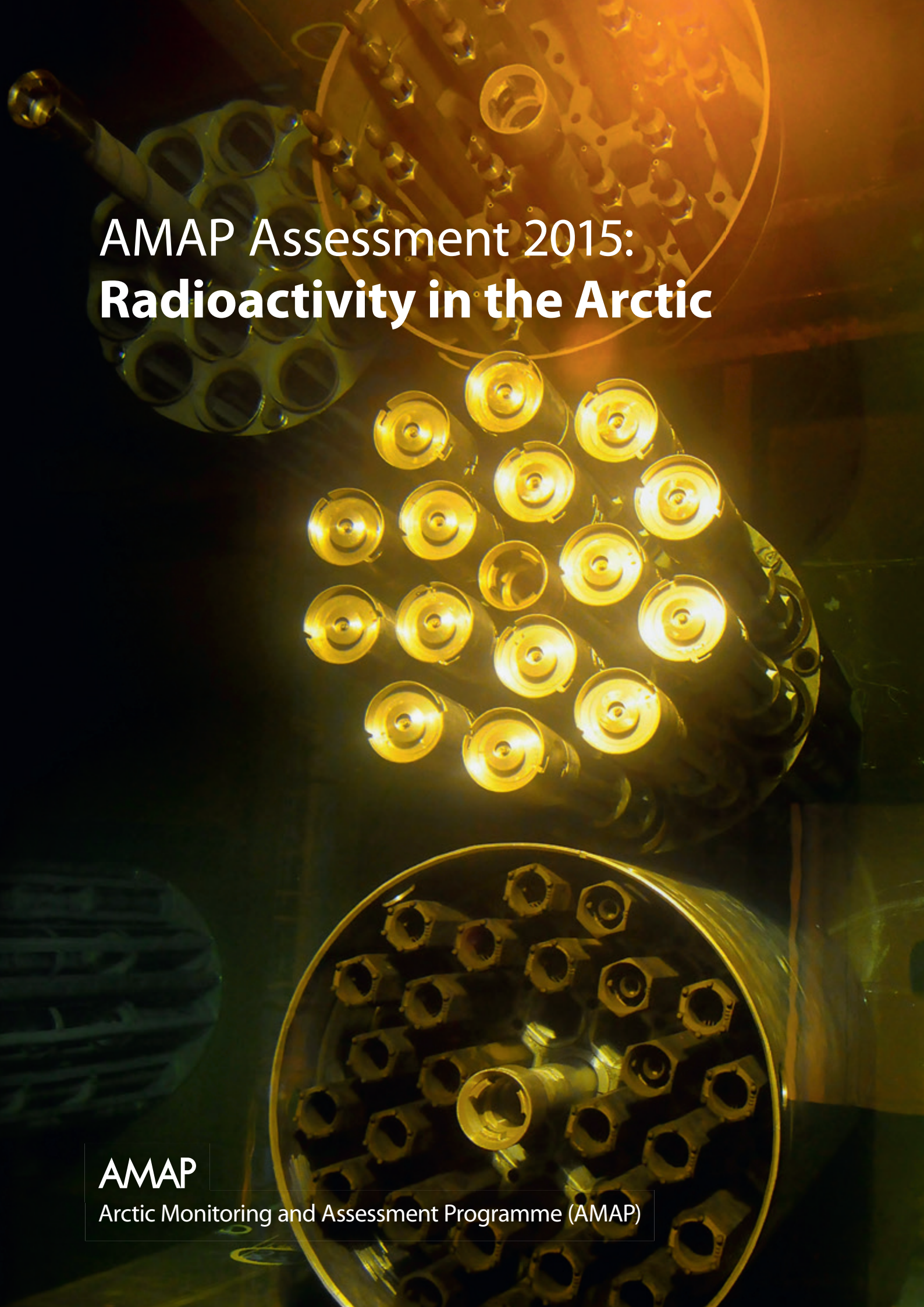

AMAP Assessment 2015: **Radioactivity in the Arctic**

AMAP

Arctic Monitoring and Assessment Programme (AMAP)

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Nuclear waste in a spent fuel pool at a nuclear reprocessing plant. This plant reprocesses spent fuel from nuclear power stations, ships and submarines, as well as plutonium from decommissioned nuclear weapons. Photographed at the Mayak RT-1 reprocessing facility, Chelyabinsk Region, Russia. (Sputnik/Science Photo Library)

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DEDICATION

Alexander Nikitin (1947 – 2014)

The scientific community is deeply saddened by the death of Alexander Nikitin who died on 21 December 2014 after battling a serious illness. His death is a great loss for international science. Alexander Nikitin was a Doctor of Sciences and worked as a chief research scientist at RPA "Typhoon" (Obninsk, Russia). He was a prominent scientist who focused on radiation monitoring of the environment. His work has made a fundamental contribution to the investigation of radioactive contamination of the Arctic and Far East seas, Siberian rivers and water bodies contaminated after the Chernobyl accident. He was directly involved in studies of radionuclides transport in the Ob-Irtysh river basin and impact of radioactive discharges from West European nuclear reprocessing facilities on Russian seas, and took part in the expedition survey of the Far East seas following the Fukushima Daiichi accident. Alexander Nikitin had many-years experience in international cooperation and joined the AMAP Radioactivity Expert Group in 1994. Since 1992 he participated in the Russian-Norwegian cruises to study radioactive contamination of the marine environment in the areas where nuclear wastes have been dumped in the Kara and Barents Seas and was the head of scientific research on many of these cruises. He was a member of scientific and organizational committees at numerous international conferences. He was an expert representing Russia on radioactive contamination assessment in the Arctic Monitoring and Assessment Programme (AMAP). Alexander Nikitin is author and co-author of more than 120 research papers, the results of his work were reported in, amongst others, the Journal of Environmental Radioactivity. He received several awards from Russia for excellence in science. Alexander Nikitin will forever remain in the hearts of those who were privileged to know him.

Yuri Tsaturov

AMAP Vice-Chair

Head of Russian delegation to AMAP

Contents

Preface

This assessment report presents the results of the 2015 AMAP Assessment of Radioactivity in the Arctic. This is the fourth AMAP assessment dealing with this issue and updates the assessments delivered in 1998, 2002 and 2009.

The Arctic Monitoring and Assessment Programme (AMAP) is a group working under the Arctic Council. The Arctic Council Ministers have requested AMAP to:

- produce integrated assessment reports on the status and trends of the conditions of the Arctic ecosystems
- identify possible causes for the changing conditions
- detect emerging problems, their possible causes, and the potential risk to Arctic ecosystems including indigenous peoples and other Arctic residents
- recommend actions required to reduce risks to Arctic ecosystems.

This report provides the accessible scientific basis and validation for the statements and recommendations made in the *Summary for Policy-makers: Arctic Pollution Issues 2015* report[i] that was delivered to Arctic Council Ministers at their meeting in Iqaluit, Canada in April 2015. It is also the basis for a related AMAP State of the Arctic Environment report *Arctic Pollution Issues 2015: Overview*[ii]. It includes extensive background data and references to the scientific literature, and details the sources for graphics reproduced in the overview report. Whereas the Summary for Policy-makers report contains recommendations that focus mainly on policy-relevant actions concerned with addressing contaminant impacts on Arctic human populations, the conclusions and recommendations presented in this report also cover issues of a more scientific nature, such as proposals for filling gaps in knowledge, and recommendations relevant to future monitoring and research work.

This assessment of Arctic radioactivity issues was conducted between 2012 and 2014 by an international group of experts. AMAP Radioactivity expert group members and lead authors are appointed following an open nomination process coordinated by AMAP. A similar process was used to select international experts who independently reviewed this report.

Information contained in this report is fully referenced and based first and foremost on peer-reviewed and published results of research and monitoring undertaken since 2009. It also incorporates some new (unpublished) information from monitoring and research conducted according to well established and documented national and international standards and quality assurance/quality control protocols. Care has been taken to ensure that no critical probability statements are based on non-peer-reviewed materials.

Access to reliable and up-to-date information is essential for the development of science-based decision-making regarding ongoing changes in the Arctic and their global implications. Related assessment summary reports[i],[ii] have therefore been developed specifically for policy-makers, summarizing the main findings of the assessment. The assessment lead authors have confirmed that both this report and its derivative products accurately and fully reflect their scientific assessment. All AMAP assessment reports are freely available from the AMAP Secretariat and on the AMAP website: www.amap.no, and their use for educational purposes is encouraged.

AMAP would like to express its appreciation to all experts who have contributed their time, efforts and data, in particular the lead authors who coordinated the production of this report. Thanks are also due to the reviewers who contributed to the radioactivity assessment peer-review process and provided valuable comments that helped to ensure the quality of the report. A list of contributors is included in the acknowledgements at the start of this report and lead authors are identified at the start of each chapter. The acknowledgements list is not comprehensive. Specifically, it does not include the many national institutes, laboratories and organizations, and their staff, which have been involved in various countries in radioactivity-related monitoring and research. Apologies, and no lesser thanks are given to any individuals unintentionally omitted from the list.

The support from the Arctic countries and non-Arctic countries implementing research and monitoring in the Arctic is vital to the success of AMAP. The AMAP work is essentially based on ongoing activities within these countries, and the countries that provide the necessary support for most of the experts involved in the preparation of the AMAP assessments. In particular, AMAP would like to acknowledge Norway and the Russian Federation for taking the lead country role in this assessment and thank Canada, Norway, and the Nordic Council of Ministers for their financial support to the radioactivity assessment work.

The AMAP Working Group is pleased to present its assessment to the Arctic Council and the international science community.

William Standring (Assessment Co-lead, Norway)

Yuri Tsaturov (Assessment Co-lead, Russia)

Morten Olsen (AMAP Chair, April 2015)

Lars-Otto Reiersen (AMAP Executive Secretary)

Oslo, June 2016

[i] AMAP, 2015. Summary for Policy-makers: Arctic Pollution Issues 2015. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. 12 pp.

[ii] AMAP, 2015. Pollution Issues 2015: Overview report. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway.

1. Introduction

This report presents the outcome of the 2015 AMAP assessment of radioactivity in the Arctic. This is the fourth AMAP assessment of radioactivity in the Arctic, and builds on information presented in earlier assessments (AMAP 1998, 2004, 2010b). This report contains data concerning actual and potential sources of radioactive contamination in the Arctic that have become available since 2010, and provides updates in cases where new information has become available that either warrants revised assessment or relates to operations and sources that were not previously considered.

Earlier work by AMAP dating back to the mid-1990s has been instrumental in drawing attention to the risks posed by radionuclide sources in Arctic Russia. Since the last AMAP radioactivity assessment in 2009, progress has been made in addressing poorly stored nuclear waste, removing and decommissioning radioisotope thermoelectric generators (RTGs), dismantling nuclear submarines and handling their spent nuclear fuel, and cleaning up the temporary storage sites at Gremhika and Andreeva Bay. In this regard, the AMAP approach of actions based on scientific study and assessment can be seen as being very effective in reducing radiation risks in the Arctic region. But new potential sources of radioactivity have been identified. These include the planned decommissioning activities at a number of nuclear power plants in Europe and elsewhere, which may lead to temporary increases in radioactive discharges that could eventually reach the Arctic. New power plants are also planned in areas distant from the Arctic, and many older plants have been granted extensions to their operating licenses; accidents such as those at the Chernobyl and Fukushima Daiichi nuclear facilities have demonstrated that even accidents far from the Arctic have the potential to affect the Arctic region.

Chapter 2 presents an update of some of the main issues concerning sources of radioactivity in the Arctic raised in the previous AMAP assessment (AMAP 2010b); this includes a further consideration of how climate change could affect the remobilization of radioactivity within the Arctic. For some topics, the material is also considered in greater depth in later chapters of this report. More detailed information on radioactive waste handling and decommissioning in the Russian Federation is presented in Chapter 3.

Elevated levels of both naturally-occurring and anthropogenic sources of radioactivity found in the Arctic remain a concern. Monitoring the levels of radionuclides within the Arctic environment is a central part of the AMAP program. Chapter 4 reports the most recent results from national monitoring programs across the Arctic region. Figure 1.1 presents an overview of the environmental monitoring locations and some of the main sources of radioactivity addressed in this assessment.

Chapter 5 describes the catastrophic accident at the Fukushima Daiichi Nuclear Power Plant in 2011 and examines the potential for a future accidental release of radioactivity, from planned as well as existing sources, particularly within the Arctic. The accident at the Fukushima Daiichi nuclear power plant underlined the importance of environmental monitoring in the region. Long-term radioactivity monitoring carried out by the national monitoring networks that contribute to AMAP allow radioactive isotopes from Fukushima to be detected and tracked in the Arctic environment so that any associated impacts can be assessed. Such monitoring is invaluable in helping scientists understand the pathways for radioactive transport to, within and from the Arctic, and the long-term effects of radioactive contamination within different environments and foodwebs.

The development of extractive industries in the Arctic is expected to increase in coming years and will inevitably lead to increased pollution in the region. The waste streams produced in the extraction of hydrocarbons and minerals – including uranium mining – contain naturally-occurring radioactive substances found in bedrock. As climate change increases the accessibility of the Arctic, the likely increase in oil, gas and uranium extraction would lead to enhanced releases and mobilization of naturally-occurring radionuclides. Pollution risk through enhanced release of naturally-occurring radioactive material from increased exploitation of Arctic resources is discussed in Chapter 6.

Chapter 7 provides an overview of the conclusions of the assessment, which were reported to the Arctic Council at their Ministerial meeting in May 2015 (AMAP 2015).

Figure 1.1 Monitoring results and sources and potential sources of radioactivity discussed in this assessment.

2. Update on sources of radioactive contamination in the Arctic and possible effects of climate change

Authors: William Standring, Frits Steenhuisen, Louise Kiel Jensen, Edyta Łokas, Sven Poul Nielsen, Jing Chen, Per Roos, Xiaolin Hou, Jixin Qiao

Key findings

- *Levels of anthropogenic radioactivity measured in the Arctic and attributable to identified sources are very low and generally declining*
- *Specific sources have the potential to increase local contamination levels, such as radioactive waste dumped in the Barents and Kara seas and accidents at nuclear power plants, including those located far away from the Arctic*
- *Thawing permafrost and decreased snow cover in the Arctic due to global (and especially Arctic) warming could result in a widespread and substantially increased radiation dose to Arctic residents from radon gas and daughter products*

2.1 Introduction

This chapter presents an update of some of the main issues concerning sources of radioactivity in the Arctic raised in the last AMAP assessment (AMAP 2010b). For some topics, the material is considered in greater depth in later chapters of this report; more detailed information is available on radioactive waste and decommissioning (Chap. 3), monitoring of radioactivity (Chap. 4), the accident at the Fukushima Daiichi nuclear power plant and potential future accidents (Chap. 5) and naturally-occurring radioactive material, including technologically enhanced naturally-occurring radioactive material (Chap. 6).

2.2 Sources of artificial radionuclides in the Arctic

2.2.1 Global fallout

Global fallout is the term for artificial radionuclides found mainly in the northern hemisphere that can be attributed to the atmospheric nuclear weapons tests run from 1945 to 1995. These have been described in detail (Minatom 1996; UNSCEAR 2000) and comprise two periods of intensive testing: 1957–1958 and 1961–1962. Nuclear tests have been conducted by all nuclear powers. The main test sites were located in the Pacific (US/France), in Semipalatinsk and on Novaya Zemlya (Norris and Arkin 1996). Recent data based on different plutonium (Pu) isotope ratios in air filters collected daily during 1957–1963 and real time meteorological data show that fallout from the nuclear tests undertaken in Semipalatinsk in 1962 was transported directly to Scandinavia, via Russia, and then to the Arctic (Wendel et al. 2013a, 2015). Filters collected in periods with many detonations contained small-sized radioactive particles (Wendel 2013b).

The Novaya Zemlya test site is the only location within the Arctic where atmospheric tests were conducted (Sect. 2.2.1.1). Underground nuclear tests took place at the only other Arctic test site, Amchitka (Alaska) (Sect. 2.2.1.2). Worldwide, Norris and Arkin (1996) reported 2043 tests, of which 528 were atmospheric. Deposition from the atmospheric tests constitutes the largest contribution to the enhanced levels of artificial radionuclides still found today. As the tests were mainly undertaken in the northern hemisphere, this is where the highest activities are currently detected (Wright et al. 1999). Many radionuclides are produced during a nuclear detonation but it is the long-lived radionuclides such as ^{90}Sr and ^{137}Cs that are of particular interest when estimating the long-term effects of global fallout. Modelling studies show the greatest ^{137}Cs deposition occurred in Iceland, followed by Norway and Sweden, and that fallout occurred in all Arctic regions above 60°N (Wright et al. 1999). Because the amounts of radionuclides deposited per unit surface area and per unit volume of precipitation are relatively constant for any given latitude band, the amount of radionuclide deposition is directly proportional to the amount of precipitation (Bouville et al. 2002).

2.2.1.1 Nuclear weapon testing on Novaya Zemlya

From 1955 to 1990, 130 nuclear weapon tests (totaling about 265 megatons) were conducted at Novaya Zemlya in the atmosphere, underground, at sea or underwater. Regionally therefore, these tests are possible contributors to environmental activity concentrations of anthropogenic radionuclides. Atmospheric tests generally took place over the southern part of the northern island, while most of the underground tests were situated on the northern end of the southern island. Five nuclear weapon tests (three underwater, one above-water and one on the surface) took place in Chernaya Fjord on the southwestern coastline, and one above-water test was conducted further west in Bashmachnaya Fjord. Monitoring in Chernaya Fjord revealed ^{239,240}Pu sediment concentrations greater than 15000 Bq/kg as well as elevated levels of ^{137}Cs and ^{60}Co (Smith et al. 2000). An estimated ~11 TBq of ^{239,240}Pu is present within the sediments of Chernaya Fjord, while ^{240}Pu:^{239}Pu ratios in adjacent areas suggest that some of this Pu has been transported along the southern coastline of Novaya Zemlya (Smith et al. 2000), confirming observations from 1992–1994 (JNREG 1996).

2.2.1.2 Nuclear weapon testing on Amchitka

The only Arctic test site used by the United States was at Amchitka Island in the Aleutian chain. Three underground test explosions were conducted between 1965 and 1971; the Long Shot in 1965, the Milrow in 1969 and the Cannikin in 1971 (Kohlhoff 2002). With a reported yield of five megatons, Cannikin was the largest underground nuclear US test. With the exception of small concentrations of tritium detected in surface water shortly after the Long Shot test, radioactive fission

products from the tests remain in the subsurface at each test location, as documented by continued monitoring of the area (Burger et al. 2006a,b, 2007; Bu et al. 2013; US DoE 2013).

2.2.1.3 Peaceful Nuclear Explosions

From the 1960s to the end of the 1980s, several countries carried out 'peaceful nuclear explosions' (PNEs). Of the roughly 2050 nuclear devices detonated across the world between 1945 and 1996, over 150 were for peaceful purposes, mainly carried out by the former Soviet Union and the United States. Such events are essentially no different from weapons tests in terms of their potential for adverse effects on human health and the environment. The former Soviet Union undertook 124 PNEs between 1965 and 1988 (80 in Russia, 39 in Kazakhstan, and five in Ukraine, Uzbekistan and Turkmenistan) while the United States undertook 27 PNEs between 1961 and 1973 (four in Colorado and New Mexico, and 23 at the National Test Site in Nevada).

2.2.2 Thule

In January 1968, an American B-52 bomber crashed into the sea ice in the vicinity of Thule Air Base in Greenland (Fig. 2.1). As a result of the accident, radioactive material from the damaged nuclear weapons was dispersed in the fire and smoke from the burning engine fuel. Most of the radioactive material settled on the sea ice surrounding the crash site and was subsequently cleaned up by the United States. Smaller amounts of radioactive material were transported southward over Greenland by the wind. Since the accident, the Danish Risø National Laboratory (now Danish Technical University, DTU) has undertaken studies of the marine environment and recorded significant contamination on the seafloor beneath the crash site, with low levels of transfer to animals and marine plants. Indeed, relatively large mixed U/Pu particles are still present in sediments collected from the site (Lind et al. 2005). In 2003, terrestrial contamination was recorded at the coastal Narsaarsuk hunting station, 8 km south of the crash site. Studies of terrestrial contamination have been undertaken to assess the potential risk for humans of inhaling radioactive particles stirred up from the soil by wind and by dust-producing activities (Nielsen and Roos 2011).

The studies involved sampling soil and radioactivity measurements in the Thule area in summer 2007 and summer 2008. Besides Narsaarsuk, studies were conducted at Thule Air Base, Saunders Island, Wolstenholme Island, Kap Atholl (on the coast, 15 km south of Narsaarsuk, occasionally used for recreational activities), Grønnedal (a coastal hunting area 20 km south of Narsaarsuk), and Moriusaq (40 km northwest of Thule Air Base).

The Thule area is characterized by uneven terrain and changeable weather conditions. Motorboats and all-terrain vehicles were used for transport, while research activities were undertaken primarily on foot. A large number of soil samples were collected as well as air samples, rainwater and airborne particulates near Narsaarsuk, where contamination was previously recorded. Collection of air, rainwater and airborne particulate samples took place over periods of two to eight weeks in 2007 and 2008. Portable equipment was used to make additional measurements of soil contamination in places with raised contamination levels. Samples were transported to the DTU Risø Campus and analyzed for radioactive materials, including Pu-isotopes as well as americium. Americium-241 is the decay product of ^{241}Pu, which itself may have been an impurity in the bomb-plutonium.

Analyses of soil samples show great variation in radioactive contamination near Narsaarsuk. Results vary from background levels for plutonium (20–40 Bq/m^2) to more than 1 MBq/m^2. Elevated contamination levels occur sporadically on northward slopes that are moist in summer, presumably due to the deposition and subsequent melting of snow that was contaminated following the accident. Contamination occurs primarily in the uppermost soil layer (0–2 cm depth).

Figure 2.1 Thule area, showing the crash site and the hunting area around Narsaarsuk, Saunders Island, Wolstenholme Island, Grønnedal, Thule Air Base, and the Moriusaq settlement. The photo shows the clean-up operation.

The total amount of Pu in the soil near Narsaarsuk is estimated at 0.1 kg, compared with the estimated 4 kg on the seabed (Eriksson 2002). Earlier estimates of Pu on the seabed were based on partial leaching of Pu from U/Pu particles in the laboratory, and so are deemed underestimates. This is in relation to the 7–8 kg of Pu that the B-52 plane is estimated to have been carrying. The area around the Narsaarsuk hunting station was systematically surveyed for radioactive contamination, but it was not possible to undertake a similar systematic search across the region as a whole. As a result, the existence of other sites with raised levels of radioactive contamination, besides those that have already been identified, cannot be ruled out. A later study was undertaken in 2007–2008. Soil samples from Kap Atholl and Grønnedal had lower contamination levels than those near Narsaarsuk but activity concentrations were still significantly above the background level. Activities in soil samples from Thule Air Base, Moriusaq, Saunders Island, and Wolstenholme Island were all at background levels: there were no signs of contamination from the accident. Near Narsaarsuk, extremely small quantities of Pu were found in airborne particulates collected using air filters and sticky foils. These results showed air concentrations to be at the same low levels as found in Europe. Rainwater samples collected near Narsaarsuk also contained small quantities of Pu, also implying very low airborne levels. No samples of air or rain from Narsaarsuk showed signs of Pu concentration above expected ambient levels.

Dose assessments undertaken on the basis of conservative assumptions for three routes of radiation exposure (ingestion, inhalation, wound contamination) indicate that, even under extreme conditions and assumptions, the total dose for individuals in the Thule area resulting from the accident is significantly less than 1 mSv/y and therefore of no significance to health based on internationally recognized norms (NBH 2011).

2.2.3 Radioactive waste dumping in the Barents and Kara Seas

Since this topic was last covered by AMAP (AMAP 2004) a revised estimate of the amount of radioactive wastes dumped in the Barents and Kara Seas has become available (Sivintsev et al. 2005). Regular dumping of liquid and solid radioactive waste in the Arctic was practiced by the former Soviet Union and later by Russia from the early 1960s until the early 1990s. Dumping of radioactive wastes was also carried out by thirteen other countries in the Atlantic and Pacific oceans, although under strict international guidelines. The guidelines stipulated that there should be no dumping in coastal or fishing areas, that deep sea dumping was preferable, and that no high-level or liquid wastes should be dumped. Ocean disposal of radioactive waste has since been banned by international treaties (BASEL Convention 1989; London Convention 1972; MARPOL 1973). Assessments of the total activity of liquid and solid radioactive waste dumped into the Barents and Kara Seas were first reported by the Yablokov Commission (1993) in the 'White Book 1993', subsequently revised by the International Arctic Seas Assessment Project (IASAP) in 1993–1996 (Sjoeblom and Linsley 1998; IAEA 1999b) and then summarized by the International Atomic Energy Agency (IAEA) in a technical document (IAEA 1999a). Sivintsev et al. (2005) reassessed the information originally published in the White Book 1993 and identified a number of inaccuracies and omissions. This section gives an overview of the updated information available in the 'White book 2000' (Sivintsev et al. 2005). The total activity of liquid and solid radioactive waste dumped in the Barents and Kara Seas is reported to be just over 38800 TBq (Table 2.1), equivalent to about 45% of the total activity of radioactive waste dumped in the global oceans. This is a slight increase on the previous estimate reported by AMAP of 36600 TBq (AMAP 2004). However it is likely that the actual figure is even higher because Sivintsev et al. (2005) identified a number of hitherto undocumented dumping operations within the Barents and Kara Seas without being able to provide specific information on the activities of the waste dumped.

Table 2.2 presents a detailed breakdown of information for the liquid radioactive waste dumped at sea in the Arctic region. Liquid radioactive waste with a total activity of 522.6 TBq was deliberately dumped in the Barents Sea inside five designated areas (Fig. 2.2); an additional 435.3 TBq was inadvertently dumped through operational accidents in the Barents, Kara and White Seas (Sivintsev et al. 2005). Low- and intermediate-level solid radioactive waste was principally dumped at eight locations in the fjords east of Novaya Zemlya and the Novaya Zemlya trough in the open Kara Sea. By volume, the bulk of the solid radioactive waste comprised material produced during the routine operation of naval ships, icebreakers, and submarines containing nuclear reactors.

Low and intermediate-level solid radioactive waste dumped in the Kara Sea was generally sealed in metal containers. Leakage from dumped objects, especially containers was identified during the Joint Norwegian–Russian expeditions of 1992, 1993 and 1994 (JNREG 1996). Large waste objects were dumped separately or inside specially allocated ships, such as a barge, lighter or tanker. The total estimated activity of low and intermediate-level solid radioactive waste dumped in the Kara

Table 2.1 Total activity at the time of dumping for the various types of radioactive waste dumped at sea in the Arctic region by the former Soviet Union and Russia (Sivintsev et al. 2005).

Waste type	Total activity at time of dumping, TBq	Percentage of total activity
Reactor units with spent nuclear fuel	21780	56.1
Reactor units without spent nuclear fuel	14800	38.1
Reactor components	20.8	0.1
Low-level solid waste	1240	3.2
Low-level liquid waste	958	2.5
Total	38800	100

Table 2.2 Liquid radioactive waste dumped at sea in the Arctic region (Sivintsev et al. 2005). See Fig. 2.2 for dumping areas.

Location	Dumping area	Years	Total volume, m^3	Total activity, TBq		Comments
				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 icebreaker in lighter PSSN-328
Total			205190	957.8	348.9	

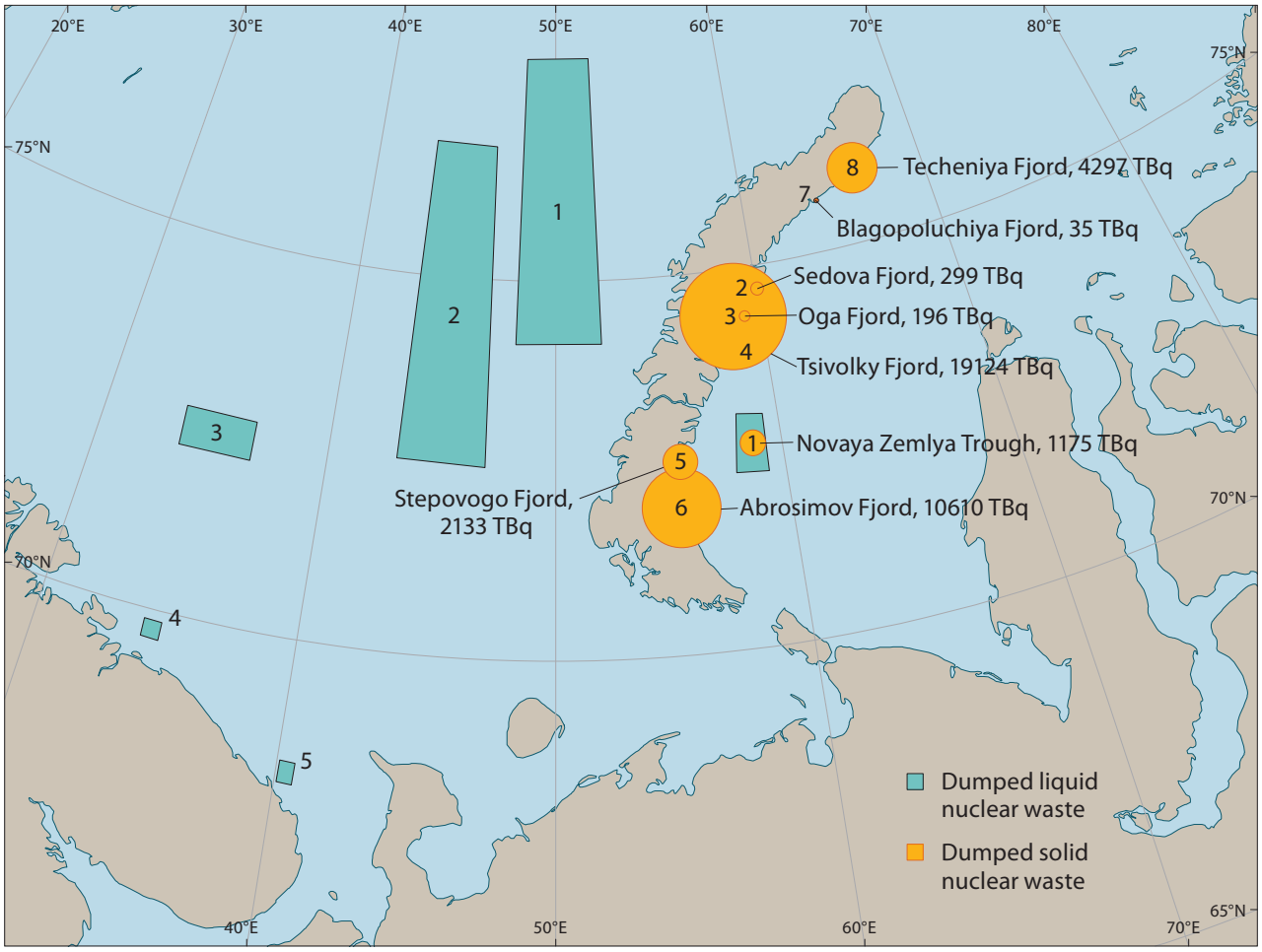


Figure 2.2 Main radioactive waste dumping areas in the Barents and Kara Seas and revised estimates (see Tables 2.3–2.5 for details) of the maximum total activity of the dumped solid waste at the time of dumping as reported in the White Book 2000 (Sivintsev et al. 2005).

