace elements are present (low K_d values), transfer within an ecosystem will be a relatively rapid. If particles are present, (high K_d values), transfer within the ecosystem can be delayed or inhibited. In addition, the physico-chemical form of radionuclides and trace elements deposited in the environment can change with time due to chemical interactions in the environment or to particle weathering.

K_d values derived for all radionuclides and trace elements in this study were based on single water samples, so a degree of caution should be shown with regard to their interpretation. Despite this, K_d values for Cs-137, Sr-90 and Am-241 in surface sediments from Stepovogo Fjord (Table 4.25) were in general in agreement with values reported for these radionuclides by Fisher et al. (1999) for Stepovogo Fjord based on samples collected during previous investigations in 1994. K_d values for

Pu-239,240 were not reported previously for Stepovogo Fjord, but the values derived in this study compare well with values reported for the Kara Sea (Fisher et al., 1999). Overall, the variations in K_d values for individual radionuclides in the Stepovogo Fjord were small, indicating that radionuclides are in general homogenously distributed within sediments. K_d values for all radionuclides from the Pechora Sea were of a similar magnitude to those derived for Stepovogo Fjord with the exception of Sr-90 which was one to two orders of magnitude higher. With the exception of Sr-90, all K_d values for radionuclides from the Pechora Sea were in good agreement with recommended values for the open ocean (IAEA, 2004).

For trace elements (Table 4.26), the derived K_d value for Co for the inner part of Stepovogo Fjord was similar to values reported by Carroll et al., (1997) and Fisher et al., (1999) for Co-60, while the derived K_d value for Co for the outer part of Stepovogo Fjord was similar to the recommended value for ocean margins (IAEA, 2004). The derived K_d value for Mn in the inner part of Stepovogo Fjord was two orders of magnitude higher than the value for the outer part of the fjord and one order of magnitude higher than the recommended value for ocean margins (IAEA, 2004). Derived K_d values for Sr showed a similar trend with a higher value for the inner part of Stepovogo Fjord compared to the outer part of the fjord, but both K_d values were lower than the corresponding values for Sr-90. Derived K_d values for U for the outer and inner parts of Stepovogo Fjord were similar and an order of magnitude lower than the recommended value for ocean margins (IAEA, 2004). Derived K_d values for trace elements from the Pechora Sea were similar to those for the outer part of Stepovogo Fjord.

Table 4.25. *Derived sediment distribution coefficients (K_d) for radionuclides in the Pechora Sea and Stepovogo Fjord in 2012.*

	Pechora Sea	Stepovogo Fjord	
		Outer part	Inner part
Cs-137	3.7E+03	2.5E+03 - 6.2E+03	1.3E+03 - 4.3E+03
Sr-90	2.0E+03	1.5E+02 - 4.8E+02	6.8E+01 - 3.0E+02
Pu-239,240	2.8E+05	1.4E+05 - 3.1E+05	1.0E+05 - 2.2E+05
Am-241	4.1E+05	1.4E+05 - 3.1E+05	5.4E+04 - 1.4E+05

All K_d values based on activity concentration data from samples of bottom water (<0.45 µm) collected at one sampling station at each location and mean activity concentrations for surface sediments from sampling stations in each area. K_d values for Stepovogo Fjord expressed as a range.

Table 4.26. *Derived sediment distribution coefficients (K_d) for selected trace elements in the Pechora Sea and Stepovogo Fjord in 2012.*

	Pechora Sea	Stepovogo Fjord	
		Outer part	Inner part
Co	-	4.4E+05	2.0E+06
Cu	2.1E+04	4.1E+04	4.6E+04
Mn	3.7E+05	8.3E+05	1.7E+07
Sr	7.2E+00	7.7E+00	1.7E+01
U	3.1E+02	4.9E+02	6.3E+02

All K_d values based on concentration data from samples of bottom water (<0.45 µm) collected at one sampling station at each location and mean concentrations for surface sediments from sampling stations in each area.

4.6.2 Bioconcentration factors (BCF)

Bioconcentration factors (BCF) can vary according to the chemical species, the biological species and different internal organs and tissues. The speciation of radionuclides and trace elements can affect their biological uptake, accumulation and biomagnification. For example, LMM-species can cross biological membranes, directly or indirectly after interactions with ligands or carrier molecules. Uptake in fish and invertebrates is commonly dependent on ionic species interacting with external organs (e.g. gills and skin) or through the digestion of food. In the case of filter feeding organisms, particles and colloids can be retained which may then undergo changes in bioavailability in the gut or be taken up by phagocytosis.