Sea amounts to 1240.21 TBq (Table 2.3), with a further 11.1 TBq having been dumped in the Barents Sea (Sivintsev et al. 2005).

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 (Table 2.4). A number of reactor components have also 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. In total, five objects containing six reactors with SNF and a radiation shielding assembly with 60% of the fuel taken from the OK-150 unit in the icebreaker *Lenin* as well as five objects containing ten reactors without SNF have been dumped (Table 2.5) (Sivintsev et al. 2005). All reactors containing SNF were dumped with their reactor compartments filled with a furfural mixture and bitumen as a protective barrier against seawater ingression. According to design specifications, filling the reactor compartments with furfural should prevent SNF from coming into contact with seawater for a period of up to 500 years (Sivintsev et al. 2005).

Table 2.3 Low and intermediate-level solid radioactive waste dumped in the Barents and Kara Seas (Sivintsev et al. 2005). See Fig. 2.2 for dumping areas.

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 icebreaker *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 solid radioactive waste
Total			17105	991	1240.21	412.07	

Table 2.4 Total activity at the time of dumping for reactor units with and without spent nuclear fuel (SNF) dumped in the Kara Sea (Sivintsev et al. 2005).

Location	Year of dumping	Unit	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 shielding assembly not a reactor.

Table 2.5 Total activity at the time of dumping in reactor components dumped in the Barents and Kara Seas (Sivintsev et al. 2005).

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

2.2.4 Nuclear submarine *Komsomolets*

A fire broke out in the Russian nuclear submarine *Komsomolets* on 7 April 1989 while it was located in the Norwegian Sea. The crew managed to get the titanium-hulled submarine to the surface but could not maintain buoyancy; the submarine sank shortly afterwards to the seabed at 1700 m depth, 180 km southwest of Bear Island. Forty-two Russian seamen lost their lives in the accident; 25 crew members survived. *Komsomolets* sank containing a nuclear reactor and two nuclear-warhead torpedoes (in total 6.4 kg ^{239}Pu, Gladkov et al. 1994). None of the radioactive material onboard has since been recovered. Several

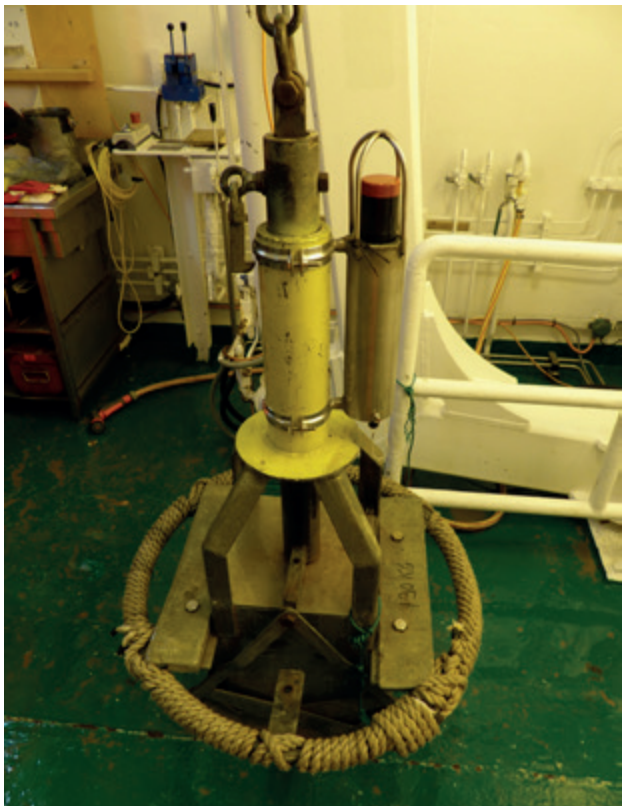


Figure 2.3 An acoustic transponder attached to the sediment grab allowed precise sediment sampling with regard to the position of the sunken *Komsomolets* submarine.

Russian expeditions monitored the *Komsomolets* wreck site in the early 1990s using manned submersible vessels. The submarine had been damaged when it hit the seabed, with large holes and cracks visible in its hull. The front section of the hull, where the torpedoes are located, was especially damaged. Expeditions completed in 1991 and 1992 reported that ^{137}Cs was leaking into the surrounding seawater through a ventilation pipe from the reactor section; although as reported by AMAP (1997) minimal contamination of the surrounding area had occurred.

Corrosion is expected to lead to more radioactive substances being released into the surrounding environment, with increasing amounts of the long-lived fission products, uranium and plutonium over the long term. However, predicting the course of these releases is inherently difficult as the hull, fuel rods and warheads will corrode at different rates. Estimates by Russian scientists predict that the reactor compartment will maintain integrity for roughly 2000 years. Nuclear warheads however do not have the same level of protection against corrosion and so are expected to release radioactive material earlier than the reactor. It is important to note however that separate assessments by Norwegian, Russian and NATO experts have concluded that the *Komsomolets* wreck presents only a small health risk to humans in the foreseeable future.

The Norwegian Institute for Marine Research has conducted regular annual monitoring of the wreck site area. They reported ^{137}Cs concentrations in seawater and sediments around the *Komsomolets* similar to background levels during the period 1993–2010 (NRPA 2012). Background levels for ^{137}Cs in sediments are generally 0–20 Bq/kg in the Norwegian Sea area while surface waters in the Barents Sea and the Norwegian Sea typically have ^{137}Cs levels of less than 3 Bq/m^3.

Sampling at 1700 m is difficult due to the research vessel at the sea surface drifting and water currents affecting the wires connecting sampling devices to the ship. Since 2013, a new position tool has been used during sampling campaigns. This instrument is attached both to water samplers and boxcorers. A Simrad MST342 wireless acoustic transponder was attached to the box-corer used for sediment sampling (Fig. 2.3). The transponder communicated with a dynamic positioning system on the research vessel (the Kongsberg HiPAP, High Precision Acoustic Positioning), while collecting the sediment samples. This method resolves difficulties previously encountered during sampling and ensured that the sediment samples were taken from the desired locations. Analyses of the sediment cores revealed that ^{137}Cs activity concentrations in all sediment layers collected north and west of the wreck were below the detection limit. ^{137}Cs activity concentrations in cores collected to the south and east of the wreck ranged from below the detection limit to 6.4±0.9 Bq/kg. Activity concentrations of ^{238}Pu, ^{239,240}Pu and ^{241}Am in the upper layers of the five cores were also low, ranging from below the detection limit to 0.05±0.01, from 0.02±0.01 to 0.9±0.1 and from 0.07±0.02 to 1.16±0.09 Bq/kg, respectively, while ^{238}U activity concentrations ranged from 8.3 to 22.1 mBq/g (5% SD). These levels of radionuclides are comparable to earlier studies in the area and to background levels observed in the Barents/Kara Seas indicating that no leakage of radioactivity had occurred from the *Komsomolets* to date (Heldal et al. 2014).

2.2.5 Sellafield, UK

Discharges from the nuclear fuel reprocessing activities at Sellafield Ltd can be transported by ocean currents through the North Sea and into Arctic areas. As reported in previous AMAP assessments, these discharges are an important contributor to the elevated radioactivity levels detected in the Arctic today. Discharges of liquid radioactive effluents from the site began in 1952, when a total of 370 TBq of radioactivity were discharged to the north-eastern Irish Sea (Howells 1966).

Discharges of most radionuclides released from Sellafield into the Irish Sea peaked in the mid-1970s with a ~180 TBq peak in total α-activity in 1973 and a ~9000 TBq peak in total β-activity in 1975. The amounts and composition of discharges have changed over time; quantities of shorter-lived fission product nuclides such as ^{95}Zr, ^{106}Ru and ^{144}Ce declined steadily from the early 1970s, longer-lived nuclides such as ^{137}Cs peaked in the mid- to late 1970s and declined thereafter, while discharges of the major transuranic radionuclides, ^{241}Am and ^{239,240}Pu peaked in the early- to mid-1970s. The composition of discharges has changed over time for several reasons, including the SNF type, its burn-up time, storage of the SNF prior to reprocessing, the reprocessing method used and the type of effluent treatment used. By 1992, discharges for all radionuclides were generally two orders of magnitude or more less than their peak value. However discharges of ^{99}Tc peaked in 1995 (190 TBq/y) before decreasing due to the implementation of a new rinse technology to remove ^{99}Tc from reprocessing effluents (AMAP 2010b). Figure 2.4 shows selected discharge patterns between 1995 and 2014. It is clear from the graphic that the discharges of total α-activity and total β-activity have remained relatively constant since the last AMAP assessment (AMAP 2010b).

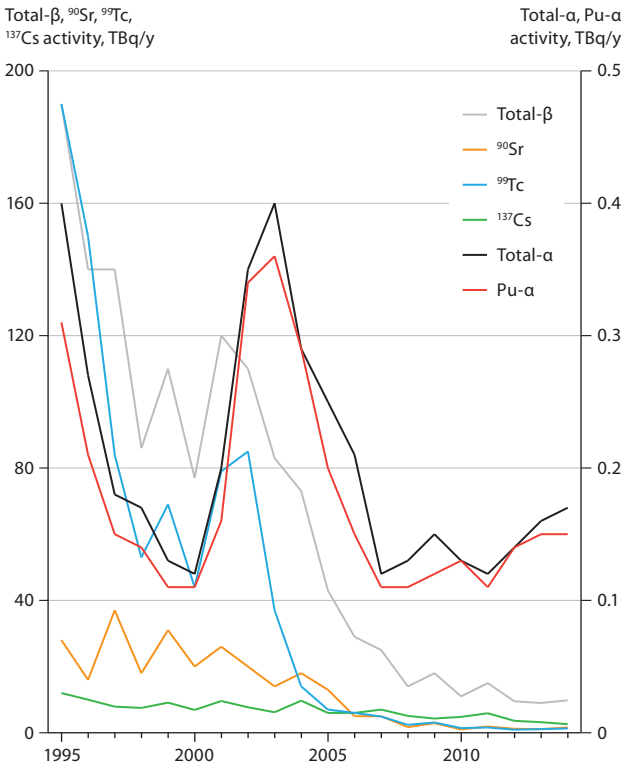


Figure 2.4 Annual discharges of selected radionuclides to the Irish Sea from the Sellafield nuclear fuel repossessing plant (OSPAR 2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014).

Owing to many years of discharges, the mud patch outside Sellafield is heavily contaminated by many radionuclides including Pu particles. As a result of remobilization, this contaminated mud patch will continue to act as a diffuse source of radionuclides to the surrounding seawater. It is expected that the mud patch will represent a source of Pu particles for at least the next 50 years (Mitchell et al. 1999).

2.2.5.1 Future decommissioning plans for Sellafield

Recognizing that Sellafield Ltd is a large and very complex nuclear site with considerable amounts of SNF and radioactive waste located there, it is clearly a significant potential source of future increases in radioactive contamination in the Arctic.

Highly Active Liquor

The UK Nuclear Installations Inspectorate (NII) has specified limits for permissible amounts stored on site of one of the most hazardous forms of waste present on-site: Highly Active Liquor (HAL) generated from reprocessing activities. The previous AMAP assessment report stated that 1225 m^3 of HAL were stored at Sellafield in 2007 (AMAP 2010b). Storage of such highly active, heat-generating waste in a liquid form is recognized as having an accident potential, and accidental releases of this material could possibly be transported to Arctic areas. The HAL is stored in Highly Active Storage Tanks (HASTs) located in the HAL Evaporation and Storage plant (HALES) at Sellafield. In 1990, the Waste Vitrification Plant (WVP) began converting the HAL into glass to retain the hazardous radioactivity in an immobile form enabling long-term passive storage of the waste. In January 2001, the NII introduced 'Specification 343' requiring the backlog of HAL stocks accumulated since reprocessing began in the 1950s to be reduced to a minimal working level, known as the buffer volume, by mid-2015. According to the previous AMAP assessment report, HAL stocks were to be no more than 625 m^3 by 14 April 2013 and down to the buffer level of 200 m^3 by 1 July 2015 (AMAP 2010b). Since then, limits imposed on the total volume (m^3) of HAL were deemed by the NII to unintentionally discourage Sellafield Ltd from carrying out basic operations such as HAL tank wash-out and clean-out of near-empty tanks after HAL had been removed (the diluted HAL residue still being categorized as part of the total volume). In October 2007, 'Specification 679' changed the units applied to 'Oxide HAL' limits, from volume (m^3) to the mass of Uranium (tonnes equivalent Uranium, or teU) in the unprocessed fuel from which the HAL was derived. HAL stocks were approximately 20,000 teU in 2001 and have now been reduced to the intended buffer level (5500 teU) (pers. comm. Sellafield Ltd). HAL stocks are now considerably lower than at the time of the last AMAP assessment and are at their lowest level since the 1980s (ONR[1] 2015).

B205 Magnox reprocessing facility

Two nuclear fuel reprocessing plants exist at the Sellafield site. The first is the Thermal Oxide Reprocessing Plant (Thorp) which was designed to reprocess SNF from Advanced Gas-cooled Reactors (AGR) and Light Water Reactors (LWR). The other reprocessing plant reprocesses SNF from Britain's early nuclear reactors ('magnox' fuel). Magnox fuel assemblies were designed with characteristic cooling fins that allowed maximum heat transfer despite low operating temperatures. The name magnox came from the alloy used for the cooling fins and cladding around the fuel rods which mostly comprised magnesium and aluminum (as well as some other metals). Magnox was the acronym for 'Magnesium non-oxidising' and the alloy was deemed advantageous as it gave a low neutron capture cross-section. However, use of the alloy as fuel cladding placed a limit on the maximum operating temperature of the power plant, thereby reducing its thermal efficiency. In addition, magnox alloy reacts with water, making long-term storage of SNF under water problematic and requiring the SNF to be reprocessed relatively quickly after its removal from the reactor core. This shorter period from removal to reprocessing increases the fission product hazard compared to other types of SNF, and requires costly remote handling facilities at the reprocessing plant. Before reprocessing can take place, magnox SNF must be stored for at least 180 days in specially designed ponds such that the short-lived fission products present in the SNF can decay away. The SNF is then transported to Sellafield where the cladding is later removed. The magnox reprocessing plant at Sellafield began operations in 1964. After arrival at the magnox plant, the SNF is dissolved in nitric acid, before a series of solvent extraction processes separate out the uranium, plutonium and fission products. The three product streams are then converted elsewhere at Sellafield into solid uranium trioxide, solid plutonium oxide, and nitric acid liquor (i.e. the raffinate, containing the waste fission products). The latter is then stored in the HASTs destined

[1] The Nuclear Installations Inspectorate was the nuclear regulatory body in place up to 1 April 2011, when it became the Office for Nuclear Regulation.

for vitrification. The magnox plant is a very important part in the Nuclear Decommissioning Authority's risk and hazard reduction operations, reprocessing SNF from operating and closed magnox power stations around the UK. The original closure date for the facility was predicted as 2012. The newly scheduled closure date of December 2019 is currently in doubt due to the plant shutdown on 23 February 2014 for seven weeks caused by a 'system blockage' (Sellafield Ltd info). This setback and others have led to the plant not meeting annual targets for reprocessing for nine consecutive years. Despite an annual reprocessing target for the plant of 664 t for 2013/2014, the facility had reprocessed only 352 t by the end of January 2014, just weeks before having to shut down due to the 'system blockage'.

Annual reprocessing targets are presented in the Sellafield Plan (Sellafield Ltd 2011). According to the plan, 2159 t of magnox fuel should have been reprocessed by the facility during the three-year period 2011–2014. However, only 1338 t of SNF were actually reprocessed over this period. An average annual reprocessing rate of about 440 t (representative of the last nine years of operation at the plant) and an outstanding stock of 3125 t magnox fuel still to be reprocessed implies that the reprocessing work will need to continue until at least 2022.

This means that discharges from the magnox reprocessing operations at Sellafield, acknowledged as the largest source of radioactive discharge to the marine environment from the facility and one that could potentially affect Arctic ecosystems, will continue for many years. In general, however, discharges from Sellafield have greatly decreased since their peak in the 1970s and this is reflected in recent monitoring results from the Irish and North Sea, as well as the Kara and Barents Seas (NRPA 2012).

2.2.6 Cap de la Hague, France

Annual updates on liquid discharges from nuclear installations to the North-East Atlantic are available from the OSPAR Commission (OSPAR 2014). The reprocessing plant at Cap de la Hague in France reprocesses fuel from pressurized water reactors (PWR) and boiling water reactors (BWR). In 2012, this facility was the third largest contributor to discharges of total β-activity in the OSPAR Convention area accounting for 14% of the total. Annual discharges have progressively decreased and are now less than a third of those in 2005 (Fig. 2.5). Discharges of ^{90}Sr, ^{137}Cs and ^{99}Tc remain low (OSPAR 2014).

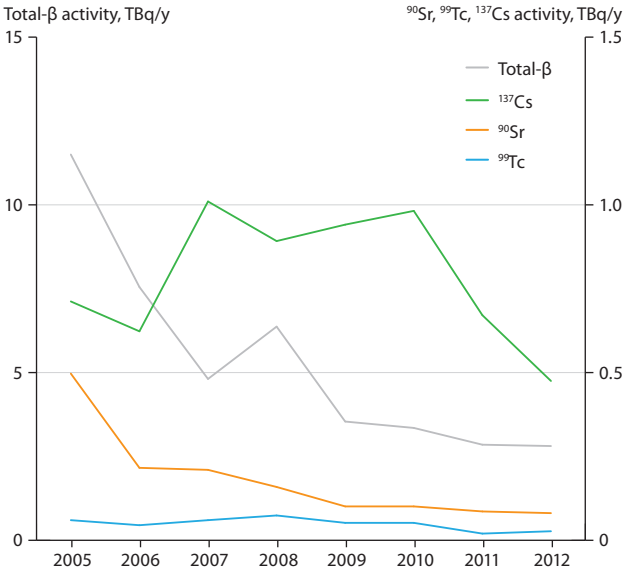


Figure 2.5 Annual marine discharges from Cap de la Hague repossessing plant of selected radionuclides (OSPAR 2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014).

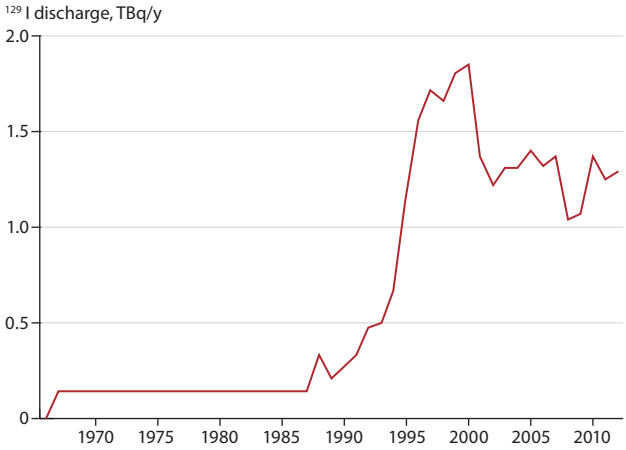


Figure 2.6 Annual discharge of ^{129}I from the nuclear fuel reprocessing plant at Cap de la Hague to the Channel (OSPAR 2007, 2008, 2009, 2010, 2011, 2012, 2013, 2014).

Discharges of ^{129}I from the reprocessing plant at Cap de la Hague increased from <0.5 TBq/y in the early 1990s to a peak of 1.83 TBq/y in 1999, and then declined to their current level of 1 to 1.5 TBq/y (Fig. 2.6). Owing to the extremely long half-life of ^{129}I (15.7 million years) these discharges result in a negligible dose.

2.2.7 Ob and Yenisey Rivers: Possible influence of nuclear sites on the Kara Sea

The Arctic Ocean is strongly influenced by river discharges. Freshwater inflows from the major northern rivers are roughly 3300 km^3/y, which is equivalent to 10% of total global runoff. According to McClelland et al. (2004), the Yenisey, Ob and Lena rivers are major contributors of freshwater to the Arctic Ocean with discharges of 620, 404 and 530 km^3/y, respectively. The Ob and Yenisey rivers, in particular, are recognized as potential sources of anthropogenic radioactive contamination to the Kara Sea due to the presence of three major nuclear sites within their catchments (see Fig. 2.7 in Sect. 2.2.8).

The Mayak Production Association (Mayak PA) is located within the Ob river drainage basin. Here, five nuclear reactors have been used for the production of Pu, with two reprocessing facilities on site. Of the original seven military reactors, five uranium-graphite reactors were shut down between 1987 and 1991 and the remaining two produce radionuclides for military and civilian use (JNREG 1997). Direct discharges of radionuclides were made to the Techa River system, which eventually flows into the Ob, between 1949 and 1956. This caused severe contamination along the entire length of the River Techa (Trapeznikov et al. 1993; Bradley and Jenquin 1995). In 1957, a tank holding high level radioactive waste exploded spreading approximately 74 PBq of radionuclides over an area to the north-northeast of the site. Artificial reservoirs have been used to store liquid wastes, reducing the amounts of radionuclides entering the River Techa (Strand et al. 1999). The upper Techa sediments, notably in the Asanov Swamp, still contain artificial radionuclides from previous direct discharges

(Romanov 1995). Asanov Swamp sediments also contain trace amounts of weapons-grade plutonium (Børretzen et al. 2005).

The Siberian Chemical Combine (SCC) facility, on the east bank of the River Tom that drains into the River Ob, began operation in the early 1950s to produce and reprocess Pu, and store radioactive waste. Two of its five reactors were still operating in 1998 (Strand et al. 1998). Large amounts of solid and liquid radioactive waste have accumulated at the SCC (Strand et al. 1998). Liquid waste is stored in stainless steel tanks and deep-well injected on site (Balonov et al. 1997). On 6 April 1993, an explosion occurred in a storage tank at the SCC reprocessing plant and resulted in localized contamination of the environment. Surface basins have also been utilized for intermediate storage of radioactive wastes (Waters et al. 1999), although these basins are no longer in active service. Relatively little information is available about surface contamination around the SCC, although the site is reported as having several surface ponds (Bradley 1997). One reactor in operation from 1955 to 1990 had an open once-through cooling system, similar to that in operation at the Mining and Chemical Industrial Combine at Zheleznogorsk, and discharged into a canal flowing into a branch of the River Tom. Radioactive discharges into the River Tom are assumed to have contained radionuclides from cladding corrosion (e.g. ^{63}Ni, ^{55}Fe, ^{60}Co and ^{65}Zn) as well as fuel rods (e.g. ^{90}Sr, ^{137}Cs and Pu isotopes), and discharges will have been substantially reduced after the open circuit reactor was shut down (Bradley 1997).

Radioactive contamination of the Yenisey River and its floodplain is largely due to artificial radionuclides released in discharge waters from the Mining and Chemical Industrial Combine, Zheleznogorsk (MCIC), where Pu was produced and reprocessed and radioactive wastes are stored (Linnik et al. 2006). The MCIC is located on the east bank of the Yenisey River about 2500 km upstream from the Kara Sea. Three RBMK-type graphite moderated reactors have been in operation at the complex. The two oldest reactors (decommissioned in 1992) had open-core cooling systems that pumped Yenisey water in through the reactor core and directly back into the river without any radiochemical treatment (Bolsunovsky et al. 1999). A substantial decrease in radionuclide concentrations close to the facility was observed after operation of these direct flow reactors ceased (Nosov 1996). The third reactor has a closed cooling system and is still in operation. The large amounts of radioactive waste generated by MCIC operations are stored on-site in stainless steel tanks, in one of four reservoirs (Waters et al. 1999) or have been injected into the ground, within the site boundary. Radioactive contamination of the river floodplain downstream of MCIC is extremely heterogeneous with maximum surface contamination levels occurring along shorelines (Nosov 1996). A study by Kuznetsov et al. (1995), following the decommissioning of the direct flow reactors, showed that river water concentrations of relatively short-lived radionuclides (^{51}Cr and ^{46}Sc) decreased to negligible levels (undetectable) within 300 km of the site. Long-lived radionuclide contamination (e.g. ^{239,240}Pu and ^{137}Cs) also decreased with increasing distance downstream, but at a slower rate.

Several studies have observed the influence of these three nuclear facilities on contamination levels downstream due to regulated and accidental discharges. In general this influence becomes markedly less apparent with continuing distance downstream from the nuclear facility. For example, contamination levels of 5800 kBq/m^2, as of 1973, were reported in riverine sediment deposits in close proximity to the MCIC site, more than 50% of this activity being attributable to the short-lived (<1 year half-life) radionuclides ^{51}Cr and ^{65}Zn (Vakulovsky et al. 1995). Areas of the Yenisey River subject to periodic flooding have also exhibited significant levels of anthropogenic radioactive contamination (e.g. Kuznetsov et al. 1995) and radioactivity in deposits on Yenisey floodplains has previously been recorded at distances up to 800 km downstream of the MCIC (Kuznetsov et al. 1995; Nosov 1996). The spatial depositional characteristics observed were that inventories of anthropogenic radionuclides tended to decrease with distance from the MCIC with the exception of the Angara River confluence where there is deposition of suspended matter during periods of low flow levels in the river. Nosov et al. (1993) observed that ^{60}Co, ^{152}Eu and ^{154}Eu were the dominant anthropogenic contaminant radionuclides in the surficial layers of floodplain deposits located up to 25 km downstream from the MCIC, whereas ^{137}Cs constituted the dominant component of anthropogenic contamination in floodplain deposits occurring further downstream. Other studies have also documented that elevated concentrations of ^{137}Cs in floodplain deposits are often at depth in the sediment profile, overlain by layers of relatively uncontaminated alluvial material (Nosov et al. 1993; Krapivin et al. 1998; Bolsunovsky et al. 1999; Klemt et al. 2002), indicative of past MCIC discharges that have been buried after the older reactors were shut down.

Yenisey floodplain deposit samples from cores studied by Standring et al. (2009) exhibited variable grain-size compositions ranging from sands to silts and sandy silts that reflect the core location within different sedimentation zones on the flood plain and the changing flooding regime. ^{239,240}Pu activity concentrations were in the range <0.01–14.2 Bq/kg dw with most cores exhibiting elevated concentrations of Pu in lower layers, again indicative of contamination occurring during the earlier years of the MCIC operations. A distinct Pu signal, discernible from global fallout, was observed in floodplain deposits in the MCIC near zone and up to the confluence with the River Angara. ^{239,240}Pu activity concentrations measured in Yenisey sediments were correlated with the silt fraction (percentage by mass <63 μm). ^{137}Cs activity concentrations were in the range 23–3770 Bq/kg dw. ^{137}Cs contamination densities appeared to be 3–6 times lower further downstream from MCIC compared to sites within 20 km of the Combine (Standring et al. 2009). Many cores exhibited sub-surface maxima, again indicative of historical discharges from the MCIC, further corroborated by ^{238}Pu:^{239,240}Pu activity ratio data.

Cesium-137 activity concentrations were also correlated with the fraction of silt present in the cores (Standring et al. 2009). No significant correlation was observed between grain-size normalized ^{137}Cs activity concentrations and distance downstream from the MCIC, confirming the possibility of ^{137}Cs transport down the Yenisey river in both soluble and suspended forms. Compared to sites downstream from the Mayak PA, Southern Ural, a site with a similar history of reprocessing activities as the MCIC, the radiological impact from activities at MCIC on the Yenisey river flood plain appear to be lower.

A recent study of the Tom and Ob rivers downstream of the Tomsk SCC (Nikitin et al. 2012) reported observations of radionuclide levels measured after the last SCC reactor was shut down on 5 June 2008. Nikitin and co-workers stated that concentrations of ^{90}Sr, ^{137}Cs, ^{239,240}Pu and other artificial radionuclides in water, bottom sediments and flood-plain soils of the Tom and Ob rivers (from Tomsk along the Tom River to its confluence with the Ob river, and 60 km downstream from the confluence) measured after shutdown of the last SCC single-pass reactor were of no radiological significance and that current activity concentrations of long-lived artificial radionuclides ^{3}H, ^{90}Sr, ^{137}Cs and ^{239,240}Pu in river water were below the intervention levels established by relevant Russian Federation regulations. The results of ^{3}H analysis in water from the Tom and Samuska rivers demonstrated no inflow of contaminated formation water to surface water from the sites where liquid radioactive wastes of the SCC were injected below the surface (Nikitin et al. 2012). However, ^{137}Cs contamination levels in flood-plain soils influenced by SCC liquid discharges were higher than the regional background and localized flood-plain contamination with ^{137}Cs and other γ-activity emitters (such as ^{60}Co and ^{152}Eu) was recorded.

Looking at the far field and estuary zones, Standring et al. (2008) concluded that the distribution of ^{137}Cs in the sediments sampled from the Ob and Yenisey estuaries and Kara Sea indicated a lithological influence, the highest levels being associated with sediments rich in clay materials. Such an influence was not observed for ^{90}Sr. Pu isotope levels measured were higher within the estuaries than for the Kara Sea sediments. ^{137}Cs, ^{90}Sr and Pu isotopes levels reported by Standring et al. (2008) did not clearly indicate an influence on radionuclide levels in the sediments from nuclear facilities that discharge or have discharged to the Ob and Yenisey river systems. However, the presence of ^{60}Co at depth in some of the sediments observed by Standring et al. (2008) provides evidence of inputs from such nuclear facilities and the results indicated a continued input of this isotope, probably from erosion of contaminated soils and sediments along the Yenisey River. To further corroborate the hypothesis that catchment-derived contaminants can be transported to and detected within estuaries, other studies using inter alia ^{239}Pu:^{240}Pu ratios concluded that evidence for non-global fallout sources of contamination is present in these areas (e.g. Cochran et al. 2000; Kenna and Sayles 2002; Skipperud et al. 2004).

Nevertheless, it should be noted that levels of radionuclides in the Ob and Yenisey estuaries are very low and although the nuclear facilities further upstream do represent a potential source of radioactivity to the Arctic, to date this has not occurred.

2.2.8 NPPs that can affect the Arctic

As of early 2015, there were some 440 nuclear power plants (NPPs) operating in 31 countries plus Taiwan, with a combined capacity of over 381 GWe (Fig. 2.7). In 2011 these provided

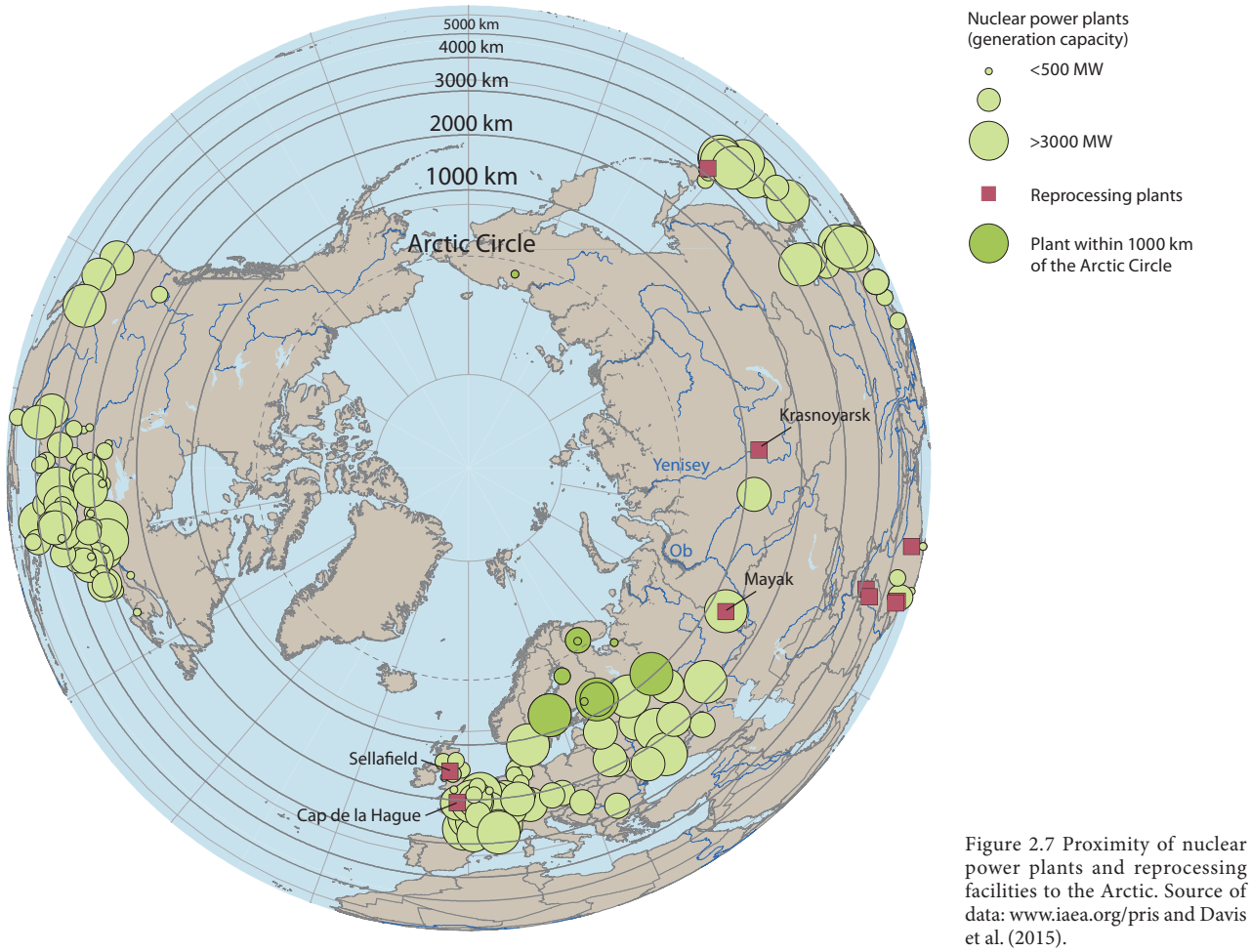


Figure 2.7 Proximity of nuclear power plants and reprocessing facilities to the Arctic. Source of data: www.iaea.org/pris and Davis et al. (2015).

2518 billion kWh, about 13.5% of the world's electricity. Over 60 NPPs are currently being constructed in 13 countries, the largest worldwide being constructed in Finland and France. In general, new reactor designs have more safety features (e.g. containment barriers) compared to older generation reactors. Although it might be assumed that effects on Arctic ecosystems from potential large-scale accidents at NPPs would be greater with closer proximity to the Arctic, as the accident at the Fukushima Daiichi NPP in Japan has shown the potential for measurable contamination in Arctic areas certainly exists following accidents at sites far outside the Arctic Circle.

The International Energy Agency (IEA) publishes annual forecasts and scenarios for NPPs worldwide. Following the Fukushima Daiichi NPP accident, the World Energy Outlook 2011 New Policies scenario predicted a 60% increase in nuclear capacity up to 2035, compared with about 90% predicted in 2010 (www.iea.org). Increased nuclear capacity in some countries will be in the form of (planned) upgrading of existing plants.

2.2.8.1 Nuclear plant construction

Most reactors currently planned are in the Asian region, where economies have experienced considerable growth in recent years coupled with rising electricity demand. Nevertheless, many countries with existing nuclear power programs have also stated plans to build new NPPs (in addition to those already under construction). This includes Argentina, Armenia, Brazil, Bulgaria, Canada, China, Czech Republic, France, India, Pakistan, Romania, Russia, Slovakia, South Korea, South Africa, Ukraine, UK, and the USA. As of March 2013, about 160 power reactors are planned worldwide and over 320 more are proposed.

The USA has plans for 13 new reactors; two combined construction and operating licenses were issued in early 2012, five more are under review. The new reactors are planned to be on line by 2020. Canada plans to add up to 2200 MWe of new capacity at Darlington in Ontario. Russia currently has ten reactors under construction, to be operational around 2017, with plans for 14 further reactors. Finland is building its fifth NPP with two more under planning. France is constructing a 1600 MWe unit at Flamanville, scheduled to come online in 2016, while the UK is planning four similar units ready for operation by 2019. Romania began operating its second NPP in 2007, and plans two further NPPs by 2017. Slovakia is constructing two new 470 MWe units at Mochovce with a planned start-up date of 2015. Bulgaria has plans for a new NPP at Kozloduy, while Belarus is planning two large new Russian reactors at Ostrovets, the first to start in 2019. Poland is planning two 3000 MWe NPPs, although start–up dates are not before 2025. Turkey has contracts signed for four 1200 MWe Russian NPPs to be built at Akkuyu and is proposing to double its nuclear capacity in the future. South Korea has planned four reactors to be operational by 2017, with an additional five by 2021. China has 15 operating reactors, with 26 under construction. India has 21 reactors in operation, and six under construction, including two large Russian reactors and a large prototype fast-breeder reactor aimed at developing a fuel cycle using thorium. The third and fourth NPP in Pakistan are being constructed at Chashma.

2.2.8.2 Nuclear plant extensions and retirements

Most NPPs were originally designed to operate for about 30–40 years. However safety and engineering assessments have led to extensions being granted for operating licenses. In the USA alone, more than 70 reactors have received license extensions of their operating lives to 60 years. The Russian government has extended the operational life of most of the country's reactors from 30 years up to 55 years, albeit with significant upgrade requirements. Replacing major reactor components has been demonstrated successfully and is attractive for operators given current difficulties obtaining public acceptance for new-builds. It should not therefore be assumed that NPPs will close when their operating licenses expire. However, during the period 1996–2012, the approximate balance was maintained with 60 reactors retired while 66 started operation. Before the Fukushima Daiichi NPP accident, the case for new-builds seemed to be strengthening, but since the accident there has been a growing negative reaction to nuclear power and the likelihood is that this will be reflected in the number of new NPPs coming online in the near future.

2.3 Changes in radiation dose due to climate change impact

The third assessment by the Intergovernmental Panel on Climate Change (IPCC 2001) included the first major scientific assessment of Arctic climate change. The second major assessment concerning climate change focusing specifically on the Arctic was the Arctic Climate Impact Assessment (ACIA 2005). This was commissioned by the Arctic Council and conducted by two of its working groups (AMAP and CAFF, Conservation of Arctic Flora and Fauna), in association with the International Arctic Science Committee (IASC). The main findings of this assessment (ACIA 2004) supported those of the earlier IPCC assessment (IPCC 2001) and detailed the likely impacts of Arctic climate change on Arctic peoples and the economic and societal impacts in general.

Another assessment related specifically to Arctic climate change was that concerning the influence of global change on contaminant pathways to, within, and from the Arctic (Macdonald et al. 2003). A brief section described likely changes regarding transport of radioactivity to the Arctic under different climate change scenarios. The report focused on the physical transport of contaminants to the region rather than how different radioecological processes within the Arctic may alter as a result of climate change. The conclusion was that pathways and amounts of anthropogenic radioactive contamination entering the Arctic were not expected to alter dramatically under future climate change. However, the report did state that potentially the most significant increase in radiation exposure to Arctic residents may arise from enhanced release of ^{222}Rn from soil (with a resulting increase in environmental levels of ^{210}Pb/^{210}Po; see Fig. 2.8) due to thawing of permafrost and decreased snow cover. ^{210}Pb and ^{210}Po constitute the greatest internal radioactive dose contributor to Arctic residents and biota and this dose tends to be higher than for inhabitants of temperate regions due, among others, to the nature of Arctic food chains and dietary factors. The primary source of ^{210}Pb

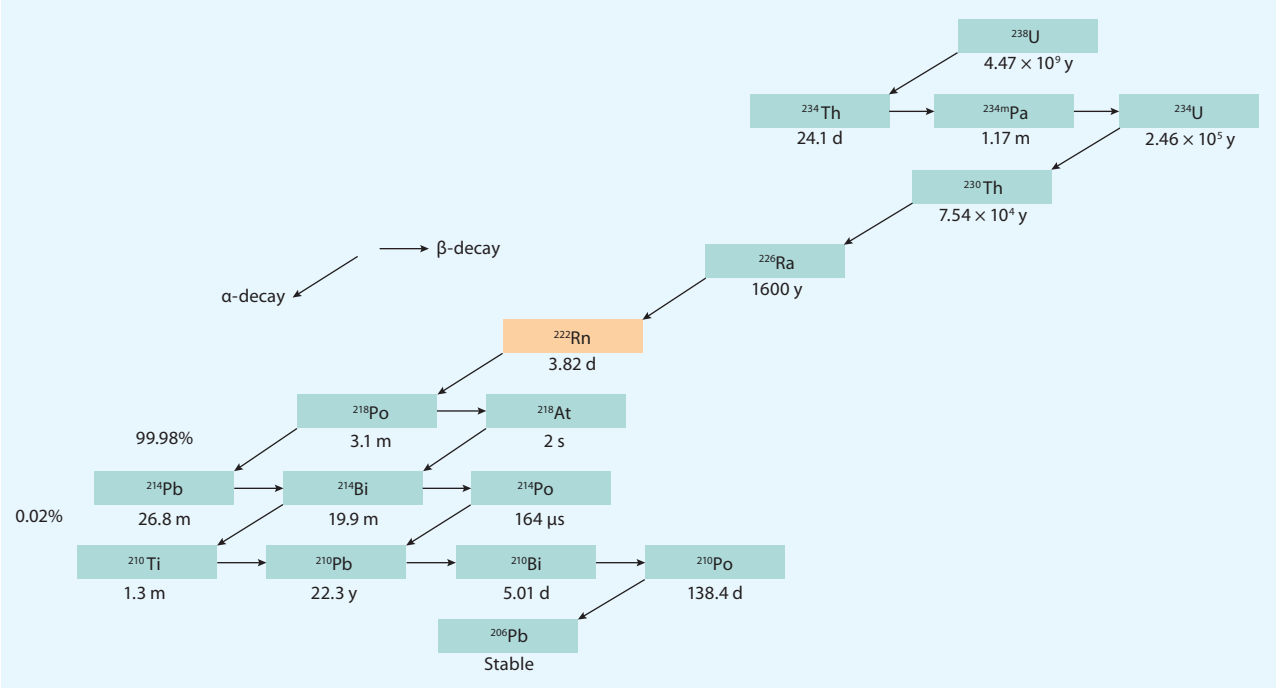


Figure 2.8 ^{222}Rn belongs to the radium and uranium decay series and has a half-life of 3.8 days. Its decay is predominantly through the sequence shown above. The short-lived radon progenies are recognized as being responsible for most of radon's biological effects.

to the Arctic is long-range transport via air masses that have passed over continental land masses. The presence of snow affects the local availability of radon daughters via the build-up of daughters within the snow cover as exhaled radon from the underlying soil decays within the snow pack. Pourchet et al. (2000) demonstrated the importance of snow in producing apparent flux densities of ^{210}Pb some 80 times greater than could be accounted for by normal deposition at a site in the French Alps and it is likely that such processes play a role in the Arctic environment. This role may increase in significance as soil is frozen for shorter periods of the year and snow depths increase; in contrast to the current situation where radon is confined within the frozen soil for long periods. The lack of research effort in differentiating the relative importance of local and long-range inputs of radon daughter isotopes to the Arctic environment and elucidating the role of Arctic-specific processes with respect to local inputs, increases the difficulty of determining the effect of a changing climate on potential levels of ^{210}Pb/^{210}Po in Arctic ecosystems. Although it is relatively simple to determine the effect of any one parameter (such as soil moisture, particle size, etc.) on potential changes in radon levels in soil or the release of radon to the atmosphere, determining the combined effects of changes in a range of parameters is more complicated. Nevertheless, given what is known about the influence of individual parameters on the behavior of radon in the environment, it is reasonable to assume that Arctic climate change will affect the levels of radon gas and daughter products (AMAP 2010b). This reflects one of the conclusions of Macdonald et al. (2003): "*any substantive increase in ^{222}Rn evasion due to warming/permafrost melting would have a widespread and substantial (doubling or tripling) effect on the radiation dose*". This conclusion with respect to the potential effect of Arctic warming on the radiation dose to Arctic residents from ^{222}Rn and its daughters is still highly relevant.

2.3.1 Enhanced radon emissions

Climate change could potentially mobilize radionuclides in the Arctic terrestrial environment. This may also affect radon emission from the ground, which is a major contributor to human exposure.

A gradual warming of Arctic areas leads to enhanced emissions of local radon, both ^{220}Rn and ^{222}Rn, to the atmosphere. ^{212}Pb is a decay product of thoron (^{220}Rn, half-life of 56 seconds). Due to the very short half-life of thoron, ^{212}Pb in the Arctic air is almost exclusively of local origin. With a half-life of 10.64 hours, ^{212}Pb can diffuse some distance from the soil through the atmosphere. ^{222}Rn decays to ^{210}Pb. Because it has a long half-life (22.3 years), ^{210}Pb can be transported over vast distances and remain in the atmosphere for extended periods. Surface air ^{210}Pb concentration in the Arctic can be of both local origin and southern latitudes far outside the Arctic. The Yellowknife (62.48°N, 114.47°W) airborne particulate monitoring station in the Canadian Low Arctic has simultaneously recorded ^{210}Pb and ^{212}Pb for decades. Figure 2.9 presents annual, summer and winter average ^{212}Pb activity concentrations in surface air for the past 11 years and shows a consistent local increase in radon, particularly in summer (Chen et al. 2014; see also Chen 2014 and Zhang et al. 2015). The graphic also presents annual, summer and winter average ^{210}Pb concentrations in surface air at the Yellowknife station. A slight increase in ^{210}Pb concentration is evident during summer while a slight decrease in ^{210}Pb is apparent during winter. This decrease may be caused by the global warming and its effect on long-range atmospheric transport. The annual averages show little change over the past 11 years (Chen et al. 2014; see also Chen 2014 and Zhang et al. 2015).

The last AMAP assessment of radioactivity in the Arctic (AMAP 2010b) included a comprehensive review of the possible effects of climate change on all aspects of radioactivity in the Arctic,

with almost all of this information still relevant in 2015. In its latest assessment the IPCC stated that warming of the climate system is unequivocal; the atmosphere and ocean have warmed, the amounts of snow and ice and Arctic sea ice have diminished, sea levels have risen, and the concentrations of greenhouse gases have increased (IPCC 2013). The assessment also states that over the 21st century the IPCC expects Arctic sea-ice cover to continue shrinking and become thinner, spring snow cover in the northern hemisphere to decrease as global mean surface temperature rises, glacier volumes to decrease globally, and global mean sea level to continue to rise. Essentially, the changes and effects of climate change with regard to radioactivity in the Arctic described by AMAP in its latest assessment (AMAP 2010b) still reflect the current state of knowledge and so do not need updating. However, the rate of climate change may accelerate the future timing of effects as described extensively in a recent AMAP assessment of climate-driven changes in Arctic snow, ice and permafrost (AMAP 2011).

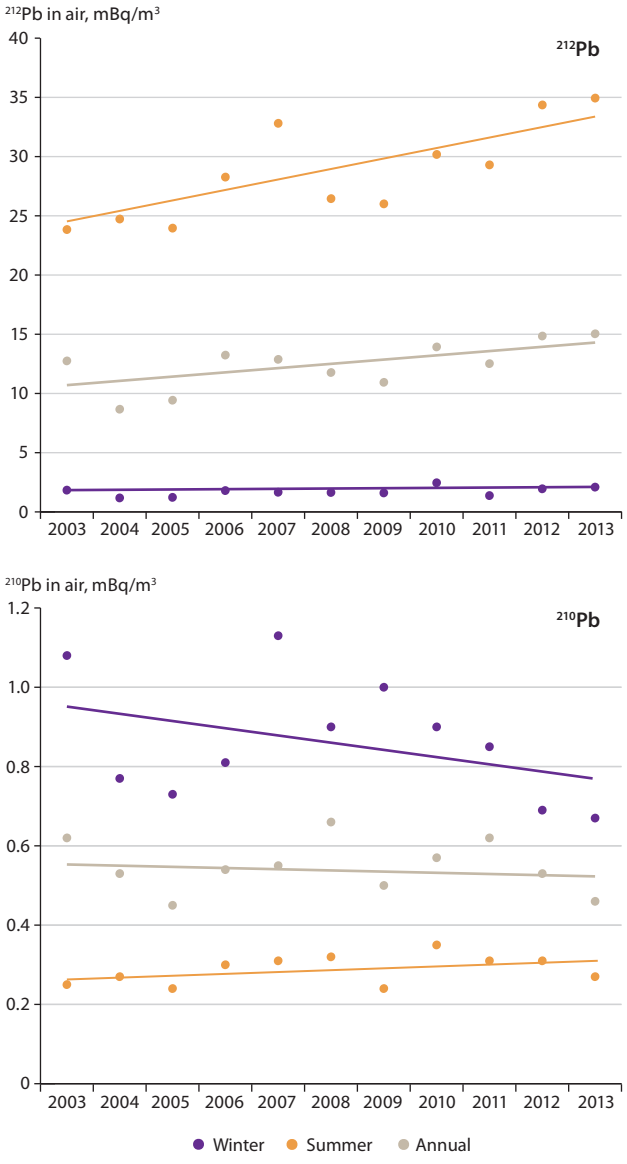


Figure 2.9 Winter, summer and annual averages of ^{212}Pb and ^{210}Pb activity concentrations in surface air at the Yellowknife airborne particulate monitoring station from 2003 to 2013 (Chen et al. 2014).

2.3.2 Climate change and its impact on radionuclide mobility

This section uses the example of glacier retreat and its impact on radionuclide mobility on Spitsbergen to illustrate how climate change can affect radioactive contamination within the Arctic environment. Łokas et al. (2013a,b, 2014) measured levels of the anthropogenic radionuclides ^{238}Pu, $^{239+240}$Pu, ^{241}Am and ^{137}Cs in peat and soil profiles from Spitsbergen between 2005 and 2008 and these results both update and provide new insight on how climate change via glacial retreat could affect the previous AMAP assessments of radioactive contamination on Spitsbergen (AMAP 1997, 2004).

2.3.2.1 Peatlands

Peatlands are a poorly investigated component of the Arctic environment, especially in the High Arctic. A major reason for increased study of peatlands in these areas is because their contribution to biogeochemical and hydrological cycles is expected to increase under climate warming (ACIA 2004). Changes in air temperature and hydrological conditions may enhance humification and/or decomposition rates of peats leading to enhanced release of the radionuclides present.

Łokas et al. (2013a) investigated two peatlands located in the southwestern part of Spitsbergen (Fig. 2.10), on the coast of the Greenland Sea. Peats were collected on a coastal flatland to the north of Hornsund. The ^{137}Cs activity concentrations varied from 2±1 to 292±17 Bq/kg dw and were mostly undetectable at the bottom of the profiles. The uppermost layers of the peat profiles comprised live moss and the ^{137}Cs activities therein

WJL: Wedel-Jarlsberg Land
DL: Dickson Land
OIIL: Oscar II Land
SG: Scott Glacier
RG: Renard Glacier
WG: Wernenskiold Glacier
PT: Peatlands

Figure 2.10 Location of sampling areas on Spitsbergen.

ranged from below the detection limit to 90 Bq/kg dw. These ^{137}Cs concentration ranges are higher than typical values for Svalbard soils reported by Negoita (1997, 1999) of <1.5 to 35.8 Bq/kg dw and AMAP (2004) of <0.5 to 63±5.6 Bq/kg dw. Elevated levels of ^{137}Cs, with average activity concentrations of 103 Bq/kg dw, have also been observed in soil samples below a seabird colony in Kongsfjord, comprising humified organic material or peat underlain by mineral layers (AMAP 2004). Activity concentrations of $^{239+240}$Pu varied between 0.05±0.01 and 22.93±2.15 Bq/kg dw, while ^{238}Pu concentrations ranged from 0.02±0.01 to 0.53±0.11 Bq/kg dw. Activity concentrations of ^{241}Am ranged from 0.04±0.02 to 10.67±1.13 Bq/kg dw (Łokas et al. 2013a).

Table 2.6 presents total radionuclide inventories for $^{239+240}$Pu, ^{238}Pu, ^{241}Am and ^{137}Cs in all peat profiles. The ^{137}Cs inventories agree with the decay-corrected estimates of deposition on Svalbard (AMAP 1997; UNSCEAR 2000) estimated in the range 0.4–2.1 kBq/m^2 but are lower than the single value of 1.6 kBq/m^2 provided for the Kongsfjord area by Gwynn et al. (2004). Radionuclide inventories differ significantly between profiles (Table 2.6), possibly indicating the non-uniform initial deposition of fallout from the atmospheric nuclear weapons tests of the late 1950s / early 1960s over the study area, or differences in binding capacity for radionuclides between profiles. Inventory differences between profiles could also reflect post-depositional processes such as re-distribution of fallout-laden snow and non-uniform infiltration of snowmelt.

2.3.2.2 Tundra soils

Tundra soils in the High Arctic are characterized by slow rates of decomposition, a shallow active layer and the direct uptake of contaminants from the atmosphere and surface water. Soil profiles from tundra of 6–17 cm depth were collected between 2005 and 2008 from four parts of Spitsbergen: southern and northern Wedel-Jarlsberg Land, Dickson Land, and the western part of Oscar II Land (Fig. 2.10). Average ^{137}Cs activity concentrations in tundra soil profiles in western and central Spitsbergen were 31±3 Bq/kg dw (range <1.0–180±7 Bq/kg dw). Maximum ^{137}Cs activity was observed at 2.5–6 cm depth and in one surface layer sample. The activity concentrations were comparable with those reported for other Svalbard tundra soils, for example, the average ^{137}Cs activity concentration in surface soils in 2001 and 2002 was 21 Bq/kg dw (AMAP 2004).

Levels of actinides were also measured in tundra soil profiles. The average $^{239+240}$Pu, ^{238}Pu and ^{241}Am activity concentrations were 0.79±0.07 Bq/kg dw (range <0.03 to 4.80±0.41 Bq/kg dw), 0.07±0.02 Bq/kg dw (range <0.03 to 0.17±0.02 Bq/kg dw) and 0.54±0.04 Bq/kg dw (range <0.03 to 1.99±0.14 Bq/kg dw), respectively.

Table 2.7 presents the ^{137}Cs inventories in tundra profiles (ranging from 315±22 to 1436±24 Bq/m^2), which are consistent with previous results for ^{137}Cs (0.4–2.1 kBq/m^2) reported for Svalbard (AMAP 1997; UNSCEAR 2000). An estimate of $^{239+240}$Pu deposition from atmospheric weapon testing within the 70–80°N band is 13.3 Bq/m^2 (Hardy et al. 1973). For Svalbard, Holm et al. (1983) reported $^{239+240}$Pu deposition within the range 14–26 Bq/m^2. The $^{239+240}$Pu inventories reported in Table 2.7 sometimes exceed these values in the tundra profiles (range 6.8±0.6 to 42.2±0.9 Bq/m^2). The ^{238}Pu inventories presented in Table 2.7 (range 0.3±0.1 to 1.5±0.3 Bq/m^2) also exceed the fallout-derived deposition of 0.3 Bq/m^2 reported by Hardy et al. (1973). The data in Table 2.7 for ^{241}Am depositional fluxes on Svalbard are unique in the literature. The ^{241}Am inventories in all analyzed tundra soils ranged from 1.9±0.2 to 23.6±3.0 Bq/m^2 (Table 2.7).

2.3.2.3 Proglacial zones of glaciers

Uncovering new land is an environmentally important result of glacier retreat. De-glaciated areas are important to scientific research, especially in Arctic areas, because soils at the tip of glaciers can offer unique possibilities to study biogenic and abiogenic processes and their interactions induced by accelerated climate change. The rate of glacier transgression and regression (thawing) plays a key role in the formation of postglacial landscapes (Karczewski 1982; Lønne and Lysså 2005). Proglacial zone soils are formed from recently deposited basal moraines (Kabała and Zapart 2009, 2012) and characterized by a poor morphological development; that is, the parent material has not yet been modified by soil-forming processes.

Levels of anthropogenic radionuclides have been measured in 6–23 cm deep soil profiles collected in the proglacial zones of the Scott, Renard and Werenskiold glaciers. The average ^{137}Cs, $^{239+240}$Pu, ^{238}Pu and ^{241}Am activity concentrations were 146±5 Bq/kg (range <1 to 3293±101 Bq/kg), 1.94±0.17 Bq/kg (range <0.02 to 20.4±1.6 Bq/kg), 0.15±0.02 Bq/kg (range <0.02 to 1.33±0.12 Bq/kg) and 1.36±0.15 Bq/kg (range 0.16 to 14.2±0.9 Bq/kg), respectively.

Relatively high activity concentrations of ^{137}Cs, $^{239+240}$Pu and ^{241}Am (up to 3293 Bq/kg, 20.4 Bq/kg, and 14.2 Bq/kg, respectively) were found in soil profiles from proglacial areas adjacent to the Scott and Werenskiold glaciers. Elevated

Table 2.6 Total inventories for $^{239+240}$Pu, ^{238}Pu, ^{241}Am and ^{137}Cs for peat profiles obtained from southwestern Spitsbergen in 2005–2008 (Łokas et al. 2013a). See Fig. 2.10 for sampling locations.

Peat profile	$^{239+240}$Pu, Bq/m^2	^{238}Pu, Bq/m^2	^{241}Am, Bq/m^2	^{137}Cs, Bq/m^2
T1	29±3	0.8±0.2	13.4±1.1	479±113
T2	64±6	1.6±0.2	27.6±2.9	909±81
T3	21±2	0.9±0.6	7.7±0.9	847±71
T4	26±2	1.0±0.1	10.1±1.2	473±110
T5	26±3	1.3±0.1	4.5±0.4	1066±65
T6	32±3	0.8±0.2	11.7±1.1	192±35

Table 2.7 $^{239+240}$Pu, ^{238}Pu, ^{241}Am and ^{137}Cs inventories in tundra profiles from four areas of west and central Spitsbergen in 2005–2008 (Łokas et al. 2013a).

New tundra profile code	Old tundra profile code	$^{239+240}$Pu, Bq/m^2	^{238}Pu, Bq/m^2	^{241}Am, Bq/m^2	^{137}Cs, Bq/m^2
H1	GA	30±3	1.1±0.2	14±2	1336±129
H2	GB	23±2	-	12±4	971±56
H3	GC	24±2	-	11±4	694±83
C4	CAL4	21.0±1.8	0.7±0.1	10.2±0.7	813±127
C8	CAL8	42.2±0.9	0.8±0.3	8.8±0.7	1217±99
C9	CAL9	13.8±1.2	-	8.0±0.7	315±22
C10	CAL10	18.2±2.0	-	17.6±1.2	495±25
C11	CAL11	31.9±2.2	-	20.1±1.2	1436±24
P1	PET1	20.1±1.6	1.1±0.8	6.3±0.5	495±38
P3	PET3	27.1±2.0	1.5±0.4	12.6±0.8	811±70
P4	PET4	14.8±1.2	0.3±0.1	5.0±0.4	558±80
P6	PET6	30.7±2.8	1.3±0.3	9.5±0.9	1125±56
P7	PET7	28.4±2.6	1.5±0.3	15.7±1.1	813±117
P8	PET8	14.8±1.2	0.3±0.1	4.9±0.5	496±34
K1	S1	28.4±2.2	0.96±0.2	14.2±2.6	505±87
K3	S3	6.8±0.6	-	1.9±0.2	148±42
K4	S4	22.4±1.8	0.6±0.1	8.5±0.5	673±101
K5	S5	25.2±4.0	0.4±0.1	11.8±1.5	412±34
K6	S6	15.3±1.3	-	6.9±0.6	449±26

activity concentrations of these radionuclides in proglacial zone soils probably reflect their accumulation in cryoconites (Box 2.1), which are very effective in retaining trace metals. Cryoconites develop on the surface of glaciers and the material they accumulate is deposited on the land surface after the glaciers retreat (Łokas et al. 2013b, 2014). The elevated activity concentrations that result are of considerable interest and will be an important factor when considering the likely development of plant cover over these areas, as this affect the uptake of radionuclides into Arctic food chains.