BCF values derived for all radionuclides and trace elements in this study were based on single water samples, so a degree of caution should be shown with regard to their interpretation. Where BCF values for similar organisms have been reported previously, the BCFs derived for radionuclides in this study were similar in some cases, but not all (Table 4.27). In comparison with recommended values for fish (IAEA, 2004), BCF values were in agreement for Pu-239,240 but one order of magnitude lower for Cs-137 and higher for Sr-90 and Am-241. For molluscs, BCF values for Sr-90, Pu-239,240 and Am-241 were similar to recommended values (IAEA, 2004), but BCFs for Tc-99 were one order of magnitude higher. In the case of seaweed, BCF values for Sr-90 and Cs-137 were similar to recommended values (IAEA, 2004), but one order of magnitude lower for Tc-99 and higher for Pu-239,240 and Am-241.

For trace elements, no clear difference in BCFs could be observed between fish of either species collected from either the outer or inner parts of Stepovogo Fjord (Table 4.28). BCFs for Co and Fe were similar for gills and livers for both sculpin and navaga, whereas for Cu, BCFs for gills and liver were similar for sculpin but one order of magnitude higher in liver than gills for navaga. In the case of Mn and Sr, BCFs in gills were typically one and two orders of magnitude higher, respectively, than BCFs for these trace elements in liver. In comparison with recommended values for fish (IAEA, 2004), BCF values for Mn, Sr and U in gills were in good agreement whereas BCFs for Co were one order of magnitude higher and one order of magnitude lower for Fe.

Table 4.27. *Derived biological concentration factors (BCF) for radionuclides in the Pechora Sea and Stepovogo Fjord in 2012.*

	Tissue	Cs-137	Sr-90	Tc-99	Pu-239,240	Am-241
Pechora Sea						
Fish	Muscle	9.5E+01	-	-	-	-
Outer part of Stepovogo Fjord						
Polychaete	Worms and tubes	3.9E+02	-	-	-	-
Mollusc	Soft parts	-	3.3E+01 - 5.5E+01	2.9E+03 - 3.0E+03	9.1E+02 - 1.4E+03	3.9E+03 - 7.9E+03
Echinoderm	Whole animal	-	-	8.5E+02	1.6E+04	3.3E+04
	Soft parts	4.4E+01	-	1.6E+03	-	-
Seaweed	Whole plant	-	4.3E+01	6.9E+04	-	-
Fish	Muscle	7.1E+01 - 8.2E+01	2.7E+01	-	4.5E+02 - 9.1E+02	6.9E+03 - 8.9E+03
	Whole fish	3.3E+01 - 4.4E+01	-	-	-	-
	Bone[1]	-	3.3E+02	-	-	-
Seal	Muscle	6.6E+01	-	-	-	-
Inner part of Stepovogo Fjord						
Echinoderm	Soft parts	4.4E+01	-	1.2E+03	8.0E+03	7.3E+03
	Shell[1]	-	2.7E+02	-	6.9E+03	1.5E+04
Seaweed	Whole plant	3.7E+01 - 5.1E+02	1.5E+01 - 4.2E+01	1.4E+03 - 1.9E+04	1.4E+03 - 3.0E+04	4.7E+03 - 1.8E+04
Fish	Whole fish	8.0E+00	-	-	-	-

All BCFs based on activity concentration data from bottom water seawater samples (<0.45 µm), except for seaweed (surface water) and Tc-99 (unfiltered seawater), collected at one sampling station at each location and activity concentrations for each biota and tissue type in each area.
1 - Based on d.w.

Table 4.28. *Derived biological concentration factors (BCF) for selected trace elements in gills and liver of two fish species from the outer and inner parts Stepovogo Fjord in 2012.*

	Tissue	Outer part of Stepovogo Fjord		Inner part of Stepovogo Fjord	
		Sculpin	Navaga	Sculpin	Navaga
Al	Gill	9.4E+02	6.8E+02	3.8E+03	2.3E+03
Co	Gill	2.5E+03	1.5E+03	4.4E+03	1.1E+03
	Liver	1.9E+03	1.0E+03	4.8E+03	1.7E+03
Cu	Gill	1.2E+03	4.5E+02	1.1E+03	3.4E+02
	Liver	3.0E+03	2.0E+03	3.9E+03	1.9E+03
Fe	Gill	4.5E+03	3.0E+03	1.7E+03	6.6E+02
	Liver	4.9E+03	4.3E+03	1.4E+03	7.6E+02
Mn	Gill	9.7E+02	3.2E+03	3.1E+03	7.9E+03
	Liver	4.8E+02	3.8E+02	9.5E+02	8.9E+02
Sr	Gill	7.7E+00	8.0E+00	5.7E+00	6.1E+00
	Liver	9.8E-02	9.4E-02	5.4E-02	7.7E-02
U	Gill	2.1E+00	1.7E+00	1.4E+00	1.4E+00

Sculpin - Cottidae sp.; Navaga - Eleginus nawaga. All BCFs based on concentration data from bottom water seawater samples (<0.45 μm) collected at one sampling station at each location and mean concentrations for each fish and tissue type.