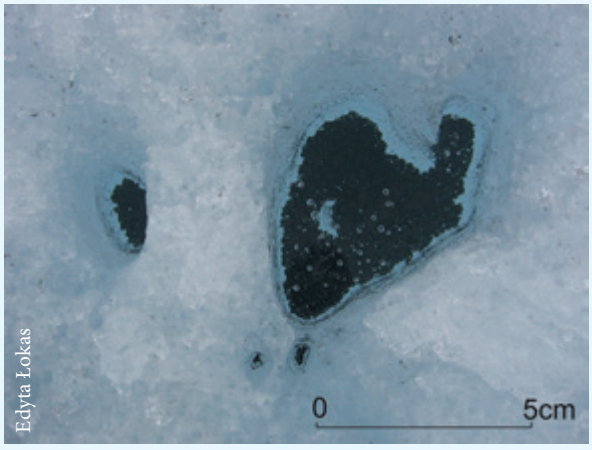
Table 2.8 show the inventories of $^{239+240}$Pu, ^{238}Pu, ^{241}Am, and ^{137}Cs in proglacial zones of the of the Scott, Renard and Werenskiold glaciers. In five soil profiles from proglacial areas adjacent to the Scott and Werenskiold glaciers the range in inventories is 131–946 Bq/m^2 ($^{239+240}$Pu), 10–63 Bq/m^2 (^{238}Pu), 75–577 Bq/m^2 (^{241}Am) and 12,000–120,000 Bq/m^2 (^{137}Cs), which means levels are much greater than previously reported reference values.

2.3.3 Activity ratios and sources of radionuclides

Activity ratios of ^{238}Pu:$^{239+240}$Pu, $^{239+240}$Pu:^{137}Cs, ^{241}Am:$^{239+240}$Pu and the atom ratio ^{240}Pu:^{239}Pu may be used to identify the sources of these radionuclides. This is because Pu isotope ratios vary with reactor type, nuclear fuel burn-up time, neutron flux, and energy; and from weapon type and yield after nuclear detonations. ^{240}Pu/^{239}Pu atom ratios enable weapons-grade Pu (ratios of 0.01–0.05) to be distinguished from civil reprocessing (ratios of 0.2–0.8) and global fallout (ratios of 0.17–0.19), allowing the study of Pu transported from different sources (Oughton et al. 2000). The corresponding ^{238}Pu:$^{239+240}$Pu activity ratio for global fallout is 0.02-0.04 (Oughton et al. 1999).

Box 2.1 Cryoconite

Cryoconite is powdery windblown dust made of a combination of small rock particles, soot and microbes which is deposited and builds up on snow, glaciers, or ice caps. The dark color absorbs solar radiation causing melting beneath the deposit, sometimes creating a cryoconite hole. Cryoconite was first described and named by Nils A. E. Nordenskiöld when he traveled on Greenland's ice cap in 1870. During summer, cryoconite holes frequently contain liquid water and thus provide a niche for cold-adapted microorganisms like bacteria, algae and animals like rotifers to thrive.



Glacier surface covered with cryoconites.

Table 2.8 $^{239+240}$Pu, ^{238}Pu, ^{241}Am, and ^{137}Cs inventories in soil profiles from the proglacial zones of the Werenskiold, Scott and Renard glaciers in western Spitsbergen in 2005–2008 (Łokas et al. 2013a). See Fig. 2.10 for sampling locations.

New soil profile code	Old soil profile code	$^{239+240}$Pu, Bq/m^2	^{238}Pu, Bq/m^2	^{241}Am, Bq/m^2	^{137}Cs, Bq/m^2
WG1	GL1	164±14	3.4±0.9	70±8	4000±200
WG2	GL2	68±6	4.4±0.7	33±4	2900±150
WG3	GL3	131±10	12.2±1.7	68±6	13400±500
WG4	GL4	79±9	4.8±1.4	-	4900±500
WG5	GL5	163±14	2.0±0.2	62±9	6900±400
WG6	GL6	58±5	2.4±0.5	-	1900±200
WG10	GL10	946±83	63±7	577±64	119900±3800
WG11	GL11	482±39	18±3	162±14	15100±500
SG6	CAL6	886±80	47±6	296±19	30910±941
SG7	CAL7	188±14	9.9±1.5	75±8	12355±941
RG12	CAL12	7.0±0.6	-	3.4±0.3	276±31

2.3.3.1 Peatlands and tundra soils

An average ^{238}Pu:$^{239+240}$Pu ratio of 0.034 for the peat profiles (range 0.02±0.01 to 0.09±0.03) shows their principal origin is global fallout. The average activity ratio can also be represented by the slope of the regression line in a ^{238}Pu-$^{239+240}$Pu correlation plot (Mietelski et al. 2008), giving a value of 0.023 (Fig. 2.11, top row). These values agree with previously reported average ^{238}Pu:$^{239+240}$Pu ratios for Spitsbergen of 0.025 (Hardy et al. 1973) and fall within the range 0.03–0.05 after the burn-up of the SNAP-9A satellite (UNSCEAR 1982).

For the peatlands, the average $^{239+240}$Pu:^{137}Cs activity ratio is 0.060 (range 0.01±0.01 to 0.42±0.11); the slope of the regression line in the $^{239+240}$Pu-^{137}Cs correlation plot is equal to 0.062, which is greater than a ratio of 0.050 calculated from the data of Beck and Krey (1983). The discrepancies between their 1983 estimate and the observed ratios could reflect sources of ^{137}Cs other than global fallout, or differences in sorption-desorption dynamic equilibria due to radionuclide fractionation processes.

The average ^{241}Am:$^{239+240}$Pu activity ratio in these profiles is 0.46 (range 0.10±0.02 to 1.5±0.3). The slope of the ^{241}Am-$^{239+240}$Pu regression line is 0.41 which is slightly higher than the published global fallout ratio for Svalbard of 0.37 (Holm et al. 1983), possibly due to the higher geochemical mobility of Pu compared to Am or ingrowth of ^{241}Am from the decay of ^{241}Pu.

The atom ratio ^{240}Pu:^{239}Pu ranged from 0.142±0.006 to 0.241±0.027 (mean 0.182±0.007). Values deviating significantly from the expected stratospheric fallout ^{240}Pu:^{239}Pu range of 0.180±0.014 could possibly signal the presence of non-fallout Pu sources; however, values falling outside this range were calculated for samples with relatively low Pu concentrations, for which reliable ^{240}Pu/^{239}Pu measurements are often problematic. Notwithstanding the few outliers, the majority of peat samples exhibited ^{240}Pu:^{239}Pu atom ratios similar to the stratospheric fallout (~0.18) (Łokas et al. 2013a).

In soil profiles from tundra the average ^{238}Pu:$^{239+240}$Pu ratio was 0.041 (range 0.021±0.005 to 0.075±0.034). The average activity ratio represented by the slope of the regression line in the ^{238}Pu-$^{239+240}$Pu correlation plot was 0.030 (Fig. 2.11, middle row). The ^{238}Pu:$^{239+240}$Pu activity ratios for tundra soils again indicate that global fallout was the dominant source of Pu.

The average $^{239+240}$Pu:^{137}Cs activity ratio was 0.039 (range 0.007±0.001 to 0.21±0.05) and the slope of the regression line in the $^{239+240}$Pu-^{137}Cs correlation plot was 0.024. These values are lower than the decay-corrected value of ~0.05 expected for 2012 by Beck and Krey (1983), possibly indicating enrichment in ^{137}Cs content relative to plutonium due to the contribution from sources other than global fallout.

The average ^{241}Am:$^{239+240}$Pu activity ratio in these profiles was 0.49 (range 0.26±0.02 to 1.1±0.1). The slope of the regression line in the ^{241}Am-$^{239+240}$Pu correlation plot was 0.37 which agrees with the published global fallout ratio for Svalbard of 0.37 (Holm et al. 1983).

2.3.3.2 Proglacial soils

The average ^{238}Pu:$^{239+240}$Pu ratio was 0.059 for soil profiles from the proglacial zone adjacent to the Scott, Renard and Werenskiold glaciers (range 0.026±0.005 to 0.090±0.011). The average activity ratio as represented by the slope of regression line in the ^{238}Pu-$^{239+240}$Pu correlation plot is equal to 0.066 (Fig. 2.11, bottom row).

The average $^{239+240}$Pu/^{137}Cs activity ratio is 0.026 (range 0.006±0.001 to 0.063±0.007) and the slope of the regression line in the $^{239+240}$Pu-^{137}Cs correlation plot is equal to 0.0065. The ^{238}Pu:$^{239+240}$Pu and $^{239+240}$Pu:^{137}Cs activity ratios for proglacial soils suggest possible contributions from sources other than global fallout, such as the regionally significant nuclear weapons testing on Novaya Zemlya.

The average ^{241}Am:$^{239+240}$Pu activity ratios in these profiles were 0.48 (range 0.28±0.03 to 0.85±0.60). The slope of the ^{241}Am-$^{239+240}$Pu ratio correlation plot is equal to 0.59. This is higher than the typical values of 0.3–0.4 reported by Smith et al. (1997), while Gwynn et al. (2005) reported values of 0.38±0.08 to 0.65±0.19, and Łokas et al. (2013a) reported 0.46 for peat profiles in southern Spitsbergen. Thus, the ^{241}Am:$^{239+240}$Pu ratio seems geographically variable and in growth of ^{241}Am may contribute to the observed scatter of ^{241}Am/$^{239+240}$Pu.

The preliminary average atom ratio ^{240}Pu:^{239}Pu of 0.178±0.002 (range 0.148±0.001 to 0.201±0.001) for tundra soils indicates global fallout (~0.18) as a dominant source of Pu. However,

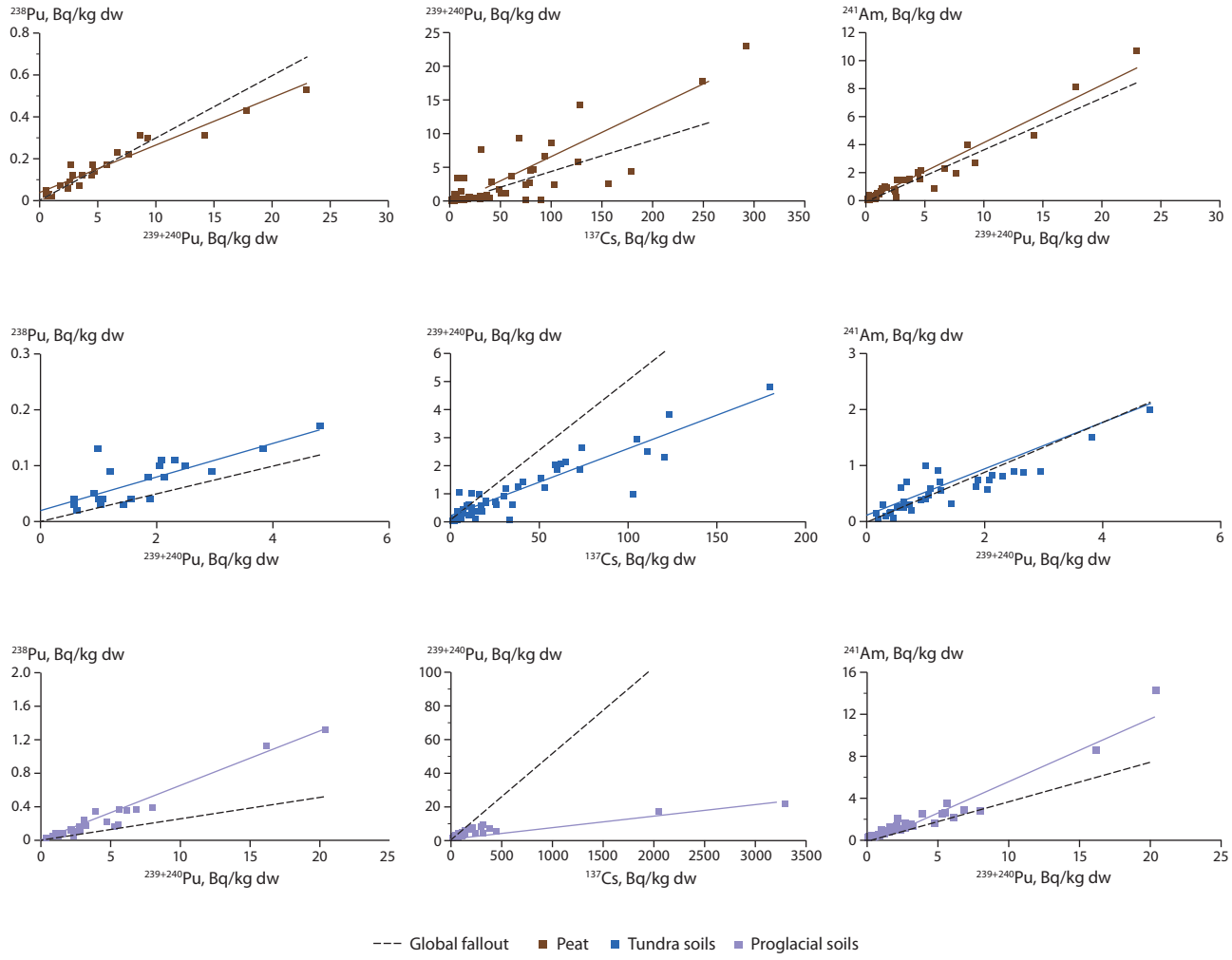


Figure 2.11 Correlation plots for Pu isotopes versus Cs and Am for soil samples from peat samples, tundra soil samples and the proglacial zone of glaciers.

non-fallout Pu sources cannot be ruled out in soil profiles from the proglacial zone of glaciers where the average atom ratio $^{240}Pu:^{239}Pu$ of 0.170±0.002 (range 0.128±0.001 to 0.234±0.005).

2.4 Conclusions

Although several of the risks associated with radioactive contamination in the Arctic have been reduced in the past five to ten years (by reducing discharges, dismantling/decommissioning nuclear submarines, clear-up work at Andreeva/Gremhika, and RTG removal, among others) there is still a clear need for scientific monitoring of radioactivity in the Arctic in relation to the large amounts of radioactivity dumped at sea in the past (and the potential for corrosion/leakage of the containers), the possible effects of climate change on the remobilization of radioactivity, and the potential for future accidents at nuclear facilities.

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3. Radioactive waste and decommissioning

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Key findings

- *Good progress has been made in mitigating risks from potential radioactive contamination in northwestern Russia – decommissioning radioisotope thermoelectric generators and nuclear submarines, the nuclear waste vessel Lepse, and managing and remediating sites of temporary storage for radioactive wastes at Gremikha and Andreeva Bay*
- *The situation at Gremikha is much improved with nearly all the spent nuclear fuel having now been removed from the site*
- *Although the decommissioning work at Andreeva Bay is not as advanced as at Gremikha, with most of the spent nuclear fuel and radioactive waste at the time of the previous AMAP assessment still on-site, the Russian authorities responsible for this work have allocated financial support and the decommissioning plan is in progress*
- *The long-term consequences of decommissioning activities in northwestern Russia for Mayak PA and its surrounding environment remain uncertain*

3.1 Introduction

Potential sources of anthropogenic radionuclides in the Arctic environment include spent nuclear fuel and radioactive wastes; much of this located in northwest Russia. Ongoing international efforts, working together with relevant Russian authorities to reduce the risk to human health and the Arctic environment, are focusing on managing and decommissioning the Gremikha and Andreeva Bay sites of temporary storage and the *Lepse* floating technical base, decommissioning nuclear submarines, and removing radioactive sources from radioisotope thermoelectric generators. Most of the removed spent nuclear fuel and radioactive waste is destined for long-term storage at Mayak PA and the consequences of the influx of radioactive materials from these decommissioning activities for Mayak PA are not yet known. A potential future source of contamination not addressed in previous AMAP assessments is the sunken *K-159* nuclear submarine.

Environmental radiological monitoring in and around decommissioning sites is important, particularly during critical phases of preparing and removing radioactive waste, because some areas of the site will still be contaminated even after the waste has been removed. Monitoring also helps identify radioactivity leakages.

3.2 Sites of Temporary Storage: Gremikha and Andreeva Bay

The previous AMAP assessment (AMAP 2010b) gave a brief introduction to the shore technical bases established during the 1960s at Gremikha and Andreeva Bay on the Kola Peninsula (see Fig. 3.1). After servicing a large number of nuclear submarines, an activity which ended in the 1990s, the sites became Sites of Temporary Storage (STS) for spent nuclear fuel and radioactive waste. Since their change of use in 2000, maintenance work at both sites has been inadequate and the facilities have rapidly deteriorated in the harsh, coastal, Arctic conditions.

Previously designated as shore technical bases by the Russian military navy, Andreeva Bay STS and Gremikha STS are now operated by SevRAO, an organization within the Rosatom Group. Their significance is due to the large amounts of spent nuclear fuel and radioactive waste which have been stored there historically. The situation at Gremikha is much improved with nearly all the spent nuclear fuel having now been removed from the site (see Box 3.1 for the current status of decommissioning). This is not the case at Andreeva Bay, where most of the spent nuclear

Box 3.1 Current status of decommissioning at the Gremikha STS

Decommissioning work at Gremikha STS had started by the time of the previous AMAP assessment (AMAP 2010b) and was on schedule with the first intended shipment of spent nuclear fuel to Murmansk and then on to Mayak scheduled for 2008 (CEG 2008). Shipment of the first 294 spent fuel assemblies took place in June 2009. A significant part of the clean-up operations at Gremikha has been completed using French financing and in cooperation with the French Atomic Energy Commissariat (CEA). Further progress has been made since 2009, with all 898 spent fuel assemblies from the Russian reactor type 'VVR' (water cooled, water moderated reactor) reportedly removed from Gremikha during 2012 (Stepennov 2013) and phase 3 of the remediation plan for the site is now underway. Phase 3 involves safely preparing and removing radioactive wastes from the site and decommissioning buildings that have been used for temporary storage of spent fuel assemblies. One of the first steps is to deal with 'safety and control' rods (high level solid radioactive waste) from the first generation type VVR nuclear submarines that are currently stored under non-standard conditions. Two pits containing about 100 control rods (which contain neutron-absorbing material and can be inserted into the reactor core to regulate power output in a nuclear reactor) are targeted, with planned removal of the control rods from the site in specially designed transport casks constructed of metal with concrete shielding on the inside (termed 'TUK' casks) for long-term storage at Saida Bay (see Sect. 3.5).

fuel and radioactive waste in the inventory at the time of the previous AMAP assessment is still on-site, totalling some 21,000 spent nuclear fuel assemblies and about 12,000 cubic meters of radioactive waste (AMAP 2010b). A limited trial recovery and transport of spent nuclear fuel from the site was undertaken in 2010, using the vessel *Serebryanka* to transport the material to the Atomflot enterprise near Murmansk (Sneve et al. 2015). The degraded state of the spent nuclear fuel at Andreeva Bay, previous failures of containment barriers and the overall poor condition of facilities (see Fig. 3.1) have previously been recognized as requiring increased attention (Ilyin et al. 2005).

Andreeva Bay STS and Gremikha STS are covered by the Norwegian Radiation Protection Authority (NRPA) regulatory cooperation program with the Federal Medical-Biological Agency of Russia (FMBA). The data described in this chapter are mostly derived from work of the Federal Medical-Biological Center (FMBC) as part of this program, under the supervision of the FMBA. The cooperation program between NRPA and FMBA initially focused on determining the most hazardous situations and activities requiring enhanced regulatory supervision. This culminated in the development of new and updated norms and standards, and related regulatory procedures. The latter were needed to cope with the often atypical conditions at nuclear legacy sites (Ilyin et al. 2005; Shandala et al. 2008a). Regulatory developments also took place in the radiological protection of workers, medical radiological aspects of emergency preparedness and response, and radiological criteria for site remediation (Savkin et al. 2008; Shandala et al. 2008b; Simakov et al. 2008). Sneve et al. (2008) gave an overview of the initial regulatory developments. Information concerning further progress has been reported by Roudak et al. (2011), Chizhov et al. (2014) and Sneve et al. (2015).

The Russian Federation has operated a program of site rehabilitation at Andreeva Bay STS with international collaborators over several years. Shandala et al. (2008a) and Roudak et al. (2011) gave a substantial account of progress from a safety and regulatory perspective, as well as presenting data on the radioactive source terms and other relevant information about the Andreeva Bay site. The decommissioning work planned at Andreeva Bay STS is discussed in detail later in the chapter (see Sect. 3.2.4). An effective and efficient regulatory framework is important to decrease future risk to human health and the Arctic environment when managing and decommissioning Andreeva Bay STS. As shown in Box 3.1 for the Gremikha STS, the environmental threat represented by such sites can be greatly reduced.

3.2.1 Radiological characterization of terrestrial and marine environments at Andreeva Bay

The main facilities at the Andreeva Bay site are shown in Fig. 3.2. New buildings and infrastructure have been constructed on-site

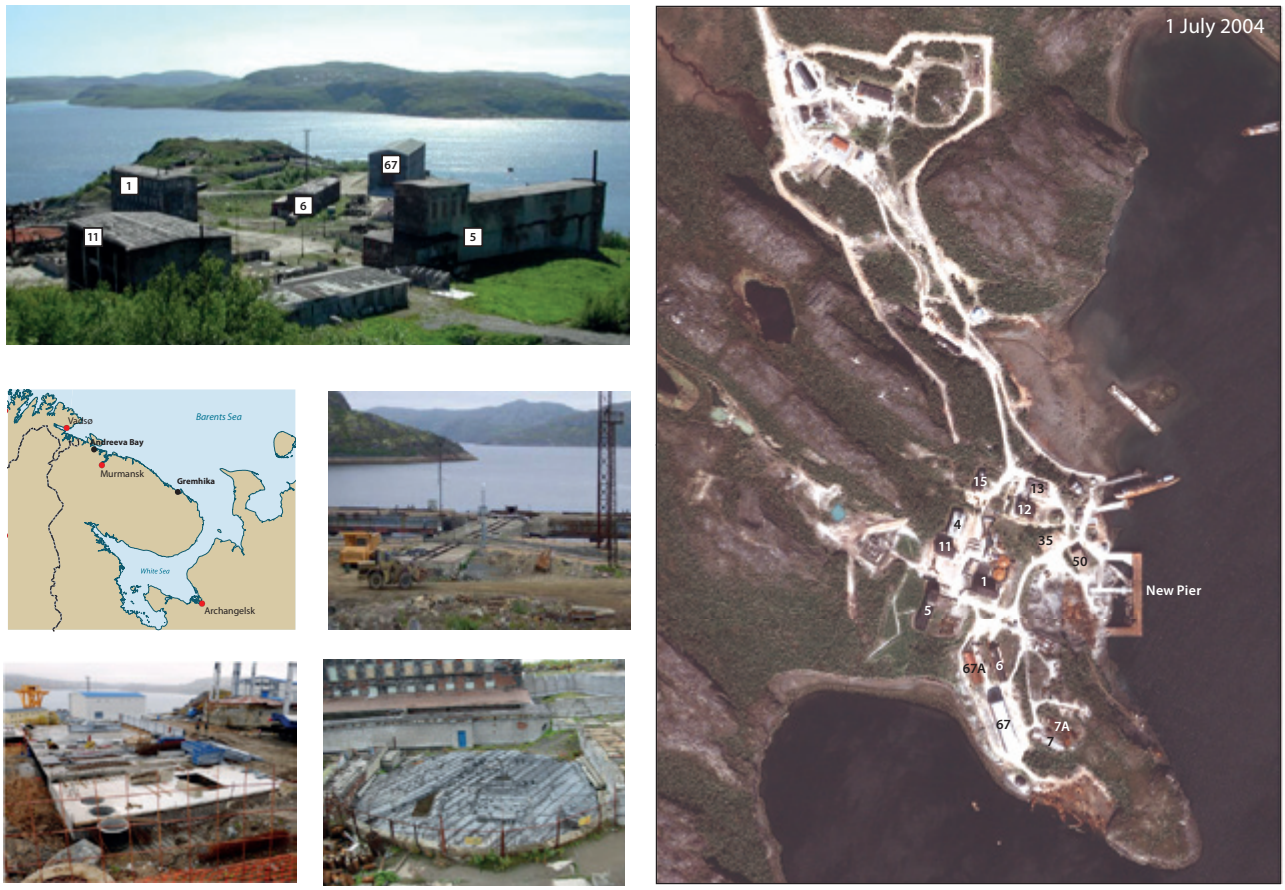

Figure 3.1 View of the Andreeva Bay STS site in 2004 (right) and general views of the Andreeva Bay site prior to major remediation activities (photos) (Sneve et al. 2014). Numbers indicate building numbers referred to in the text.

for recovering and handling spent nuclear fuel and managing radioactive waste. A system of zoning for radiation control has been adopted at the site. The areas where spent nuclear fuel and radioactive waste are stored and where the most radiation-hazardous operations are undertaken are designated as the Controlled Area (CA). The area outside the CA which falls under STS administrative and technical provision is termed the Health Protection Zone (HPZ). The external border of the HPZ is delineated by a system of physical protection. A zone with a radius of about 10 km around the STS is termed the Supervision Area (SA). This is subject to radiological monitoring and control appropriate to possible impacts on the environment and the public. Designation of an area according to these three categories is not necessarily permanent. The extent of different zones may change over time according to progress with decommissioning operations.

Preliminary investigations indicated that ^{90}Sr and ^{137}Cs were the most prevalent anthropogenic radionuclides of radiological significance in the external environment (i.e. not inside buildings or stores). The main sampling and measurement activities have therefore been gamma dose rate measurements and environmental sampling to identify and measure the distribution of these radionuclides.

Figure 3.3 indicates the locations of gamma dose rate measurements made by FMBC in cooperation with SevRAO in 2010 and 2012. The energy photon spectra were recorded using a portable spectrometer MKS-01A Multirad-M combined with a portable 63×63 mm NaI scintillation detector. The photon energy spectra were registered with the locations of gamma dose rate measurements, performed at 1 m above the ground surface. Over 1000 measurements were taken each year. The measurement locations were selected to reflect the likely pathways that decommissioning workers frequent when moving around the site. This was done in order to optimize such journeys from a dose control perspective. An important aspect of this work was therefore to document changes in dose rates relevant for workers on-site as remediation operations progress, particularly during critical phases of the preparation and removal of spent nuclear fuel and radioactive waste.

Table 3.1 summarizes the gamma dose rate results at Andreeva Bay STS in 2010 compared to previous measurements performed

Table 3.1 Comparison between gamma dose rate measurements (1 m height above ground) at Andreeva Bay STS in 2010 (Sneve et al. 2014) and 2002–2003 (Reistad et al. 2008).

Zone of measurement	Dose rate, μSv/h		
	Minimum	Mean	Maximum
2010			
Controlled Area (CA)	0.07	3.7	103.6
Health Protection Zone (HPZ)	0.1	0.9	7.3
Supervision Area (SA)	0.07	0.13	0.2
2002–2003 Entire site	0.1	14.7	3000

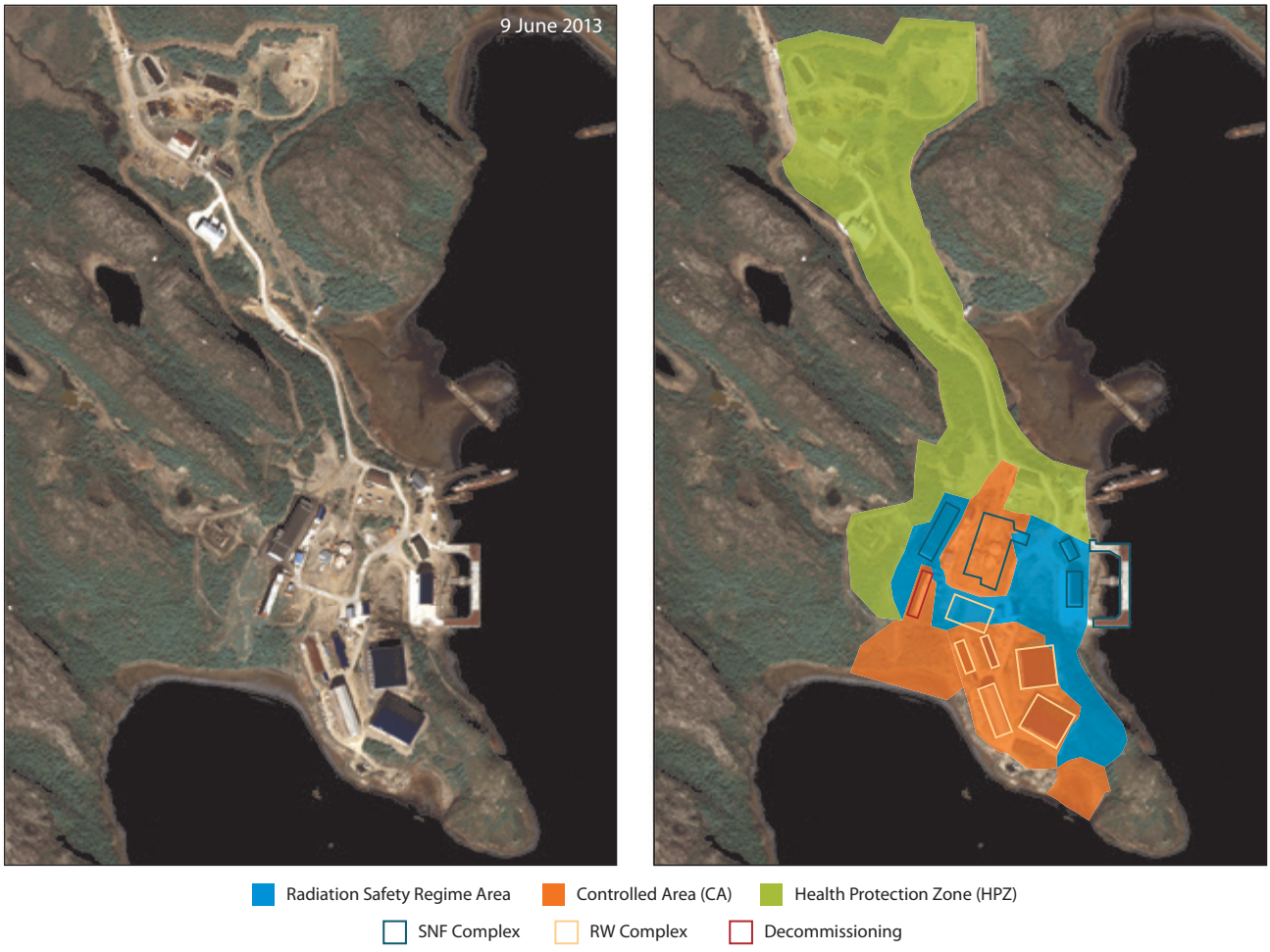

Figure 3.2 View of the Andreeva Bay STS site in 2013, showing the extent of the Radiation Saftey Regime, Controlled Area and Health Protection Zone. Main facilities at Andreeva Bay STS comprise the spent nuclear fuel (SNF) complex, the site of decommissioning operations, the radioactive waste (RW) complex, various supporting facilities and Pier PMK-67 (modified from Sneve et al. 2014).

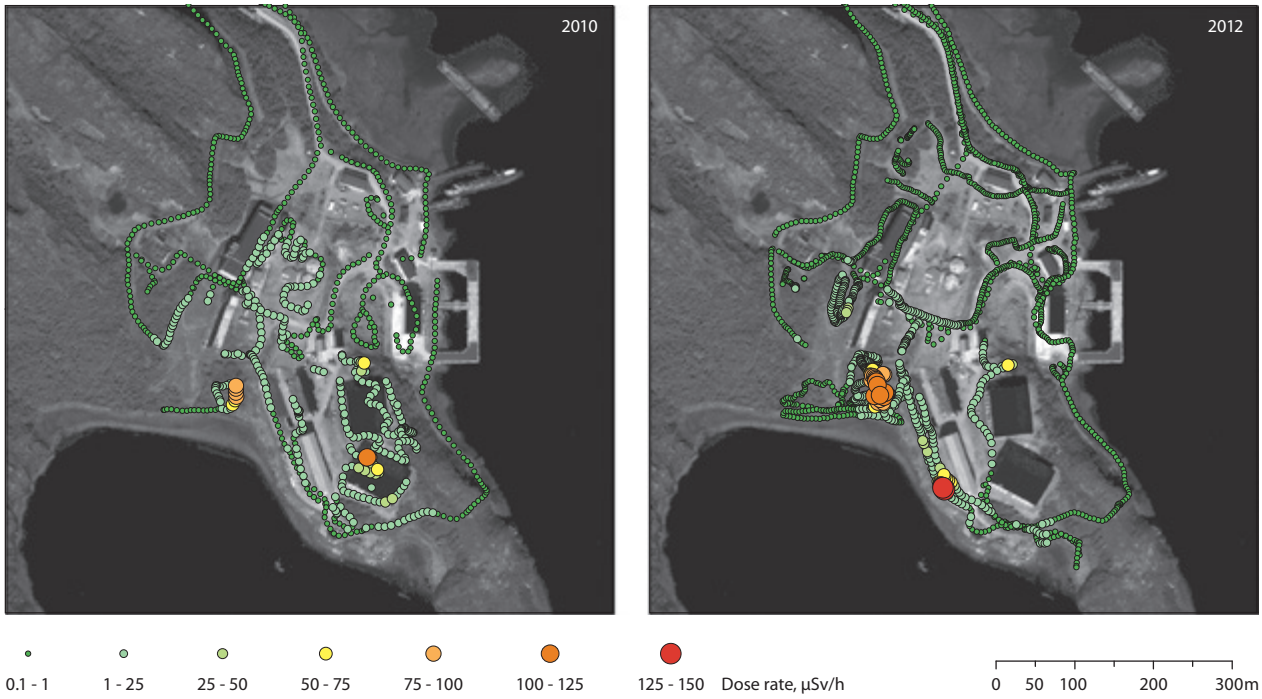

Figure 3.3 Gamma dose rate measurement routes at Andreeva Bay STS made by the Federal Medical-Biological Center (FMBC) of Russia in cooperation with SevRAO in 2010 and 2012 (Sneve et al. 2014).

in 2002–2003 (n=1035: Reistad et al. 2008). The natural background level in the region is in the range 0.1–0.15 µSv/h. The correspondence between dose rates and zoning at the site (based on 2010 measurements) is clear, and spectrometry results confirmed that ^{137}Cs made the dominant contribution to radiation dose rate.

The data indicate a significant reduction in the maximum gamma dose rate in 2010 compared to 2002–2003. Maximum dose rates of up to 1000 µSv/h close to the ground (0.1 m) and up to 3000 µSv/h at 1 m above ground were recorded during 2002–2003. The higher doses at 1 m were mostly explained by the presence of local contaminated equipment rather than ground contamination. The highest dose rates were measured close to buildings used to store spent nuclear fuel and other radioactive wastes. However, isolated patches with elevated dose rates were also observed throughout the site. For example, elevated dose rates were observed near the former channel of a small brook, which became heavily contaminated by radioactive leaks in the early 1980s from the spent nuclear fuel storage Building 5.

The gamma dose rate measurements made in 2002–2003 (Reistad et al. 2008) during a separate Russian–Norwegian collaboration covered the site comprehensively (see Fig. 3.4) and gave a clearer picture of the very inhomogeneous spread of radioactivity than the 2010 measurements. Direct comparisons between the two sets of measurements are therefore difficult. Reistad et al. (2008) used their data set to perform rank order kriging interpolations and generate dose maps of the entire site (see Fig. 3.4). Visual comparison indicates that hot spots identified in 2002–2003 did not feature prominently in the 2010 measurement sites.

The data presented in Table 3.2, recently published by Sneve et al. (2015) indicate that with regard to the locations remediation workers frequent further progress has been made in cleaning up sources of radioactivity and lowering dose rates since 2010 (Fig. 3.3), where corresponding positions indicated dose rates several orders of magnitude higher.

Table 3.2 Change in gamma dose rate following remediation action at the Andreeva Bay STS (Sneve et al. 2015). See Fig. 3.1 for location of buildings.

Measurement site	Dose rate, µSv/h			Remediation activity
	2002	2008	2013	
Near the new pier	0.1–450	0.15–0.35	0.12–0.24	Dismantlement of the old pier
Around Building 50	0.3–1.5	0.25–0.57	0.1–0.29	Elimination of the scrap yard
Around the dry storage facility	0.58–2.7	0.38–1.1	0.1–0.37	Filling with sand on the site of the demolished buildings, liquid radioactive waste storage facilities (2C, 2D), paving the territory between the newly constructed buildings
Location of the site for vehicle decontamination and sanitary pass	0.7–2.5	0.57–0.73	0.1–0.5	Paving the territory between the constructed buildings of site for vehicle decontamination and stationary sanitary pass
Buildings 7 and 7a (solid radioactive waste storage facilities)	1.1–130	0.4–45	0.1–0.5	Shelter constructed around the SRW storage facility, and landscaping

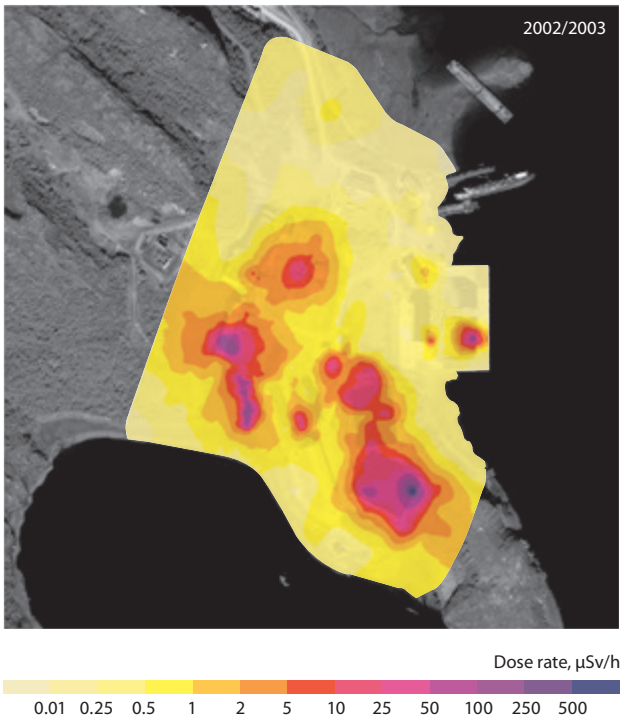

Figure 3.4 Interpolated dose rate levels at 1 m above ground level from measurements made in 2002–2003 across the Andreeva Bay STS site (Reistad et al. 2008). Note the log scale.

3.2.2 Environmental sampling and sample measurements

Samples of soil, vegetation and marine sediments were collected in the CA, HPZ and SA in 2010. Figure 3.5 shows the soil and marine sediment sample locations in the HPZ. The sample sites for vegetation in the HPZ and in the SA, described as motley grass, coincided with those for soil and used the same numbering. Vegetation samples were collected in the CA (five samples), HPZ (ten samples) and SA (ten samples). Analysis of gamma radiation dose rates was used to identify areas deemed of most interest for sampling.

Soil samples were collected using a metal corer (10 cm diameter) at depths between 0–10 cm and 10–20 cm from the surface. Each vegetation sample was at least one kilogram. Marine sediment samples were collected close to the shore at high tide. The samples of marine sediments were from the top 10 cm surface layer. The radionuclide content of samples was then measured by gamma spectrometry, radiochemical and radiometry methods at FMBC.

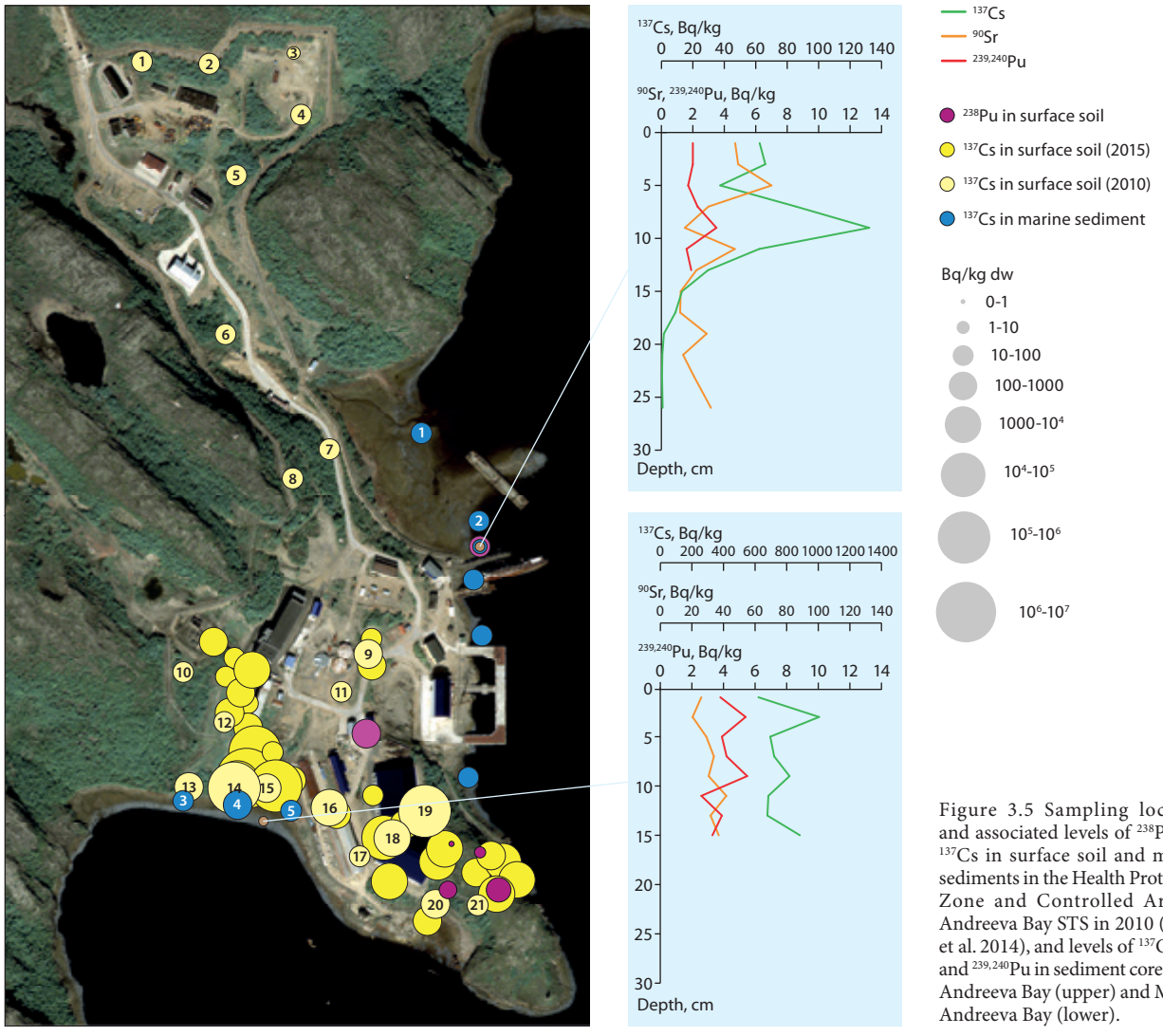

Figure 3.5 Sampling location and associated levels of ^{238}Pu and ^{137}Cs in surface soil and marine sediments in the Health Protection Zone and Controlled Area of Andreeva Bay STS in 2010 (Sneve et al. 2014), and levels of ^{137}Cs, ^{90}Sr and ^{239,240}Pu in sediment cores from Andreeva Bay (upper) and Malaya Andreeva Bay (lower).

Table 3.3 Radionuclide concentration in soil samples collected in the Health Protection Zone and Controlled Area at Andreeva Bay STS in 2010 (FMBC data) (Sneve et al. 2014). See Fig. 3.5 for sample sites.

Sample site	Soil (turf/sod unless stated otherwise)	^{137}Cs, Bq/kg dw	^{90}Sr, Bq/kg dw
1	0–10 cm	12.0	2.4
	10–20 cm	10.0	1.5
2	0–10 cm	14.3	3.4
	10–20 cm	10.1	1.9
3	0–10 cm	8.7	4.6
	10–20 cm	5.2	4.6
4	0–10 cm	14.6	3.4
	0–20 cm	10.2	3.7
5	0–10 cm	17.1	2.5
	10–20 cm	14.6	0.5
6	0–10 cm	32.5	2.0
	10–20 cm	34.0	1.3
7	0–10 cm	25.6	2.8
	10–20 cm	18.9	2.5
8	0–10 cm	21.3	0.8
	10–20 cm	14.1	0.9
9	0–10 cm	840.0	206.3
	10–20 cm	437.6	134.7
10	0–10 cm	35.7	2.8
	10–20 cm	21.6	1.5
11	Soil (verge) 0–10 cm	42.1	6.9
	10–20 cm	23.7	2.3
12	0–10 cm	21.6	3.6
	10–20 cm	15.6	2.0
13	0–10 cm	136.5	63.1
	10–20 cm	61.4	24.1
14	0–10 cm	830 000	11 000
	10–20 cm	36 000	11 000
15	0–10 cm	648.0	147.6
	10–20 cm	290.7	64.9
16	0–10 cm	2400	1300
	10–20 cm	1400	638.4
17	0–10 cm	27.6	2.8
	10–20 cm	15.1	0.6
18	0–10 cm	4100	407.1
	10–20 cm	2200	182.5
19	0–10 cm	360 000	29 000
	10–20 cm	964.1	875.3
20	0–10 cm	154.6	15.3
	10–20 cm	93.4	5.6
21	0–10 cm	36.7	5.9
	10–20 cm	12.4	3.2

Table 3.4 Radionuclide concentration in vegetation samples collected in the Health Protection Zone and Controlled Area at Andreeva Bay STS in 2010 (FMBC data) (Sneve et al. 2014). See Fig. 3.5 for sample sites.

Sample site	^{137}Cs, Bq/kg fw	^{90}Sr, Bq/kg fw
1	7.0	1.5
2	8.7	0.6
3	12.1	1.1
4	8.9	1.6
7	15.3	1.8
8	16.4	2.1
9	60.7	83.1
10	12.6	2.2
13	27.1	42.6
14	27 000	3100
15	53.6	36.8
17	12.9	3.2
18	563.2	261.8
19	14 000	10 000
20	21.6	2.7

Table 3.5 Radionuclide concentration in marine sediment samples collected within the Health Protection Zone at Andreeva Bay STS in 2010 (FMBC data) (Sneve et al. 2014). See Fig. 3.5 for sampling locations.

Sample site	^{137}Cs, Bq/kg dw	^{90}Sr, Bq/kg dw
1	22	3.1
2	85	12
3	15	2.8
4	140	5.1
5	40	10.1

Tables 3.3 and 3.4 show the results of the soil and vegetation sampling, respectively. In some areas, ^{137}Cs and ^{90}Sr activity levels at the soil surface are significantly above the background values typical for this region (<15 Bq/kg for ^{137}Cs and <4 Bq/kg for ^{90}Sr). Levels of ^{137}Cs and ^{90}Sr in the 10–20 cm soil layer are typically lower than in the top 10 cm, and levels of ^{137}Cs are typically 4–20 times greater than for ^{90}Sr. The contamination of soil and vegetation are correlated. These results are also reflected in the measurements of gamma radiation dose rate.

Table 3.5 shows the results of measurements of the marine bottom sediment samples collected within the HPZ in 2010. Elevated levels of radioactivity compared to the regional background can be observed. However, it is not apparent to what degree this is due to historic contamination or to continuing releases into the terrestrial environment.

Figure 3.6, Tables 3.6 and 3.7 show the results for soils and vegetation contamination in the SA. The mean ^{137}Cs and ^{90}Sr contents in soil and vegetation outside the area of radiation safety control, that is, beyond the industrial site, are typical for background levels in uncontaminated regions of northern Russia (i.e. <15 Bq/kg for ^{137}Cs and <4 Bq/kg for ^{90}Sr).

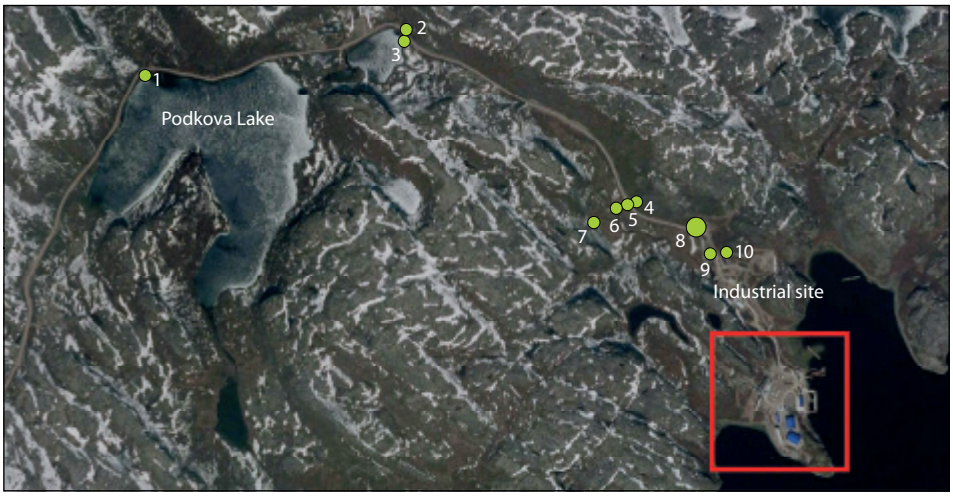

Figure 3.6 Activity levels of ^{137}Cs sampled in soil in the Supervision Area at Andreeva Bay STS in 2010 (Sneve et al. 2014). The red box corresponds to the area shown in Figs. 3.3 and 3.4.

Sample site	Description	Layer	^{137}Cs, Bq/kg dw	^{90}Sr, Bq/kg dw
Shore of the stream near Podkova Lake (point 1)	Soil (marshy)	0–10 cm	10	0.8
		10–20 cm	3.1	0.5
Point 2	Soil (turf/sod)	0–10 cm	8.3	1.2
		10–20 cm	2.3	0.9
Point 3	Soil (marshy)	0–10 cm	6.5	1.0
		10–20 cm	1.5	0.9
Point 4	Soil (turf/sod)	0–10 cm	7.8	1.3
		10–20 cm	1.3	0.6
Point 5	Soil (peat bog)	0–10 cm	9.1	2.3
		10–20 cm	4.5	1.5
Point 6	Soil (peat bog)	0–10 cm	7.8	2.6
		10–20 cm	3.6	1.1
Point 7	Soil (turf/sod)	0–10 cm	8.0	1.7
		0–20 cm	3.2	1.1
Point 8	Soil (turf/sod)	0–10 cm	13.1	1.5
		0–20 cm	2.2	1.4
Point 9	Soil (peat bog)	0–10 cm	8.1	2.4
		10–20 cm	3.1	2.2
Point 10 by the road close to the industrial site checkpoint and boundary	Soil (peat bog)	0–10 cm	5.7	0.7
		10–20 cm	1.4	0.7

Table 3.6 Radionuclide concentration in soil samples collected in the Supervision Area at Andreeva Bay STS in 2010 (FMBC data) (Sneve et al. 2014). See Fig. 3.6 for sampling locations.

Table 3.7 Radionuclide concentration in vegetation samples collected in the Supervision Area at Andreeva Bay STS in 2010 (FMBC data) (Sneve et al. 2014). See Fig. 3.6 for sampling locations.

Sampling point	Description	^{137}Cs, Bq/kg fw	^{90}Sr, Bq/kg fw
Shore of the stream near Podkova Lake (point 1)	Vegetation (motley grass)	7.5	2.1
Point 2	Vegetation (motley grass)	5.5	1.0
Point 3	Vegetation (motley grass)	6.3	1.8
Point 4	Vegetation (motley grass)	4.1	0.5
Point 5	Vegetation (motley grass)	6.6	1.5
Point 6	Vegetation (motley grass, sedge)	5.4	1.9
Point 7	Vegetation (motley grass)	9.4	2.1
Point 8	Vegetation (motley grass)	10.0	1.1
Point 9	Vegetation (motley grass, sedge)	2.7	0.3
Point 10	Vegetation (motley grass)	12.4	3.6

3.2.3 Characterization of the potential for radioactive contaminant migration

In the AMAP context, one of the most important aspects of studying the Andreeva Bay STS is to determine the likelihood of radioactivity leakages into the surrounding environment. Even after spent nuclear fuel and more active radioactive waste have been recovered and removed from the site, some areas of the site will still be radioactively contaminated. How the residual contamination is managed will depend partly on radionuclide mobility during the period before substantial radioactive decay has occurred. Preliminary work by FMBC has examined how strongly ^{137}Cs and ^{90}Sr attach to soils and marine sediments. Extractable fractions have been determined using 1N solution of ammonium acetate and 1N hydrochloric acid. For soils, the mobile fractions were approximately 90% (^{90}Sr) and 10% (^{137}Cs). For marine sediments, the mobile fractions were approximately 80% for both radionuclides. The differences can be explained by the different radionuclide forms being released to the marine environment and that the marine sediments contained some organic material while soils contained predominantly clay, which is known to reduce cesium mobility (e.g. Kemner et al. 1997). Such observations indicate the potential radionuclide migration off-site, particularly for ^{90}Sr. However it is important to note that ongoing operations at the site have not compromised contamination levels off-site (Sneve et al. 2014): Contamination levels in the SA appear to be very low, and at background levels for the region. Additional data regarding different radionuclides such as plutonium and americium would also be of interest to fully assess contamination in the SA and other zones of the Andreeva Bay STS.

Continued remediation efforts are expected to further reduce contamination levels overall at the site and specific areas that were the focus of previous environmental concern have already been successfully remediated.

3.2.3.1 Available evidence regarding levels of radioactivity off-site in areas surrounding Andreeva Bay STS

A 2014 study of the radioecological status of the marine environment in the area surrounding Andreeva Bay (MMBI/ApN, 2015) concluded that activity at Andreeva Bay STS has little effect on the radioecological status of the marine environment and biota in Malaya Andreeva and Andreeva Bays. Negligible impacts were also documented in Motovsky Bay and the estuary of Zapadnaya Litsa Bay, although technogenic radionuclides were observed intermittently in discreet sediment layers.

Seawater

The ^{137}Cs concentration is low in the waters surrounding Andreeva Bay STS, at 3–10 Bq/m^3. The highest ^{137}Cs concentrations were detected in Malaya Andreeva Bay and near Pier PMK-67 partly due to localized water/soil runoff due to local topography. ^{137}Cs concentrations in water near the STS are consistent with the regional background level (^{137}Cs: 1–3.5 Bq/m^3, ^{90}Sr: 2–14 Bq/m^3) (Ilyin et al. 2011; Matishov et al. 1999). Observations in May 2012 also showed a relatively low ^{137}Cs concentration in water within the STS perimeter (3–10 Bq/m^3) (Ilyin et al. 2015). ^{137}Cs concentrations around 5 Bq/m^3 in Zapadnaya Litsa Bay were observed in 1996 (Matishov and Matishov, 2001). ^{90}Sr concentrations are above background levels due to seasonal water runoff processes (MMBI/ApN, 2015). The volume of runoff generally governs ^{90}Sr levels in the local marine environment, due to its high solubility. The ratio of ^{137}Cs and ^{90}Sr activity in these waters is relatively constant but several times higher than in the deep Andreeva Bay water (Ilyin et al. 2015).

Further afield in the Zapadnaya Litsa and Motovsky Bays, both ^{137}Cs and ^{90}Sr activities are low and do not exceed regional background levels. Runoff from Zapadnaya Litsa Bay influences Motovsky Bay as water from spring runoff is relatively contaminated. When this water enters the coastal zone it forms an area of increased concentrations near the mouth of Zapadnaya Litsa Bay, especially for ^{90}Sr (MMBI/ApN, 2015).

Sediment

Sediment data indicated an increase in ^{137}Cs and ^{90}Sr specific activity compared to the regional background level. The regional background level was estimated from Motovsky Bay sediments: ^{137}Cs at 1–5 Bq/kg dw (average: 2.5 Bq/kg dw); ^{90}Sr at 0.1–2 Bq/kg dw (average: 1 Bq/kg dw) (Ilyin et al. 2011; Matishov et al. 1999). Highest ^{137}Cs and ^{90}Sr concentrations were observed near Pier PMK-67 and in Malaya Andreeva Bay. Radionuclide distributions reflect the release of contaminants from STS sites and sediment depositional processes in the bays (MMBI/ApN, 2015).

^{137}Cs activity concentrations in Andreeva Bay sediments were 30–360 Bq/kg dw, typically lower in the upper littoral zone which dries at low tide. Here, sediments are mainly morainic material (sand and gravel) which has a very low sorption capacity (MMBI/ApN, 2015). Higher ^{137}Cs activity levels are observed in the lower littoral zone where more fine-grained sediments are deposited. The highest ^{137}Cs concentration of 386 Bq/kg dw was observed in sediment from Malaya Andreeva Bay, in an area close to the mouth of the small runoff catchment. This source supplied eroded material during the construction and operation of STS, resulting in elevated ^{137}Cs concentrations. ^{90}Sr activity levels in sediments are much lower than ^{137}Cs, although spatial variation is apparent. A ^{90}Sr concentration of 17 Bq/kg dw was recorded adjacent to Pier PMK-67, where there appears to be evidence of groundwater flow from the STS. A higher concentration (35.8 Bq/kg dw) was registered at the Malaya Andreeva Bay site. The data generally agree with the 2010 data from FMBA (Table 3.5; Sneve et al. 2014).

^{238}Pu and ^{239,240}Pu were present in sediments throughout the littoral zone, generally at low activity concentrations of 1.9–4.7 and 2.5–15.4 Bq/kg dw, respectively, the site adjacent to Pier PMK-67 had the highest contamination level. The ratio of ^{238}Pu/^{239,240}Pu in the sediments was 0.3–0.9 indicating Pu runoff from the STS (MMBI/ApN, 2015).

Activity concentrations several orders of magnitude above background for both ^{137}Cs and ^{90}Sr were observed in a single sediment core collected from Malaya Andreeva Bay. The sample site was close to the HPZ fence, near the outlet of the stream that drains part of the STS territory. The surface layer contained 620 Bq/kg ^{137}Cs, with a maximum of 1050 Bq/kg in the 2–4 cm depth layer. ^{90}Sr activity concentrations were 20.4–40.8 Bq/kg.