5. Overall conclusions

The joint Norwegian-Russian cruise to Stepovogo Fjord in 2012 to investigate the radioecological status of dumping sites for solid radioactive waste and the nuclear submarine K-27 was organized through the Norwegian-Russian expert group as one aspect of the greater cooperation between Norway and Russia with regard to nuclear safety and radiological environmental assessments. The joint Norwegian-Russian cruise in 2012 followed on from previous joint Norwegian-Russian cruises to dumping sites of radioactive waste in the Kara Sea and Novaya Zemlya fjords in 1992, 1993 and 1994.

In 2012, the nuclear submarine K-27 was observed lying upright and clear of bottom sediments at a depth of around 30 m in the outer part of Stepovogo Fjord with no obvious corrosion damage of the outer hull. Based on the in situ gamma measurements taken next to the submarine, the observed activity concentrations in seawater at St. 36 and the observed activity concentrations in sediment samples taken close to and in the area around the submarine in the outer part of Stepovogo Fjord, there was no indication of any leakage from the reactor units of K-27. No release of enriched uranium from the reactors of K-27 was indicated on the basis of U activity ratios in seawater and sediment, although further analysis (U-236 measurements) of samples will be required to confirm this situation.

In the inner part of Stepovogo Fjord, where around 2000 waste containers have been reported to be dumped (Sivintsev et al., 2005), only a limited number of such objects were inspected during the investigation. Therefore, it was not possible to draw any direct comparisons with observations of dumped containers in the previous investigations in 1993 and 1994. Equally, it was not possible on the basis of the 2012 investigation to provide any overall assessment of the status of such dumped objects in the inner part of Stepovogo Fjord.

With regard to the radioecological status of Stepovogo Fjord, activity concentrations of all radionuclides in seawater, sediment and biota in 2012 were in general lower than reported from the previous investigations in the 1990s and were comparable or lower than reported values for other marine areas for a similar time period.

However in 2012, the activity concentrations of Cs-137 and, to a lesser extent, those of Sr-90 remained elevated in bottom water from the inner part of Stepovogo Fjord compared with surface water and the outer part of Stepovogo Fjord. This is likely due to a combination of leakages from dumped containers, the subsequent remobilisation of these radionuclides from contaminated sediments and the reduced mixing and flushing of this bottom water with inflowing Kara Sea water. Peak activity concentrations of Cs-137 in sediment cores occurred at slightly deeper sediment depths than was observed in previous investigations in 1993 and 1994, indicating the slow burial of contaminated layers.

The general picture of activity ratios of Pu-238/Pu-239,240 in sediment in Stepovogo Fjord in 2012 indicated a combination of global fallout in conjunction with other sources of Pu isotopes that may include long transport of discharges from European reprocessing plants, local fallout from nuclear weapon tests on Novaya Zemlya and/or dumped waste within Stepovogo Fjord and the wider Kara Sea region. Deviations from expected Pu-238/Pu-239,240 activity ratios and Pu-240/Pu-239 atom ratios in some sediment samples from the inner part of Stepovogo Fjord observed in this study and earlier studies may indicate the possibility of dumped waste from different nuclear sources.

As activity concentrations of Cs-137 in surface sediments from the inner part of Stepovogo Fjord remain elevated, it is likely that sediments will continue to act in the future as a diffuse source of Cs-137 to bottom water in this part of the fjord. Due to the bathymetry and physical oceanography of the inner part of Stepovogo Fjord, any future releases from dumped containers lying in the deeper parts of the inner part of Stepovogo Fjord are likely to have only limited impacts on the wider marine environment. However, future releases from dumped containers in shallower areas in the inner part of Stepovogo Fjord may be more readily transported to the outer part of Stepovogo Fjord and further afield.

Although the current environmental levels of radionuclides in Stepovogo Fjord are not of immediate cause for concern, further monitoring of the situation is warranted. In particular, a better understanding of the amount, source and status of waste that has been dumped in the inner part of Stepovogo Fjord is required. Additionally, the situation with regard to the nuclear submarine K-27 in the outer part of Stepovogo Fjord should be followed, especially in connection with any future plans involving the recovery of K-27 from the fjord.

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