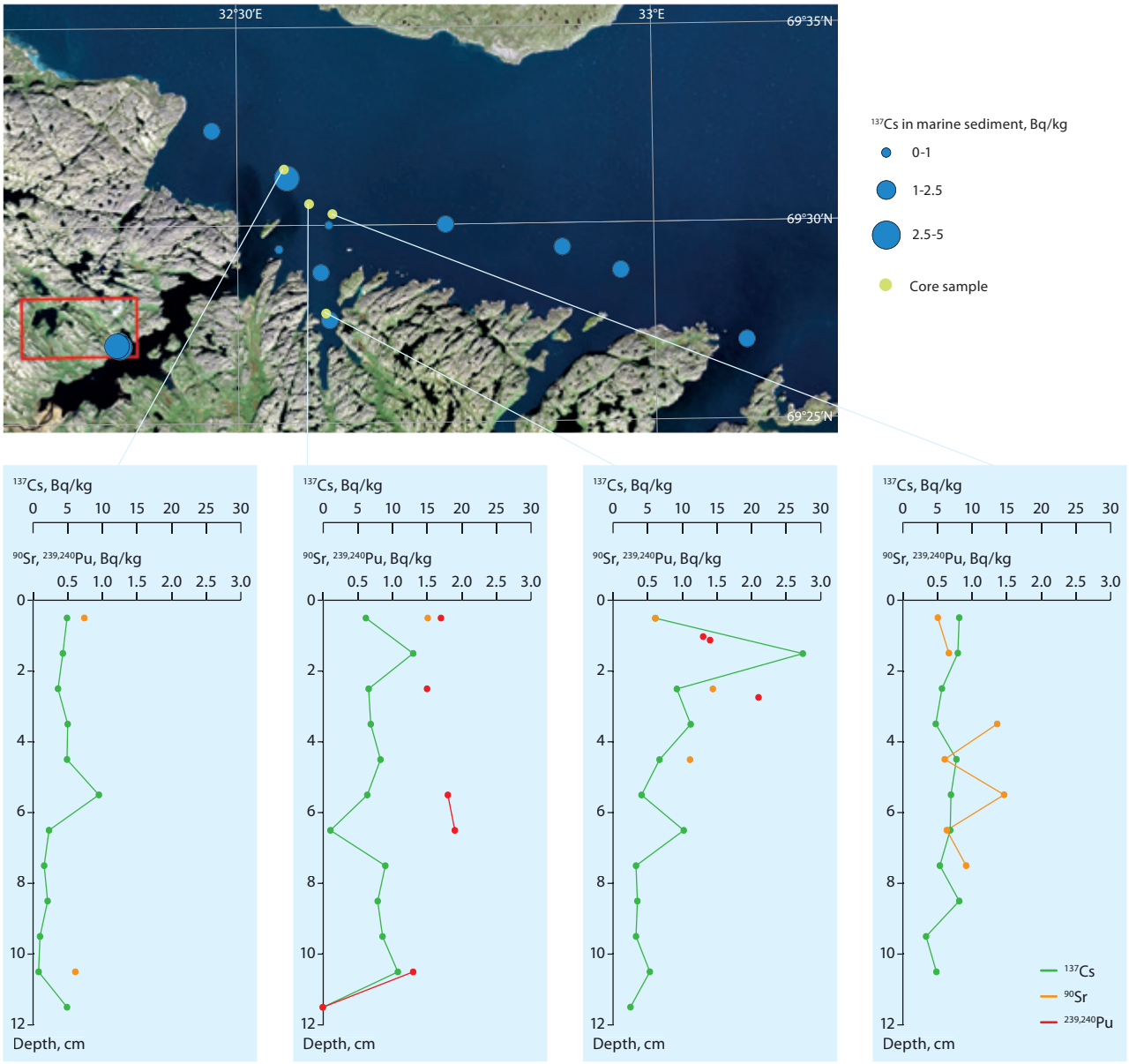

Figure 3.7 Sample locations and ^{137}Cs, ^{90}Sr and $^{239/240}$Pu concentrations in sediments collected from the Motovsky Bay area. The red box corresponds to the area shown in Fig. 3.6.

^{238}Pu and ^{239,240}Pu activity in this sediment core was 1.9–4.2 and 2.6–5.2 Bq/kg, respectively, 3- to 4-fold higher than in other areas of Andreeva Bay and Motovsky Bay. The ^{238}Pu/^{239,240}Pu average ratio was 0.91, again indicative of a local source (MMBI/ApN, 2015).

Further afield (Fig. 3.7), specific activities of ^{137}Cs and ^{90}Sr in Motovsky Bay were considerably lower than in Malaya Andreeva and Andreeva Bays. Measured concentrations were relatively uniform throughout the Motovsky Bay area and consistent with regional background levels (MMBI/ApN, 2015).

^{238}Pu and ^{239,240}Pu in surface sediments (0–1 cm depth) were found at two sample sites near the entrance to Zapadnaya Litsa Bay. Activity concentrations were very low (^{238}Pu: 1.1–1.5; ^{239,240}Pu: 1.7–2.1 Bq/kg dw) with ^{238}Pu/^{239,240}Pu ratios of 0.6–0.7. Pu isotopes in the 1–2 cm depth layer adjacent to Vichana Bay had ^{238}Pu: 1.1; ^{239,240}Pu: 2.1 Bq/kg dw and a ^{238}Pu/^{239,240}Pu ratio of 0.5. A common source of plutonium in the Motovsky Bay mouth and Andreeva Bay is likely due to similar plutonium concentrations and isotope ratios in the two areas. The isotope ratio values do not correspond to the ratio in atmospheric precipitation or in discharges from the Sellafield reprocessing plant (Lindahl et al. 2010; Zaborska et al. 2010). Thus, the STS is the most likely additional source of technogenic radionuclides to sediments in these areas (MMBI/ApN, 2015).

Macroalgae

Macroalgae accumulates radioactive isotopes from seawater and are therefore widely used in studies of radioactive contamination in water bodies (MMBI/ApN, 2015). The accumulation depends on the activity concentration of radionuclides in the local environment, salinity and the length of time the seaweed is exposed to air at low tide. As part of the 2014 study of the radioecological status of the marine environment in the area surrounding Andreeva Bay seaweed samples were collected in two zones: *Fucus vesiculosus* and *Ascophyllum nodosum* from the drying littoral zone (stations 1, 2, 3), and *Laminaria saccharina* from the lower part of the littoral zone which is dry during low tides (stations 4, 5).

Table 3.8 Specific activity of radionuclides in three seaweed species: *Fucus vesiculosus* (Andreeva Bay and Motovsky Bay, October 2014 and August 2013), *Ascophyllum nodosum* (Malaya Andreeva Bay, October 2014) and *Laminaria saccharina* (Andreeva Bay, October 2014).

Species	Site	^{137}Cs, Bq/kg dw	^{90}Sr, Bq/kg dw
Fucus vesiculosus	1	<0.2	1.03±0.06
	2	1.1±0.4	5.45±0.88
	3	9.0±2.7	35.5±1.1
	15	<0.2	<0.4
	16	<0.2	0.4±0.08
Ascophyllum nodosum	3	9.3±1.5	34.62±1.02
Laminaria saccharina	4	12.3±0.02	<MDA
	5	2.5±0.1	<MDA

Table 3.8 presents summary data for ^{137}Cs and ^{90}Sr activity levels in seaweed samples.

Seaweed sampling data from long-term (2005–2014) observations in the Barents Sea coastal areas give an idea of regional background levels. In these studies, ^{137}Cs concentrations in *Fucus* were 0.04–0.84 Bq/kg; ^{90}Sr, about 0.4 Bq/kg in *Fucus* and in Laminariaceae 0.3–2.3 and 0.4–1.5 Bq/kg dw in the coastal zones of the Kola Peninsula (Pechenga Bay, Ura, Teriberkskaya Bay) and Kola Bay (inlets Belokamennaya and Mishukovo, Abram Cape), respectively (Matishov et al. 2014). Compared to background levels, therefore, concentrations in *Fucus* samples (May 2012 and October 2014) indicate chronic contamination of seaweed within the littoral zone of Andreeva and Malaya Andreeva Bays. ^{137}Cs and ^{90}Sr accumulation displays seasonal variability associated with runoff from the STS, but is always higher than in the coastal zone of Motovsky Bay and the Kola Peninsula (MMBI/ApN, 2015).

One sample site yielded enough samples of the filter-feeding Blue mussel (*Mytilus edulis*) to allow meaningful analyses (near station 1, littoral zone at Andreeva Bay, October 2014). The ^{137}Cs and ^{90}Sr activity concentrations in muscle tissue were 9.3±1.5 and 0.1±0.02 Bq/kg dw, respectively. Compared to regional background levels where ^{137}Cs activity in soft tissues and ^{90}Sr activity in mussel shell, regardless of size, are below the MDA in other bays of the Kola Peninsula (e.g. Zelenetskaya, Ura, Belokamennaya Bays) it appears that mussels in the Andreeva Bay littoral zone live in conditions of chronic low radioactive contamination.

3.2.4 Continuing decommissioning work at Andreeva Bay STS

Decommissioning work at Andreeva Bay STS is managed by the Federal Centre for Nuclear and Radiation Safety (FCNRS) [Rosatom]. FCNRS plans to remove the spent fuel assemblies from Andreeva Bay by ship to the Atomflot enterprise near Murmansk, about 80 km to the east, before being transported by purpose-built rail cars to the Mayak PA reprocessing plant. Preparations for removing spent nuclear fuel from the site within an improved regulatory framework are underway to prepare for and complete this task, coordinated by the FCNRS and an international Technical Steering Group. The work is financed by the European Bank for Reconstruction and Development (EBRD) via the Northern Dimension

Box 3.2 Current status of the main decommissioning activities at Andreeva Bay STS

The six spent fuel assemblies removed from the right pool of Building B5 are currently in transport casks ready for intended shipment off-site. High dose rates have been measured above the concrete shielding, such that remotely operated devices will be used to conduct work activities. An estimated 17 tonnes of intermediate-level solid radioactive waste are contained in the ponds (5.2×10^{13} Bq). Urgent repairs to B5 were completed in November 2010, intended to keep the building in an acceptable condition until 2025.

Work on constructing the Dry Storage Unit enclosure (B153), which was expected to be operational in 2016, has encountered problems during site preparation. The discovery of contaminated groundwater, as well as underground concrete structures, has hindered progress. A retrieval machine to support the recovery and repacking of spent fuel assemblies is under manufacture. An on-site transporter to move casks from B153 to B151 is expected to be operational in 2015.

The main structure and engineering services have been completed at the cask accumulation pad (B151) for the special transport casks for spent fuel assemblies. The accumulation pad trolley will be used to transfer the casks to the pier for shipment from Andreeva Bay STS. Acceptance tests for the pier crane are now complete, and the crane is currently being registered and commissioned. The trolley and cranes needed for moving/handling spent fuel assemblies in the Dry Storage Unit are scheduled to be operational in 2015.

Many other smaller decommissioning projects involving other facilities, electrical supply networks, managing radioactive wastes, assessing environmental impacts and improving navigational capabilities for attending ships are also underway. These are important to ensure the timely and safe removal of spent fuel assemblies from the site. Environmental monitoring of the surrounding area shows no evidence of any leakage of radioactive contaminants. Joint Russian-Norwegian projects are expected to continue monitoring Andreeva Bay STS in the upcoming years.

Environmental Partnership (NDEP) Fund. Original plans envisaged that the removal of the remaining spent fuel assemblies would begin in 2016. Given that there is still much construction work to be completed a delay is likely. The status of the main decommissioning projects at Andreeva Bay STS is summarized in Box 3.2 based on material from the EBRD website (www.ebrd.com/home).

The decommissioning work at Andreeva Bay STS is not as advanced as at Gremikha STS but the financial support for this work is in place and the decommissioning plan is in progress. It should be noted however that the two sites, Andreeva Bay and Gremikha, are quite different in terms of the extent and scope of the work required and their accessibility.

Both sites were similar in that initially neither had much functioning infrastructure, particularly for any remedial activities. At Andreeva Bay, the condition of the roughly 21,000 spent fuel assemblies and canisters stored in the Dry Storage Units is uncertain. Therefore, the retrieval technology selection had to allow for a wide range of possible scenarios to safely recover the spent nuclear fuel. Even without problems during retrieval, some six years of continuous operations are required to recover the spent nuclear fuel (www.ebrd.com/home). The equipment will need to be maintained, associated solid and liquid radioactive waste managed and the recovered spent nuclear fuel packed into casks and transported off-site. This requires much new infrastructure, which could not be built until all the derelict buildings/facilities were removed. Without the new infrastructure the program for recovering the spent nuclear fuel using conventional national norms would have been about 25 years and dependent on the weather conditions (www.ebrd.com/home).

For Gremikha however, the problems were much less. They included 898 spent fuel assemblies to be recovered and transported off-site (of which 235 were classed as 'damaged'), a SRW storage pad to be decontaminated and the waste packaged. The spent fuel assemblies were removed during 2010–2011 using conventional Russian technology. Early strategies did consider building hot-cells, cask loading facilities and storage pads but the difficulty of access made this work very expensive and time consuming (www.ebrd.com/home).

The main problem at Gremikha was the Alfa-class submarine reactor cores (nine were at Gremikha). Their spent nuclear fuel could not be reprocessed at Mayak PA and defueling of the cores had not previously been attempted. The current strategy is that reactor cores are removed from the submarines and defueled at Gremikha using available facilities with Russian Federation funding. The spent nuclear fuel was then to be packaged into new modified 'TUK' casks provided under US funding and the spent nuclear fuel sent to Mayak PA to be reprocessed. This effectively meant new facilities were not required at Gremikha with the exception of some improved physical protection and upgrading of cranes and the provision of conventional equipment. All spent nuclear fuel management used conventional handling techniques with the exception of the Alfa core unloading equipment which although it was developed especially for this work did not require significant new infrastructure (www.ebrd.com/home).

3.3 Decommissioning of nuclear submarines

During the Cold War, the former Soviet Union built up the world's largest nuclear submarine fleet, totaling 248 submarines (91 attack submarines, 62 cruise missile submarines, 91 ballistic missile submarines and four research submarines). Many of these were taken out of active service during the 1980s and 1990s. According to Rosatom, a total of 199 Russian navy nuclear submarines had been removed from service by the end of 2013; 120 in north-western Russia and 79 in Russia's Far East.

Decommissioning nuclear submarines has been a priority in the Russian Strategic Master Plan (SMP). The progress highlighted by AMAP in its previous assessment (AMAP 2010b) has continued to date. Russia has three nuclear submarine dismantling facilities: Zvyozdochka in Severodvinsk (Arkhangelsk region), Nerpa in Snezhnogorsk (Murmansk region) and Zvezda in Bolshoi Kamen (Far East). Nerpa has dismantled over 50 submarines since 1998, while Zvyozdochka completed its submarine dismantling program in 2011. Zvezda currently has no submarine dismantlement contracts. According to the Russian news agency RIA Novosti, the Nerpa shipyard was to dismantle the last Soviet-era nuclear submarine to be withdrawn from the Russian Fleet (the Oscar II class cruise-missile submarine *Krasnodar*) by 2014. Removal of spent nuclear fuel was completed in 2013, but during March 2014 a fire occurred during the work to remove the rubber that covers the outer hull of the submarine. Such accidents have occurred before under decommissioning at the Zvyozdochka yard in Severodvinsk (Barents Observer 2014) and pose local radiation risks, because until the reactor compartment is removed there is still a substantial amount of radioactivity inside the submarine. After the rubber covering has been removed, the outer hull is cut off before dividing up the superstructure of the hull. The final stage is to dismantle the reactor compartment and prepare it for long-term storage at the central onshore facility in Saida Bay, just west of the Nerpa shipyard (Grigoriev 2013). As of May 2014, 64 submarine reactor compartments have been positioned in the long-term storage site at Saida Bay, with an additional four three-compartment units placed on raised platforms ready for dismantling.

3.4 The *Lepse* floating technical base

From 1963 to 1981 the vessel *Lepse* was used for unloading and temporary storage of spent nuclear fuel removed from nuclear-powered icebreakers. Since 1981 it has been used as a floating store for damaged spent nuclear fuel, solid and liquid radioactive wastes, and equipment. Decommissioning of the *Lepse* has been a high priority for many years and was part of a 2008 contract between EBRD and Rosatom's Centre for Nuclear and Radiation Safety (AMAP 2010b). After several delays the decommissioning work was scheduled to begin in 2014. The *Lepse* was towed from the Atomflot enterprise in Murmansk to the Nerpa shipyard in 2013. Positioning the *Lepse* on the slipway, ready for decommissioning, was planned for December 2013 but delayed until May 2014. The delay was mostly because the intended decommissioning site was occupied by the first Soviet nuclear submarine K-3 *Leninskiy*

Komsomol. This prevented the construction of the infrastructure required to decommission the *Lepse*. EBRD's strategy was to facilitate required infrastructure improvements at Nerpa and conduct initial dismantling of the superstructure held afloat at Quay #1. The remaining superstructure would then be moved onto a prepared slipway and the work of creating of two Large Storage Packages (LSP) would start; one containing the spent nuclear fuel tanks and one containing radioactive waste. A shelter structure is to be constructed on the slipway where the specialist defueling operations will take place.

3.5 Saida Bay long-term interim storage facility for reactor components

There has been much progress at Saida Bay, a long-term storage facility for nuclear reactor compartments in northwest Russia, since the previous AMAP assessment in 2010. In November 2013, RosRAO signed a contract[2] with the Italian shipbuilder Fincantieri to build a pontoon to lift three-compartment reactor units onto the slipway at Saida Bay. The contract includes the design, manufacture and launch of the pontoon, outfitting and testing. Completion is planned for 2015. Finance was provided by the Ministry of Economic Development of the Italian Republic under an Agreement between the Government of the Russian Federation and the Italian Republic concerning cooperation in the disposal of Russian nuclear submarines retired from the Navy and the safe management of radioactive waste and spent nuclear fuel. As of May 2014, 64 of the 120 one-compartment units (OCUs) afloat at Saida Bay had been positioned on land.

3.6 Radioisotope thermoelectric generators

Radioisotope thermoelectric generators (RTGs) have been used at remote sites in the Arctic and elsewhere as local sources of electricity for facilities such as lighthouses and navigation beacons that are not connected to an electricity net (AMAP 1998, 2002, 2010b).

The radioactive heat source within an RTG core typically incorporated ^{90}Sr. The RTG structure includes layers of shielding material around the ^{90}Sr heat source. The heat sources contained original activities ranging from 0.74 to 14.8 PBq, depending on the type of RTG (Reka et al. 2006; Standring et al. 2007). The most commonly used RTG type, Beta M, contained a single radioactive heat source with an approximate activity of 1.3 PBq. The long half-life of ^{90}Sr (29.1 years) and the total amount of radioactivity involved, means that these sources can pose a radiological hazard for many decades. Direct skin contact with an RTG core may give rise to serious and sometimes life-threatening burns depending on the strength of the source and the time spent in its vicinity. The radiation dose rates at the surface of an unshielded core can reach 10 Sv/h, which has the potential to deliver a lethal dose to humans within half an hour of exposure.

The relatively limited physical protection around RTGs and the lack of maintenance and control made them accessible to intruders. The RTG has been categorized by the International Atomic Energy Agency (IAEA) as one of the radioactive sources with the highest activity and therefore the highest risk (IAEA 2005a). One problem reported for the Beta-M type design is that the components were sometimes screwed together, rather than welded, leaving the RTG more prone to tampering. Several attempted thefts of metal parts from RTGs demonstrate the potential for such radioactive sources to go astray. By removing the RTGs and replacing them by solar cell technology, the danger of loss and contamination of the environment is reduced.

Multilateral cooperation involving Norway, Russia, USA, Sweden, Finland and Canada has contributed to the removal and disposal of RTGs from the Arctic and near Arctic environment, replacing them with solar powered units. Good progress has been made. As of October 2012, 324 RTGs were still present in Russia (with a further four Russian RTGs in the Antarctic) compared to 519 operational in February 2008 (AMAP 2010b). Sixty-eight RTGs were operational in the Arctic region, 56 of which are located at sites along the western and central parts of the Northern Sea Route, including 31 Beta-M RTGs, 4 Gong RTGs, 14 Gorn RTGs, and seven Efir-MA RTGs (Table 3.9).

Damaged RTGs require special measures when decommissioning. Of the 124 RTGs stored at DalRAO in 2012, 11 were listed as damaged (five Beta-M RTGs, five IEU-2 RTGs and one IEU-1 RTG: All were stored in special containers). Furthermore, seven Beta-M RTGs, two Gong RTGs and one Gorn RTG of the 31 RTGs stored at JSC V/O Isotope were listed as 'problematic'. Information regarding the planned disposal of different types of RTG has been summarized by Grigoriev and Katashev (2013).

Decommissioning activities completed in 2012 included the removal of 34 RTGs along the central part of the Northern Sea Route (Yenisey River Estuary), funded by the US and the disassembling of 24 Beta-M RTGs at JSC V/O Isotope. Under the Federal Targeted Program for Nuclear and Radiation Safety, six RTGs were decommissioned along the Northern Sea Route (Tiksi Hydrographic Base). US funding has enabled the decommissioning of four RTGs from the Taymir Peninsular and Wrangel Island. These ten RTGs were transported to JSC V/O Isotope for disassembly. In addition, 24 radioactive heat sources were placed in long-term storage at the Mayak Production Association (Mayak PA; Grigoriev and Katashev 2012).

During 2013, 38 RTGs from the western and central part of the Northern Sea Route (37 from the Taymir Peninsular, Yamal Peninsular and Gydansky Peninsular and one from Wrangel Island) were decommissioned with US funding. In addition, 30 of the 124 RTGs stored at DalRAO (US DoE presentation at Annual CEG meeting, 2013) were transported to final disposal. As of October 2013 the Global Threat Reduction Initiative (GTRI) had recovered 468 RTGs throughout Russia, including 59 RTGs recovered via joint US–Canadian efforts (Grigoriev and Katashev 2013). Other activities in 2013 included the Federal Targeted Program for Nuclear and Radiation Safety that planned to decommission six RTGs from the Northern Sea Route and disassemble 26 of the 70 RTGS stored at Mayak PA (Grigoriev and Katashev 2013) to prepare for long-term storage

[2] http://navaltoday.com/2013/12/09/fincantieri-rosrao-ink-nuclear-submarine-reactor-compartments-contract/

Table 3.9 Status for decommissioning and disposal of radioisotope thermoelectric generators (RTGs) as of 10 October 2012 (adapted from Grigoriev and Katashev 2012).

RTGs operated and stored in Russia	RTGs in operation	RTGs at temporary storage facilities	RTGs being disassembled (disposed of)	RTGs under transport
Western and central parts of the Northern Sea Route	56			9[a]
Eastern part of the Northern Sea Route				1[b]
Far East (Russian Navy)		1		[c]
Special-purpose Missile Forces, Ministry of Defence, Russia	12			
Roshydromet (Antarctic)	4			
At storage facilities				
DalRAO		124		
RosRAO (Irkutsk & Novosibirsk)		22		
Mayak		65		
Isotope			31	
NIIFTA				
Kurchatov Institute				
Total	72	212	31	10 (+ 3)

[a]Special case (one RTG not located at Lishny Island); [b]special case (one RTG not located as a result of beacon destruction at Chukotka); [c]special case (one RTG lost during transportation by helicopter at Sakhalin island).

of the radioactive heat sources. According to information presented at the IAEA CEG meeting in November 2014, only 20 RTGs remained in the field, all in the Far East.

3.7 Consequences of decommissioning for Mayak PA

The success in decommissioning work detailed in earlier sections has decreased the amounts of spent nuclear fuel and radioactive wastes in areas important in terms of the Arctic environment. The spent nuclear fuel removed from sites in northwest Russia is ultimately transported to Mayak PA for handling and storage. No new information is currently available to update the status at Mayak PA as given by AMAP (2010b).

3.8 Nuclear submarine *K-159*

A potential future source of radioactive contamination in Arctic areas not mentioned in earlier AMAP assessments is the sunken *K-159* nuclear submarine. This was a first generation, November class, nuclear attack submarine. On Saturday 30 August 2003, it sank in heavy seas with the loss of nine crew members northwest of Kildin Island in the Barents Sea while under tow from Gremikha to Polyarny for dismantling (Fig. 3.8). One crew member survived the accident. Today, the *K-159* lies at 240 m depth on the seabed within Russian territorial waters, but less than 130 km from the Russian–Norwegian border.

The two 70 MWt nuclear reactors in *K-159* had been shut down since 1989 but still contained approximately 800 kg of spent nuclear fuel which could not be removed at Gremikha STS. An official (IAEA) estimate of the 2003 inventory of the reactors on board *K-159* is 7.4 PBq at the time of sinking.

3.8.1 Monitoring the *K-159* site

Environmental monitoring of the *K-159* site to detect possible leakage of radioactivity was conducted by the Russian Northern Fleet and the Kurchatov Institute shortly after it sank. Monitoring was repeated later in 2003 and in 2004. No leakage appeared to have occurred from either of the two reactors. In 2007, an international expedition took place under the framework of the International Program for Arctic Military Environmental Cooperation (AMEC). Radiation levels were measured above the reactor section and within the external hull of the submarine, and seawater and sediment samples were collected around *K-159*. No indication of any leakage was found. Sonar surveys of *K-159* revealed that the aft end of the hull had snapped off, presumably after the submarine hit the seafloor stern first. The remaining main part of the submarine now lies upright on the seafloor. The *K-159* reactors had not been prepared for dumping at sea. Furthermore, the submarine hit the seafloor with enough force to snap the hull (Fig. 3.9). Hence, there is legitimate concern over the potential for future leakage of radioactivity from the two reactors.

3.8.2 Joint Russian-Norwegian 2014 expedition

A joint Russian-Norwegian expedition in August-September 2014 measured radiation levels around the hull of *K-159* and visually documented the submarine using a remotely operated submersible. In addition, samples of seawater, sediments and fish were collected close to *K-159* and in adjacent areas to study the levels of radioactive contamination in the surrounding environment in more detail. Preliminary results show no evidence of leaks to the marine environment from the nuclear reactors in the sunken nuclear submarine *K-159*, confirming the observations made in a similar expedition in 2007. A final report on the state of the environment surrounding the *K-159* is due shortly.

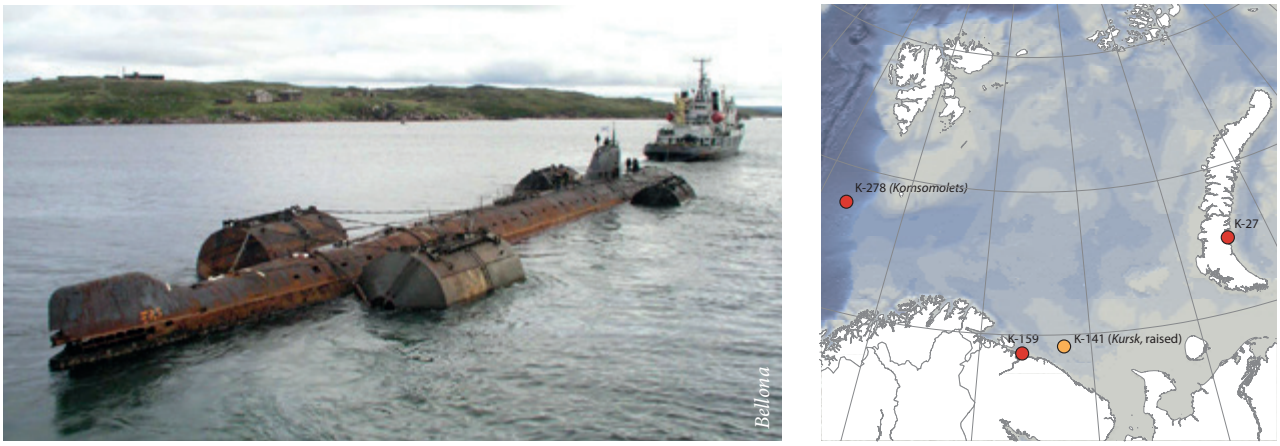

Figure 3.8 Nuclear submarine *K-159* under tow and the location at which it sank. The map shows the location of *K-159* and other sunken nuclear submarines in the Barents region, *K-27* (Sect. 4.5.1), *K-278* (Sect. 2.2.4) and *K-141* (*Kursk,* now raised).

Figure 3.9 Damage to the outer hull of the nuclear submarine *K-159* stern.

3.9 Conclusions

Since the previous AMAP assessment in 2010, there has been good progress in managing and remediating the Gremikha and Andreeva Bay sites of temporary storage, decommissioning nuclear submarines, dealing with radioactive waste and removing radioisotope thermoelectric generators from the environment.

The situation at Gremikha is much improved and nearly all the spent nuclear fuel has now been removed from the site. Although the decommissioning work at Andreeva Bay is not as advanced as at Gremikha, with most of the spent nuclear fuel and radioactive waste at the time of the previous AMAP assessment still on-site, financial support has been allocated by the relevant Russian authorities responsible for this work and the decommissioning plan is in progress. New buildings and infrastructure have been constructed for recovering and handling spent nuclear fuel and managing radioactive waste, and a system of zoning has been adopted for radiation control. Levels of radioactivity at Andreeva Bay (mostly ^{137}Cs and ^{90}Sr) in environmental samples and measured as dose rates have decreased since the previous AMAP assessment. This is due both to radioactive decay and to clean-up activities. Decommissioning operations have not compromised contamination levels off-site and levels in the Supervision Area (a zone with a radius of about 10 km around the site) are at background levels for the region.

Decommissioning nuclear submarines is a priority in the Russian Strategic Master Plan and the progress highlighted by AMAP in its previous assessment has continued to date. While there has been a delay to the start of work on decommissioning the *Lepse*, a floating store for radioactive wastes, there has been much progress at Saida Bay, a long-term storage facility for nuclear reactor compartments. Good progress has also been made on the removal and disposal of RTGs from the Arctic and near Arctic environment; only 20 RTGs remain in the field.

Spent nuclear fuel and radioactive waste removed during decommissioning is ultimately transported to Mayak PA for handling and storage. New information on the status of this site is not available, and the long-term consequences of decommissioning activities in northwestern Russia for Mayak PA and its surrounding environment remain uncertain.

A potential source of radioactive contamination in the Arctic not mentioned in previous AMAP assessments is the sunken *K-159* nuclear submarine. Although there appears to have been no leakage from either of the two reactors on board, there is concern about the potential for future leakage of radioactivity at this site.

4. Monitoring of radioactivity in the Arctic

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Key findings

- *Activity concentrations of radionuclides in the Arctic Ocean are low compared to levels measured between the 1970s and 1990s, and are still decreasing*
- *Monitoring in the freshwater environment reveals the same decreasing trend in activity concentrations of anthropogenic radionuclides as for the marine environment*
- *Data from atmospheric monitoring stations across the Arctic have identified accidental radioactive releases, such as those from the Fukushima Daiichi Nuclear Power Plant accident in Japan, as well as the redistribution of radionuclides from natural events such as forest fires*
- *Continued cooperation between the Arctic nations on radioactivity issues is essential, particularly in relation to increasing understanding of the pathways and uptake of radionuclides in the Arctic environment, identifying changes in these parameters under a changing climate, and decreasing risk potential from known sources*

4.1 Introduction

Monitoring the levels of radionuclides within the Arctic environment is a central part of the AMAP program. Trends in activity concentrations within the various environmental compartments provide information about the transport and distribution in time and space of radionuclides in the environment. Identifying those compartments that act as sinks and as secondary sources for delayed release into the environment when conditions alter is another major goal for the monitoring programs. It is important to maintain up-to date knowledge on the current status of radionuclide contamination in the event of any future additional contamination. Chapter 2 updates information provided by AMAP (2010b) on the main sources of radionuclides in the Arctic while the present chapter addresses the most recent results from national monitoring programs across the Arctic region.

Since the previous AMAP assessment (AMAP 2010b) there has been a major accidental release of radionuclide contaminants to the environment. On 11 March 2011, the Fukushima Daiichi Nuclear Power Plant (FDNPP) in Fukushima, Japan experienced two unprecedented natural disasters resulting in a severe failure of three on-site nuclear reactor units. Fifteen days after the initial releases of radioactive contaminants to the air and sea, traces of radioactivity originating from the Fukushima accident could be detected throughout the northern hemisphere. That small traces of radioactive isotopes released from the accident were detected in the Arctic is an important reminder that the Arctic region is not isolated from the rest of the world and emphasizes the importance of routine radioactivity monitoring within the Arctic. The Fukushima accident and the detection and spread of FDNPP-derived radionuclides within the Arctic region is addressed in detail in Chapter 5.

Monitoring data provide information about the pathways through which radionuclides travel to, within and from the Arctic. They can also be used to calculate the effective ecological half-lives of radionuclides and so provide an understanding of the long-term impacts of radionuclides in the physical environment and food webs. Radionuclides in food webs can be taken up directly from the air or sea. They can also be taken up indirectly, for example through root systems in which case uptake is dependent on factors such as soil type, root depth, and competition with stable elements. This variability in uptake routes leads to different levels of radionuclides in different species and these levels may vary in time and space. The geographical distribution of radioactive contamination and differences in animal diets and metabolism also result in a range of activity concentrations at different trophic levels. This chapter presents monitoring data for the marine environment and associated species, the atmosphere, the terrestrial and freshwater environment and associated species, and humans. This chapter also reports on two case studies from areas with a known increased risk of radioactive contamination. The first is at Stepovogo Fjord on Novaya Zemlya off northern Russia, where numerous objects containing radioactive waste were dumped by the former Soviet Union and later Russia until the early 1990s. The second is Port Radium, a mining area on the eastern shore of Great Bear Lake Northwest Territories, Canada, which has elevated levels of naturally-occurring radionuclides due to the former mining and milling of radium and uranium.

4.2 Marine monitoring

This section on marine monitoring within the Arctic is based on data from specific national monitoring programs, supplemented by data from smaller monitoring programs and efforts where appropriate. Some discrepancy in what is reported by different countries means that for comparison reasons some radionuclide data have been excluded from this report. The Appendix to this chapter provides an overview of the data included in this section.

4.2.1 Canada

4.2.1.1 Marine fish

To address public concern after the Fukushima Daiichi nuclear accident, samples of commonly consumed salmon (coho salmon *Oncorhynchus kisutch* and chum salmon *O. keta*) and groundfish (Pacific spiny dogfish *Squalus suckleyi*, Pacific halibut *Hippoglossus stenolepis* and sablefish *Anoplopoma fimbria*) were harvested from the Canadian west coast in 2013

and analyzed for radioactive cesium. None of the fish samples analyzed contained levels of ^{134}Cs or ^{137}Cs above the detection limit of ~2 Bq/kg (Chen et al. 2015).

Naturally-occurring ^{210}Po was measured in 34 coho salmon, chum salmon and sablefish. Activity concentrations of ^{210}Po in fish fillets varied from below the detection limit (0.2 Bq/kg wet weight, ww) to 3.5 Bq/kg ww. Activity concentrations were below the detection limit in six of 34 samples. Since ^{210}Po is always present in fish in varying concentrations, samples with undetectable levels of ^{210}Po were given a ^{210}Po concentration of half the detection limit (i.e. 0.1 Bq/kg ww). Using this assumption the average ^{210}Po activity concentration in muscle was 0.63 Bq/kg (Chen et al. 2015).

4.2.2 North Atlantic

Environmental radioactivity has been monitored in Greenland and the Faroe Islands since the early 1970s and the data have been reported in previous AMAP assessments (Aarkrog et al. 1997; AMAP 1998, 2000, 2003, 2004) and by Nielsen and Joensen (2009). Marine monitoring has focused on radioactivity in the ocean currents around Greenland as these carry radioactivity from nuclear facilities in Western Europe and Russia, as well as radioactivity in the waters around the Faroe Islands.

Seawater concentrations of ^{137}Cs in the North Atlantic are shown in Fig. 4.1 for 1972–2013. Sampling was undertaken by several partners. Seawater samples from Greenland were provided by the Greenland Institute of Natural Resources and DTU Aqua at the Technical University of Denmark. Seawater samples from the Faroe Islands were provided by the University of the Faroe Islands. The samples were subject to various pre-treatments and analyzed for ^{137}Cs by gamma spectrometry.

In the 1970s, activity concentrations approached and exceeded 10 Bq/m^3 and there was a tendency for Denmark Strait (East Greenland) water to have higher activity concentrations than water off West Greenland and Faroe Island waters. Water masses in the Denmark Strait are under the influence of the North Atlantic Current, and the higher activity levels found west of Greenland are assumed to originate from the Sellafield nuclear fuel reprocessing plant in the UK. Activity concentrations have since declined and are currently in the range 1–3 Bq/m^3. The difference between areas has also decreased.

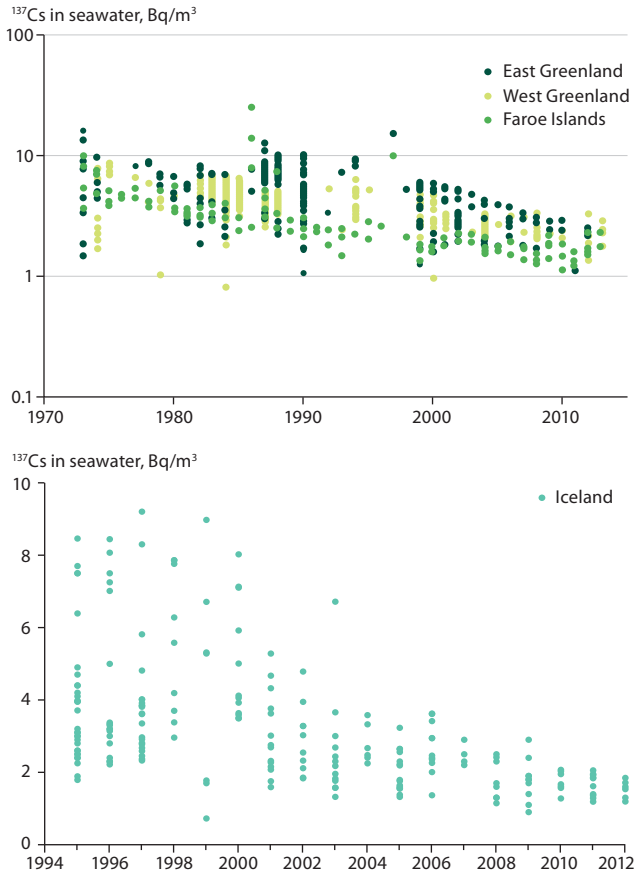

Figure 4.1 Activity concentrations of ^{137}Cs in seawater from the east and west coasts of Greenland and the Faroe Islands during 1970–2013 (upper) and Icelandic waters during 1995–2012 (lower). Note logarithmic scale on upper plot.

4.2.3 Icelandic waters

4.2.3.1 Seawater

The Icelandic Radiation Safety Authority has conducted extensive monitoring of the marine environment in Icelandic waters. Activity concentrations of ^{137}Cs in seawater have declined since the early 1990s and the spread in values has also decreased (Fig. 4.1). Uncertainty for the seawater data is 15%.

4.2.3.2 Marine fish

The Icelandic Radiation Safety Authority has also measured radioactivity in a large number of marine fish species: tusk *Brosme brosme*, Atlantic herring *Clupea harengeus*, lumpfish *Cyclopterus lumpus*, Atlantic cod *Gadus morhua*, Atlantic halibut *Hippoglosus hippoglosus*, common dab *Limanda limanda*, haddock *Melanogrammus aeglefinus*, blue ling *Molva dypterygia*, European plaice *Pleuronectes platessa*, saithe *Pollachius virens*, Greenland halibut *Reinhardtius hippogloides*, Atlantic mackerel *Scomber scombrus*, golden redfish *Sebastes marinus* and beaked redfish *S. mentella*. The data show very low levels of ^{137}Cs activity in fish from Icelandic waters between 1996 and 2012 (Fig. 4.2). Measurements (not shown) of minke whale (*Balaenoptera acutorostrata*) muscle yielded similar results.

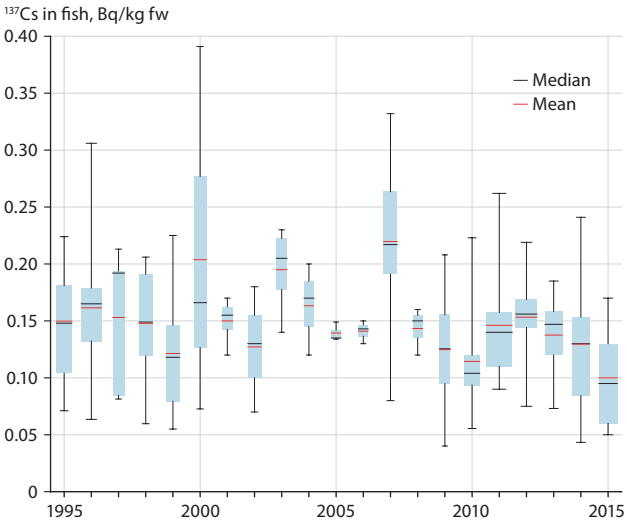

Figure 4.2 Activity concentrations of ^{137}Cs in various fish species between 1996 and 2012 in Icelandic waters. Box widths represent the number of samples.

4.2.4 Barents Sea

Levels of anthropogenic, and to a certain extent also naturally-occurring radionuclides, are regularly monitored in seawater, sediments and biota from the Barents Sea. This section presents data from three initiatives: the regular Norwegian monitoring program, which collects samples every three years; a Norwegian-Russian monitoring program under the Joint Norwegian-Russian Expert Group (JNREG) for investigation of radioactive contamination in the northern areas with annual sampling; and a Finnish-Russian cooperation with published data for 2007 to 2009 (Leppänen et al. 2013).

4.2.4.1 Seawater

The activity levels of ^{137}Cs in seawater from 2006–2013 are shown in Fig. 4.3. The range in the data coincides with the minimum and maximum of the Finnish-Russian data set which is 0.3 and 4.0 Bq/m^3, respectively. The Norwegian (1.0–2.3 Bq/m^3) and the Norwegian-Russian (1.6–2.8 Bq/m^3) data fall within this range. Norwegian samples from 2002 taken at the entrance to the Barents Sea between the mainland and Spitsbergen show a range in concentration of 1.4–4.1 Bq/m^3 which indicates that activity concentrations are slowly decreasing (NRPA 2004b). In 2012, samples from the Norwegian part of the Barents Sea were all within 1–2.2 Bq/m^3 (Skjerdal et al. 2015).

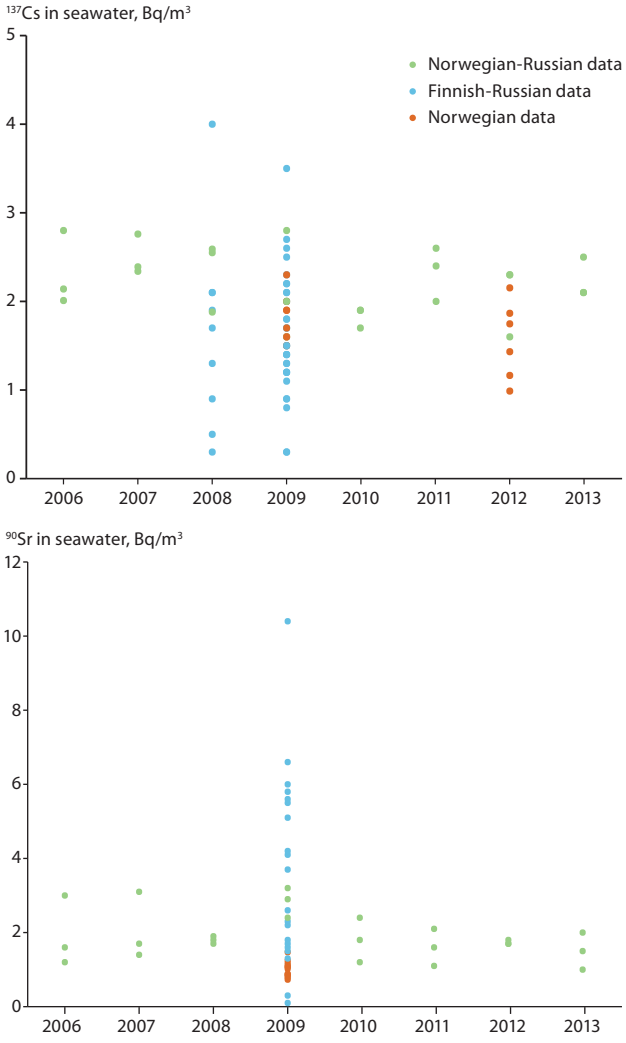

Figure 4.3 Activity concentrations of ^{137}Cs and ^{90}Sr in seawater from the Barents Sea during 2006–2013.

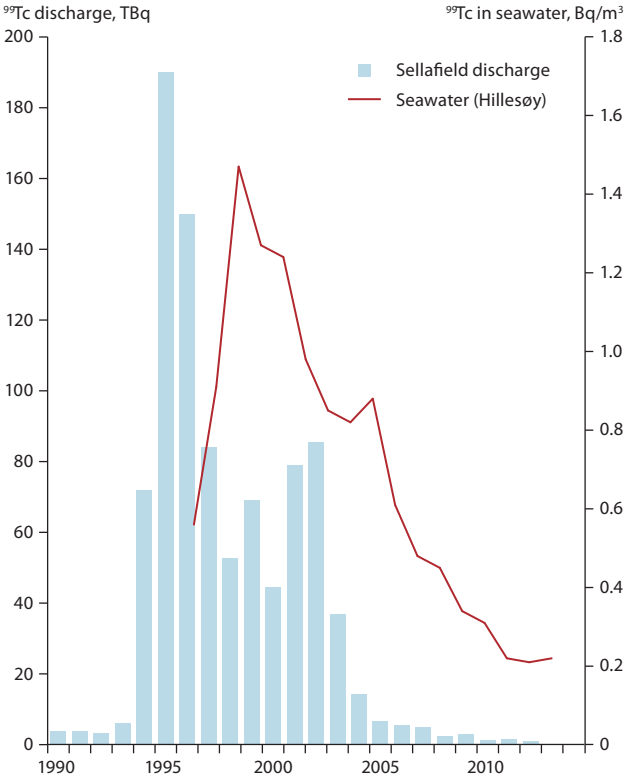

Figure 4.4 Discharge of ^{99}Tc from the Sellafield reprocessing plant on the west coast of the UK (1990–2012) and ^{99}Tc activity concentrations measured in seawater from Hillesøy, northern Norway (1997–2013).

The ^{90}Sr data are also given in Fig. 4.3. As for ^{137}Cs, the range for ^{90}Sr (0.1–10.4 Bq/m^3) coincides with the data from the Finnish-Russian project. However, in this case the Norwegian data (0.78–1.47 Bq/m^3) and the Norwegian-Russian data (1.1–3.2 Bq/m^3) fall within the lower range of concentrations. In 2002, ^{90}Sr levels on the western side of the Barents Sea were within the range 0.82–2.2 Bq/m^3 (NRPA 2004b), which suggests there has been little change in activity over time.

In general, ^{137}Cs and ^{90}Sr activity concentrations for 2006–2013 in the Barents Sea are low in surface seawater and up to an order of magnitude lower than reported for the 1970s, 1980s and 1990s (Holm et al. 1983; Kautsky 1987; Kershaw and Baxter 1995; Kershaw et al. 1997; Herrmann et al. 1998; Kryshev and Sazykina 1995).

Activity concentrations for ^{99}Tc in seawater are monitored by the Norwegian Radiation Protection Authority (NRPA) and are shown in Fig. 4.4, together with discharge data for the Sellafield reprocessing plant (see Ch. 2 for further information on discharges from Sellafield). Based on a study undertaken before ocean currents had transported the much increased ^{99}Tc discharges of the mid-1990s to the Barents Sea, Kershaw et al. (1999) reported ^{99}Tc levels in the Barents Sea for 1994 to be 0.02–0.14 Bq/m^3. Five years later (1999), ^{99}Tc levels measured in seawater from Hillesøy, northern Norway were more than an order of magnitude higher (Fig. 4.4). From a peak in 1999, ^{99}Tc levels are now declining and in 2011–2013 were around 0.2 Bq/m^3: activity concentrations in 2009 (range 0.07–0.16 Bq/m^3) were about half those reported in 2002 (NRPA 2004b). The current low activity concentration of ^{99}Tc at Hillesøy predominantly reflects the reduction in authorized discharges from Sellafield as well as dilution and sedimentation processes (Gwynn et al. 2012).

4.2.4.2 Marine fish

The Northeast Atlantic cod is an important commercial species for the Barents Sea fisheries. Owing to its importance as a resource for human consumption, it is used as an indicator species for contaminant levels in biota, including levels of anthropogenic radionuclides. Figure 4.5 shows activity concentrations of ^{137}Cs in Northeast Atlantic cod from 1976 to 2004 and 2006 to 2013. The data from the first period were from Matishov et al. (2005), while the data for the second period are combined data from the Norwegian monitoring program and the Norwegian-Russian monitoring program. The graphic shows ^{137}Cs activity concentrations in fish from the Barents Sea are currently up to an order of magnitude lower than values reported in the 1980s and 1990s (Matishov et al. 2005).

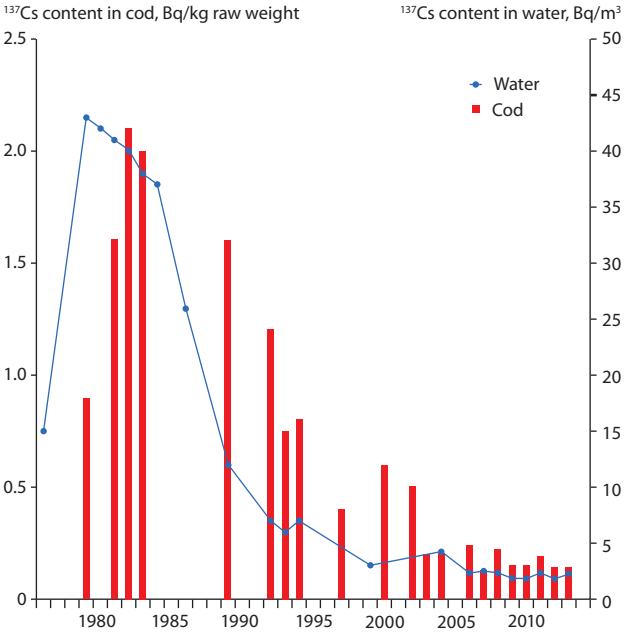

Figure 4.5 Change in ^{137}Cs activity concentrations in seawater and Northeast Atlantic cod (*Gadus morhua*) from the Barents Sea during 1976–2004 (Matishov et al. 2005) and 2006–2013. Source of data: NRPA and JNREG, unpublished.

4.2.4.3 Sediments

The Norwegian-Russian monitoring program (2006 onwards) has a coastal station where surface sediment samples are taken once a year. The samples are analyzed for ^{137}Cs activity. In addition, Leppänen et al. (2013) reported corresponding data for 16 stations in the Barents Sea in 2009. The Norwegian monitoring program (RAME) collects samples in the Barents Sea every third year. This section reports the surface sediment activity concentration of ^{137}Cs reported by RAME for 2009 and 2012. These data are compiled in Fig. 4.6.

The sediment samples taken within the Norwegian-Russian monitoring program are all from a coastal station near Teriberka. The ^{137}Cs activities in these samples are all within the range 0.25–1.60 Bq/kg dry weight (dw) for 2006–2013. As seen in Fig. 4.6, most samples analyzed in the Finnish-Russian study and in the Norwegian monitoring program have higher ^{137}Cs levels. Although this may be due to differences in analytical technique, the most likely explanation is that the data from the Finnish-Russian study and the Norwegian monitoring program

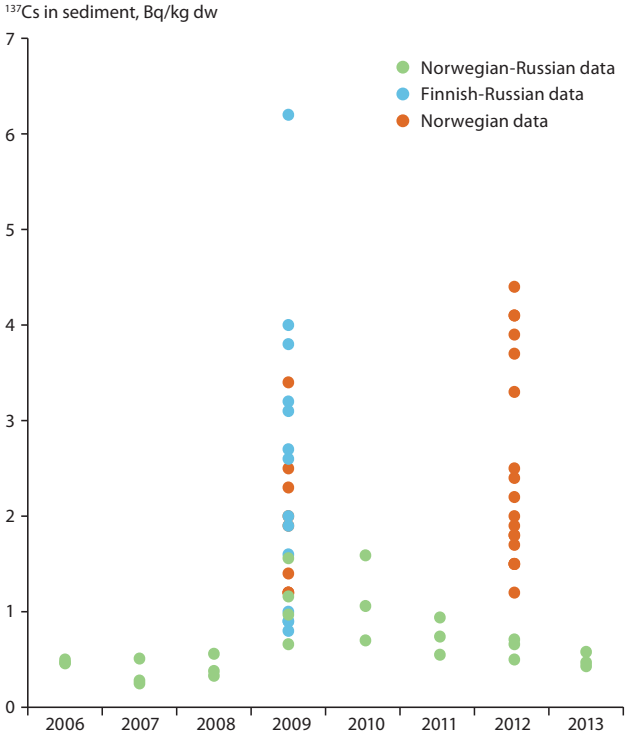

Figure 4.6 Activity concentration for ^{137}Cs in sediment samples (0–2 cm) from the Barents Sea for 2006–2013.

cover the entire Barents Sea. The geographical coverage of the Norwegian data and the Finnish-Russian data is as reported by the NRPA (2011) and Leppänen et al. (2013).

The activity concentrations of ^{137}Cs in sediments in 2009 and 2012 (excluding the Norwegian-Russian data) indicate a very slight decrease since 1991–1993, for which Føyn and Sværen (1997) found an overall mean activity concentration for ^{137}Cs in 102 sediment stations in the Barents Sea of 3.2 Bq/kg dw (range: <1.0–8.6 Bq/kg dw) with the highest values found near Spitsbergen. The stability in ^{137}Cs activity concentrations in surface sediments since the 1990s is probably due to low sedimentation rates in the Barents Sea (e.g. Zaborska et al. 2008) and because downward migration of radionuclides in these sediment profiles is governed primarily by sediment mixing (Smith et al. 1995).

Activity concentrations of plutonium isotopes in sediments have been monitored in a Russian-Finnish cooperation (Fig. 4.7). Activities in the bottom sediments of the open regions of the Barents Sea are low: for ^{239,240}Pu ranging between 0.2 and 3.2 Bq/kg dw. The minimum levels of activity are found in shallow areas characterized by sandy sediments whereas the maximum levels are found in clay silts in deep trenches and depressions. The highest activities occur in Chernaya Bay in Novaya Zemlya where levels vary between 300 and 8500 Bq/kg dw. The observed ^{238}Pu/^{239,240}Pu isotope ratio varied from 0.02 to 0.17 which generally indicates the plutonium to have originated from atmospheric nuclear weapons testing. The higher activities observed at Chernaya Bay at the southern tip of Novaya Zemlya are due to underwater and underground nuclear weapons tests conducted at the tests area at Novaya Zemlya. However, a few cases where the ^{238}Pu/^{239,240}Pu isotope ratio was anomalously high were observed in locations where the activities were generally extremely low (Matishov et al. 2011).

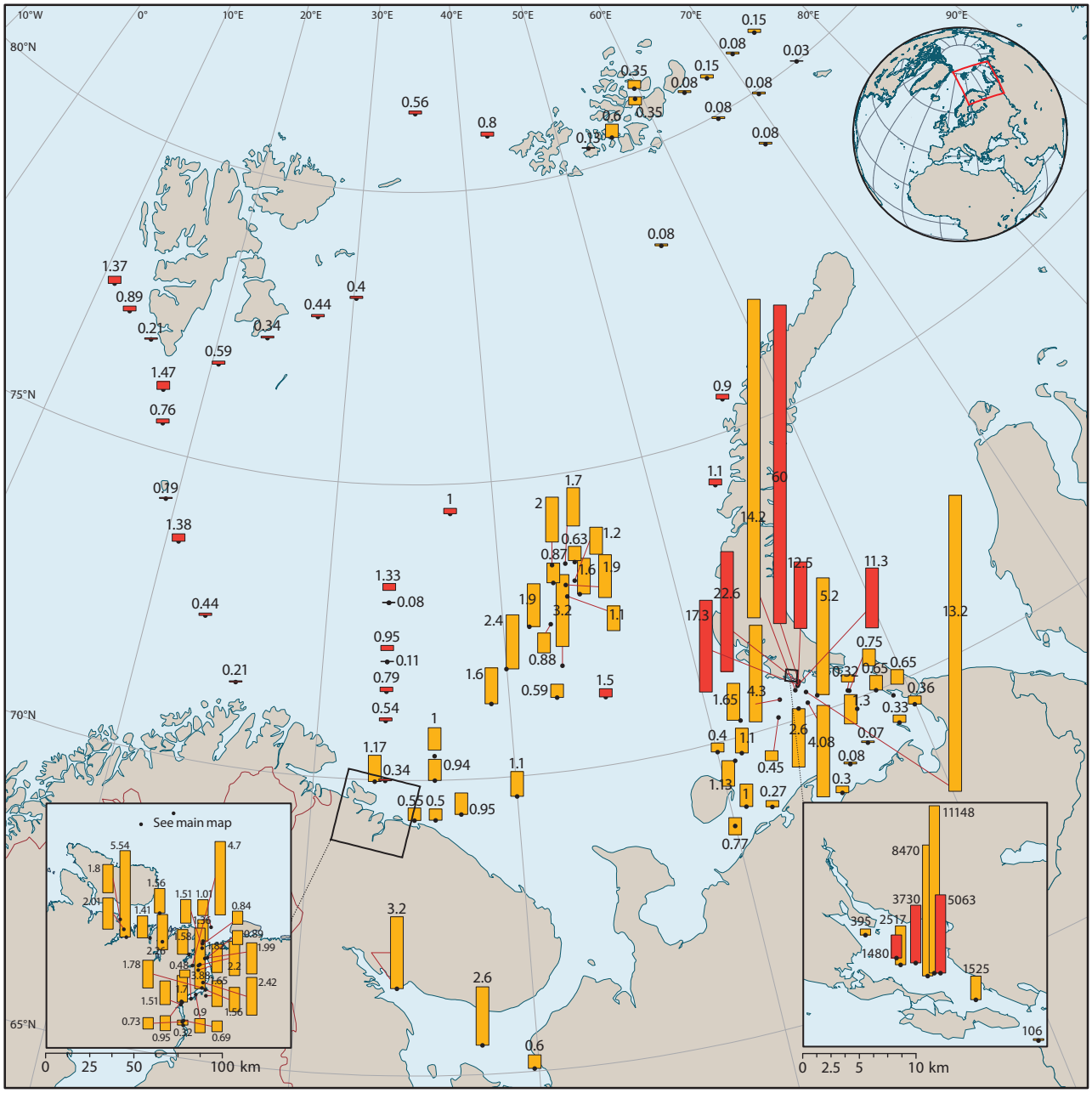

Figure 4.7 Activity concentrations of ^{239,240}Pu in Barents Sea sediment. Sampling stations, activity concentrations of ^{239,240}Pu observed in the 1990s (orange columns), and activity concentrations of ^{239,240}Pu observed in between 2000-2009 (red columns) (Matishov et al. 2011).

4.2.5 Dose and risk assessment for marine biota in the Barents Sea

Considerable effort has been directed over recent years to the development of a system for radiological protection of the non-human biota in the natural environment. This section makes a contribution to that effort with a dose and radiation risk assessment for Barents Sea biota.

The data available for this risk assessment originate from the Joint Norwegian-Russian monitoring program on the study of radioactive contamination of the Barents Sea. Abiotic and biotic samples are collected on an annual basis from a site located near Teriberka village on the Russian Barents Sea coast. Activity concentrations for technogenic radionuclides (^{137}Cs, ^{90}Sr, ^{239,240}Pu) in water, bottom sediments and marine biota (fish, aquatic plants, mollusks) are available for 2006–2012.

Dose calculations for the selected reference groups of organisms were performed using the international ERICA Assessment Tool (Brown et al. 2008). Measurement data for the activity concentrations of ^{137}Cs, ^{90}Sr and ^{239,240}Pu in water, bottom sediments and aquatic organisms were used as input data for dose calculations with ERICA. Average dose rates to representatives of the reference groups of marine biota were calculated for each year of the seven-year period (from 2006 to 2012).

The results of the dose rates calculations are given in Table 4.1. This shows the contributions made by ^{137}Cs, ^{90}Sr and ^{239,240}Pu to average dose rates, as well as to external, internal and total dose rates. For benthic fish, external dose rates to bottom-dwelling fish were higher than dose rates from radionuclides incorporated in fish tissue, and the most important contributor

Table 4.1 Average dose rates to selected reference groups of organisms from the Barents Sea in the vicinity of Teriberka village on the Russian coast.

Year	Dose rate, mGy/day					
	^{137}Cs	^{90}Sr	^{239,240}Pu	External dose	Internal dose	Total dose
Benthic fish						
2006	2.5×10^{-6}	8.4×10^{-10}	$<10^{-10}$	1.7×10^{-6}	7.4×10^{-7}	2.5×10^{-6}
2007	2.8×10^{-6}	1.6×10^{-9}	$<10^{-10}$	2.1×10^{-6}	6.2×10^{-7}	2.8×10^{-6}
2008	2.4×10^{-6}	1.0×10^{-6}	8.6×10^{-9}	1.6×10^{-6}	1.8×10^{-6}	3.4×10^{-6}
2009	4.5×10^{-6}	1.6×10^{-7}	$<10^{-10}$	3.8×10^{-6}	8.5×10^{-7}	4.7×10^{-6}
2010	4.8×10^{-6}	2.3×10^{-7}	$<10^{-10}$	4.1×10^{-6}	9.8×10^{-7}	5.1×10^{-6}
2011	3.4×10^{-6}	1.2×10^{-7}	$<10^{-10}$	2.6×10^{-6}	8.6×10^{-7}	3.5×10^{-6}
2012	2.9×10^{-6}	3.4×10^{-7}	2.2×10^{-9}	2.3×10^{-6}	9.6×10^{-7}	3.2×10^{-6}
Pelagic fish						
2006	7.1×10^{-7}	1.9×10^{-6}	8.6×10^{-10}	1.8×10^{-8}	2.6×10^{-6}	2.7×10^{-6}
2007	7.1×10^{-7}	4.6×10^{-7}	1.1×10^{-9}	1.8×10^{-8}	1.1×10^{-6}	1.2×10^{-6}
2008	7.8×10^{-7}	6.0×10^{-7}	4.3×10^{-10}	1.3×10^{-8}	1.4×10^{-6}	1.4×10^{-6}
2009	7.6×10^{-7}	2.9×10^{-7}	5.0×10^{-10}	1.4×10^{-8}	1.0×10^{-6}	1.1×10^{-6}
2010	8.3×10^{-7}	4.6×10^{-7}	6.5×10^{-9}	1.4×10^{-8}	1.3×10^{-6}	1.3×10^{-6}
2011	7.1×10^{-7}	2.9×10^{-7}	3.4×10^{-9}	1.2×10^{-8}	9.9×10^{-7}	1.0×10^{-6}
2012	4.9×10^{-7}	2.9×10^{-7}	6.0×10^{-9}	1.3×10^{-8}	7.7×10^{-7}	7.9×10^{-7}
Aquatic plants						
2006	2.4×10^{-6}	6.5×10^{-7}	8.4×10^{-9}	2.0×10^{-6}	1.1×10^{-6}	3.0×10^{-6}
2007	2.6×10^{-6}	8.7×10^{-7}	6.0×10^{-9}	2.4×10^{-6}	1.1×10^{-6}	3.5×10^{-6}
2008	2.2×10^{-6}	6.5×10^{-7}	2.9×10^{-9}	1.9×10^{-6}	9.6×10^{-7}	2.8×10^{-6}
2009	4.8×10^{-6}	3.4×10^{-7}	5.8×10^{-9}	4.6×10^{-6}	5.8×10^{-7}	5.2×10^{-6}
2010	5.4×10^{-6}	2.2×10^{-7}	6.5×10^{-9}	4.6×10^{-6}	1.0×10^{-6}	5.6×10^{-6}
2011	3.5×10^{-6}	1.1×10^{-7}	3.4×10^{-9}	3.1×10^{-6}	4.5×10^{-7}	3.6×10^{-6}
2012	3.1×10^{-6}	1.1×10^{-6}	6.0×10^{-9}	2.6×10^{-6}	1.5×10^{-6}	4.1×10^{-6}
Mollusks (bivalves)						
2006	2.1×10^{-6}	5.5×10^{-7}	$<10^{-10}$	1.9×10^{-6}	8.4×10^{-7}	2.7×10^{-6}
2007	2.4×10^{-6}	5.5×10^{-7}	6.7×10^{-9}	2.3×10^{-6}	6.7×10^{-7}	3.0×10^{-6}
2008	2.1×10^{-6}	2.9×10^{-7}	4.6×10^{-9}	1.7×10^{-6}	6.3×10^{-7}	2.4×10^{-6}
2009	4.3×10^{-6}	7.0×10^{-7}	7.4×10^{-9}	4.1×10^{-6}	9.7×10^{-7}	5.0×10^{-6}
2010	4.4×10^{-6}	5.5×10^{-7}	4.3×10^{-9}	4.3×10^{-6}	6.6×10^{-7}	5.0×10^{-6}
2011	2.3×10^{-6}	1.4×10^{-7}	$<10^{-10}$	2.9×10^{-6}	2.5×10^{-7}	3.1×10^{-6}
2012	2.7×10^{-6}	1.4×10^{-7}	$<10^{-10}$	2.4×10^{-6}	4.3×10^{-7}	2.8×10^{-6}

to the total dose was ^{137}Cs. In contrast to the situation for benthic species, external dose rates to pelagic fish were much lower than from radionuclides incorporated in fish organs and tissues. The contributions of ^{137}Cs and ^{90}Sr to exposure of pelagic fish were comparable. External dose rates to aquatic plants and mollusks were higher than internal dose rates and ^{137}Cs was the major contributor to exposure.

The calculated dose rates for the Barents Sea biota were compared with the derived consideration reference levels (DCRLs), defined by the International Commission on Radiological Protection (ICRP) for the reference organisms. The radiation risks for the *i*-th ecological group (R_i) of the Barents Sea biota were estimated as:

$$R_i = \frac{D_i}{DCRL_i} \qquad \text{Eq. 4.1}$$

where D_i is the total dose rate, mGy/day; and $DCRL_i$ is the lower bound of the preliminary DCRL interval for *i*-th reference organism (see ICRP 2008: tables 6.1–6.4).

Table 4.2 Average dose rates to the reference ecological groups of the Barents Sea biota near the Russian coast in 2006–2012 and estimated radiation risks for biota.

Reference group of biota	Dose rate, mGy/day	DCRL, mGy/day	Estimated radiation risk for biota
Benthic fish	$(3.6 \pm 1.0)\times10^{-6}$	1	$(3.6 \pm 1.0)\times10^{-6}$
Pelagic fish	$(1.3 \pm 0.6)\times10^{-6}$	1	$(1.3 \pm 0.6)\times10^{-6}$
Aquatic plants	$(4.0 \pm 1.1)\times10^{-6}$	10	$(4.0 \pm 1.1)\times10^{-7}$
Mollusks (bivalves)	$(3.4 \pm 1.1)\times10^{-6}$	10	$(3.4 \pm 1.1)\times10^{-7}$

Table 4.2 shows the average dose rates to the reference ecological groups of the Barents Sea biota in 2006–2012, DCRL values for these organisms (ICRP 2009) and estimated radiation risks for biota. These data suggest that radiation risks from ^{137}Cs, ^{90}Sr and ^{239,240}Pu to Barents Sea biota near the Russian coast (Teriberka village), calculated on the basis of data from the Joint Russian-Norwegian monitoring program, are negligible, with dose rates six to seven orders of magnitude lower than the DCRL.

4.3 Atmospheric monitoring

Radionuclides discharged to air are often rapidly dispersed by air flow and by using a network of air samplers and modeling tools, it is possible to backtrack the plume to the discharge point. Most Arctic nations (except Greenland and the Faroe Islands) have air monitoring stations and this section summarizes the main findings since the previous AMAP assessment (AMAP 2010b).

4.3.1 Canada

Since 1959, the Radiation Protection Bureau of Health Canada has continuously operated the Canadian Radiological Monitoring Network (CRMN) which comprises environmental sampling stations located across the country to routinely monitor radioactivity levels in the environment. The original purpose of the network was to measure radioactive fallout from the intensive nuclear weapons testing of the Cold War era. Since the 1986 Chernobyl Nuclear Power Plant accident, radiation surveillance in Canada has been significantly enhanced with the expansion of the CRMN, Canada's participation in and access to, the International Monitoring System of the Comprehensive Nuclear-Test-Ban Treaty Organization (CTBTO), the creation of a nation-wide, real-time Fixed Point Surveillance Network (FPSN) for external dose monitoring, and by the integration of Environment Canada's atmospheric transport modelling capability into Health Canada's operation. Environmental surveillance activities from the 26 monitoring stations of the CRMN, the 76 monitoring stations of the FPSN, and the 80 globally distributed monitoring stations of the CTBTO (including four situated in Canada) provide sensitive and comprehensive coverage for radionuclide detection and impact assessment in Canada. Among these monitoring stations, eight are located in the AMAP region. These stations are located in Whitehorse (60.734°N, 135.099°W), Yellowknife (62.734°N, 114.469°W), Coral Harbour (64.189°N, 83.347°W), Inuvik (68.318°N, 133.534°W), Resolute (74.705°N, 94.969°W), Alert (82.499°N, 62.342°W), Churchill (58.769°N, 94.169°W) and Kuujjuarapik (55.275°N, 77.758°W).

Airborne radioactivity monitoring data are freely accessible on the Government of Canada's website[3]. The data available are activity concentrations (Bq/m^3) with the associated measurement uncertainty and the minimum detectable concentration for the naturally-occurring radionuclides, ^{7}Be and ^{210}Pb, and the anthropogenic radionuclides, ^{134}Cs, ^{137}Cs, and ^{131}I. The data are from the analysis of particulates accumulated in filter media, drawn by high-volume air samplers fixed in the field and are typically dominated by natural radionuclides, such as ^{7}Be and ^{210}Pb.

All the monitoring stations show a small increase in airborne ^{134}Cs, ^{137}Cs and ^{131}I activity measured between March and May 2011, attributable to the nuclear accident at the FDNPP. However, it is important to note that, even at their respective peaks, the measured activity concentrations of ^{134}Cs, ^{137}Cs and ^{131}I represent only a small fraction of typical background exposure from natural sources of radiation. Occasionally, other small increases in activity concentration of anthropogenic radionuclides are observed. Spikes in ^{137}Cs activity are often associated with forest fires, which can lead to the re-suspension of ^{137}Cs already present in the environment, most likely from atmospheric nuclear testing in the 1960s.

Figure 4.8 shows airborne radioactivity measurements for the eight stations located within the AMAP region. Data are collected daily at the Yellowknife and Resolute stations and so these plots show more data points. The air sampler at Coral Harbour was not operational at the time of the Fukushima accident which explains the break in the data. All the other stations show a small increase in ^{134}Cs, ^{137}Cs and ^{131}I following the Fukushima accident. These results are similar to those for stations located further south.

Figure 4.9 shows the ^{134}Cs:^{137}Cs ratio derived from particulate monitoring data at five stations around the time of the Fukushima accident. The ratio was typically 1:1, which is characteristic of the relative quantities known to have been released into the air from the FDNPP accident. Notably, ^{134}Cs and ^{137}Cs were also detected at the Yellowknife station in April 2012. This was due to particulate re-suspension after the Fukushima accident, probably due to forest fire activity.

[3] http://data.gc.ca/data/en/dataset/21b821cf-0f1c-40ee-8925-eab12d357668

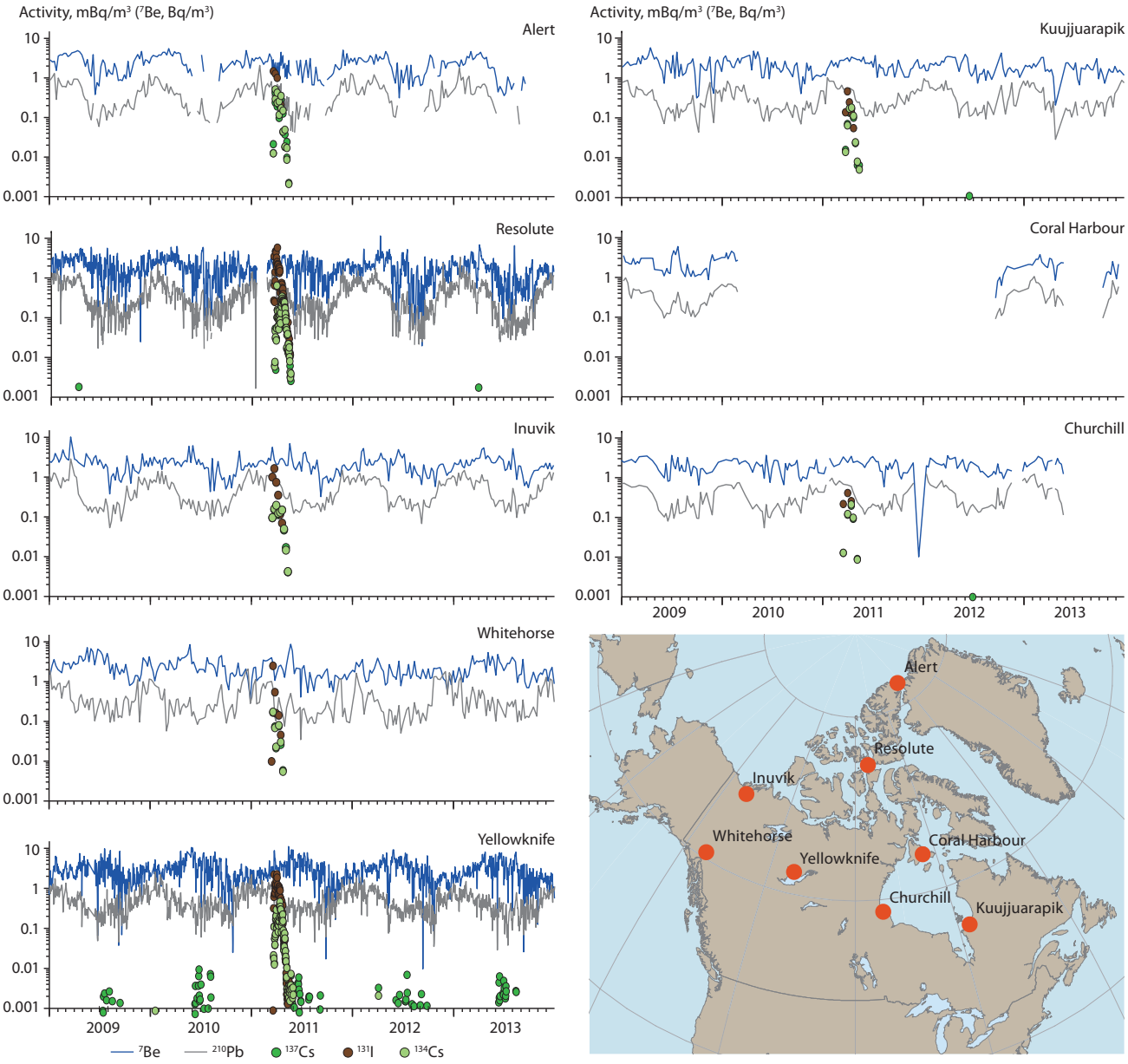

Figure 4.8 Airborne radioactivity measurements at the eight Canadian air monitoring stations located within the AMAP region.

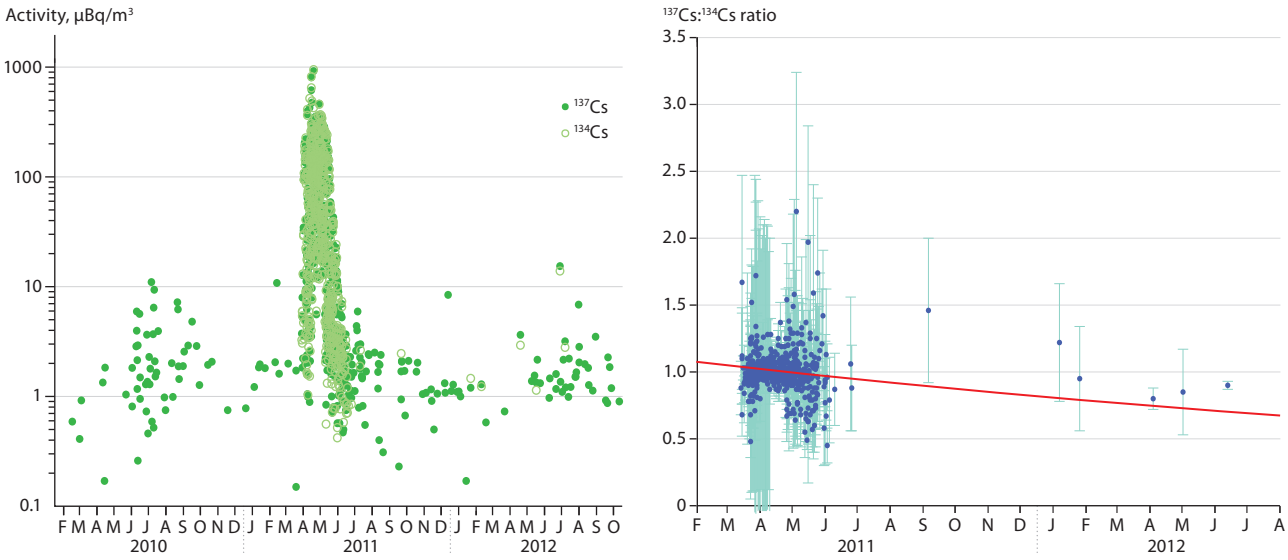

Figure 4.9 Activity concentrations and ratios of ^{134}Cs:^{137}Cs obtained from the air particulate monitoring data obtained at five Canadian air monitoring stations (Alert, Inuvik, Resolute, Whitehorse and Yellowknife) around the time of the Fukushima Daiichi nuclear power plant accident.

4.3.2 Iceland

The Icelandic Radiation Safety Authority (IRSA) has five monitoring stations that measure ambient dose rate at four sites in Iceland: Reykjavik, Bolungarvik, Raufarhöfn and Höfn. Typical values are 50 nSv/h. The stations are set to alert IRSA staff if the level rises above 80 nSv/h. The measurements are accessible via the IRSA website (www.gr.is/verkefni/gammageislun).

One automated CTBTO high-volume aerosol sampler is operated in Reykjavik, collecting daily samples. Automated, and reviewed, analyses of the measurements are provided via CTBTO, and are monitored by the Icelandic Radiation Safety Authority.

Precipitation from two sites in southwestern Iceland is continuously monitored for ^{137}Cs. The activity concentration of ^{137}Cs in these samples has always been below detection limits.

4.3.3 Norway

The NRPA has ten ambient dose rate monitoring stations in the AMAP region, and data are automatically transferred to its head office near Oslo once an hour. If high activity concentrations are detected, relevant personnel are alerted automatically. This network was established in the years following the Chernobyl accident in 1986 and was upgraded in 2006. In addition to the ten ambient dose rate monitoring stations, the NRPA has three aerosol samplers in the AMAP region: one in Svanhovd (69°28'N, 30°03'E), one at Viksjöfjell (69°36'N, 30°44'E) and one in Skibotn (69°22'N, 20°17'E).

As an example, the data for 2013 and 2014 from Svanhovd are shown in Fig. 4.10. In most weeks, ^{137}Cs activity is around the sample-specific minimum detectable activity (MDA), which is a general trend for both the area and between years (see Møller and Dyve 2011 among others). However, in week 15 in 2013, a peak in ^{137}Cs activity was detected at all three air monitoring stations in northern Norway. The most likely cause was the accidental meltdown of a cesium-containing object along with scrap metal at the smelting plant in Elektrostal, 50 km east of Moscow.

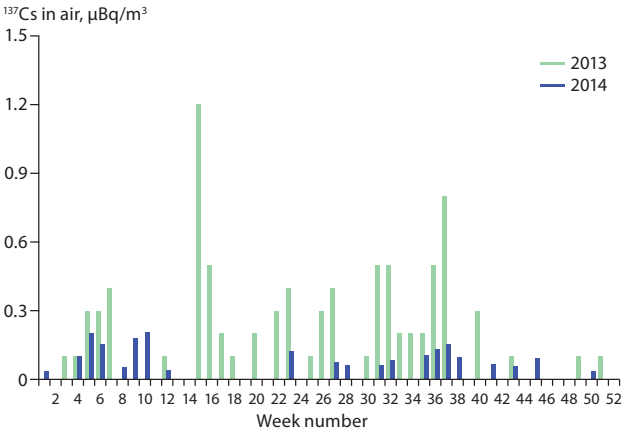

Figure 4.10 Activity concentrations of ^{137}Cs in air filters at the Svanhovd Station in 2013 and 2014. No bars: less than the sample-specific minimum detectable activity at the 95% confidence level (MDA 95%) (Møller and Dyve 2014; Møller et al. 2015).

4.3.4 Finland

The Radiation and Nuclear Safety Authority (STUK) monitors environmental radioactivity in Finnish Lapland. There are 35 stations monitoring the ambient dose rate. These stations collect dose rate information at 10-minute intervals. Two of the 35 stations are also equipped with an $LaBr_3$ spectrometer which measures the full spectrum of ambient radiation also at 10-minute intervals. In addition to ambient dose rate monitors there are three air filter stations: at Rovaniemi (66.51°N, 25.73°E), Sodankylä (67.37°N, 27.57°E) and Ivalo (68.64°N, 27.57°E). Airborne radioactivity and deposition are monitored at these sampling stations in Finnish Lapland.

Monitoring at the Finnish stations showed a steady decline in atmospheric fallout of ^{137}Cs from the early 1960s until the Chernobyl accident in 1986, which resulted in a sudden and substantial increase in airborne ^{137}Cs. These high levels declined over the following decade and have since been fairly stable. The FDNPP accident caused a short-term peak in atmospheric ^{137}Cs activity in March–May, 2011. After spring 2011, ^{137}Cs activity concentrations returned to the levels measured before the accident. ^{137}Cs activity in the atmosphere is currently at trace levels, generally below 1 µBq/m^3 (Fig. 4.11).

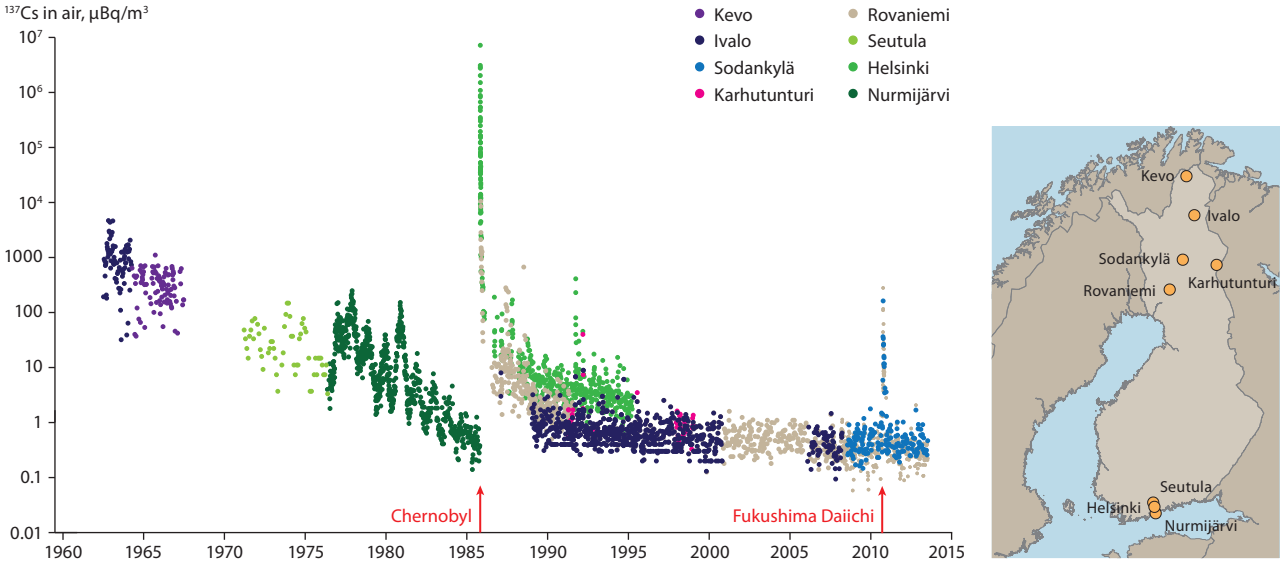

Figure 4.11 Activity concentrations of ^{137}Cs in ground level air at various sites across Finland since the early 1960s.

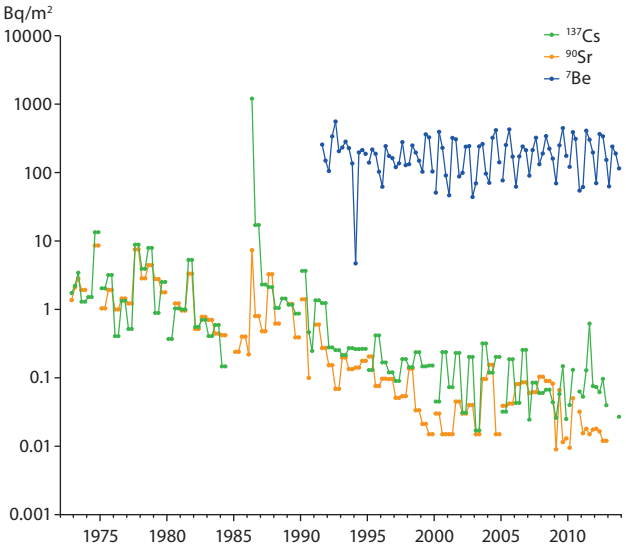

Figure 4.12 Time series of ^{90}Sr, ^{137}Cs and ^{7}Be measured in fallout at Rovaniemi, Finland from 1972 to 2013.

Fallout (wet and dry deposition) of ^{137}Cs has declined steadily since the Chernobyl accident. Annual deposition over the past 20 years has been below 1 Bq/m^2. Fallout from the FDNPP accident in 2011 caused a minor peak in ^{137}Cs fallout of about 1 Bq/m^2 which can be seen from Fig. 4.12.

It should be noted that the current level of ^{137}Cs fallout is significantly less than the fallout of natural radioactive isotopes. For comparison, the naturally-occurring isotope of ^{7}Be is also plotted on Fig. 4.12. The fallout samples are collected at three-month intervals and so allow all naturally-occurring radon daughters with short half-lives to decay away. Generally, ^{7}Be is the most active trace isotope in the fallout samples.

4.3.5 Russia

Monitoring of environmental radioactive contamination is conducted by subdivisions of the Federal Service of Russia on Hydrometeorology and Environmental Monitoring located north of the Polar circle. The measurements include daily monitoring of the gamma-radiation dose rate, volume activities in the surface atmospheric layer, and deposition of radioactive substances from the atmosphere.

Table 4.3 shows mean annual volume activities of ^{90}Sr and ^{137}Cs in the surface atmospheric layer and total atmospheric deposition of ^{137}Cs averaged for the Russian Arctic region. The elevated ^{137}Cs volume activity in 2011 (compared to previous years) is due to air masses contaminated by FDNPP-derived radioactive products. Table 4.4 shows the change in atmospheric levels of FDNPP-derived ^{137}Cs activity across the Russian Arctic through 2011.

4.4 Terrestrial monitoring

4.4.1 Freshwater

Freshwaters are monitored in Finland and Russia and it is mainly levels of anthropogenic radioisotopes that are measured. In Finland, samples have been collected from river and lake ecosystems (Fig. 4.13), while in Russia the main rivers flowing into the Arctic seas are sampled (Table 4.5). The data indicate a decline in radioactive contamination after the Chernobyl accident in freshwater systems, with activity concentrations of

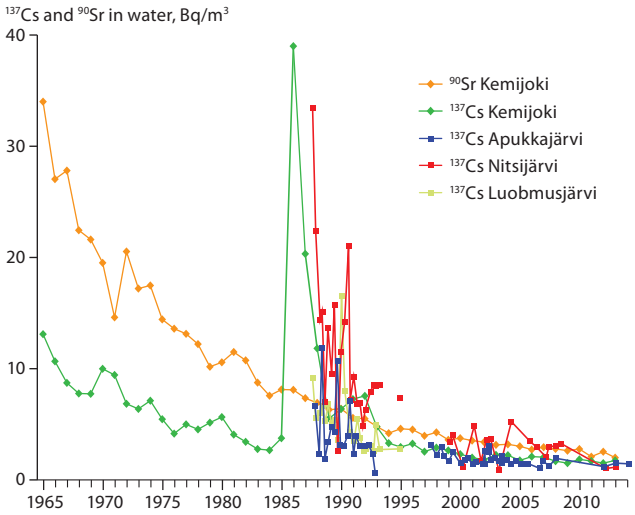

Figure 4.13 Activity concentrations of ^{90}Sr and ^{137}Cs in the Kemijoki river estuary near its entry to the Baltic Sea at the northern end of the Gulf of Bothnia and of ^{137}Cs in three lakes (Luobmusjärvi, Apukkajärvi and Nitsijärvi) in northern Finland.

Table 4.3 Mean annual volume activity concentrations of radionuclides in the surface atmospheric layer and total atmospheric deposition, averaged across the Russian Arctic (Roshydromet 2008, 2009, 2010, 2011, 2012, 2013).

Isotope	2006	2007	2008	2009	2010	2011	2012
Volume activity							
Σβ (10^{-5} Bq/m^3)	7.7	7.2	8.6	9.2	8.0	6.6	8.2
^{90}Sr (10^{-7} Bq/m^3)	0.27	0.3	0.27	0.29	0.25	0.29	0.33
^{137}Cs (10^{-7} Bq/m^3)	1.0	1.2	0.9	1.1	0.9	21.2	1.6
Deposition							
^{137}Cs (Bq/m^2/y)	<0.4	<0.4	<0.2	<0.2	<0.2	2.03	0.2

Table 4.4 Mean monthly volume activity concentrations of ^{137}Cs (10^{-7} Bq/m^3) in the surface atmospheric layer through 2011, averaged across the Russian Arctic.

January	February	March	April	May	June	July	August	September	October	November	December
5.1	5.1	5.4	139.4	50.1	42.7	1.7	1.2	1.2	1.0	1.0	1.0

Table 4.5 Volume activity of ^{90}Sr and ^{3}H in several Russian rivers.

River	2006	2007	2008	2009	2010	2011	2012
^{90}Sr, Bq/m^3							
Severnaya Dvina	-	-	-	-	5.2	-	5.1
Pechora	-	-	-	-	2.7	-	2.6
Ob	-	-	-	-	-	6.5	7.8
Yenisey	-	-	-	-	-	-	5.9
Lena	-	-	-	-	-	-	3.7
^{3}H, kBq/m^3							
Severnaya Dvina	2.5	2.2	2.3	1.6	1.6	1.6	1.3
Pechora	2.9	2.5	2.1	2.1	1.9	1.7	2.3
Ob	2.6	2.6	3.2	2.2	2.2	2.0	2.9
Yenisey	2.8	2.8	2.6	2.3	2.7	2.3	2.6
Lena	3.0	2.6	2.7	2.9	2.8	1.7	3.4

^{137}Cs currently 10 to 30 times lower than immediately after the accident. The Russian time series for ^{90}Sr and ^{3}H only extends back to 2006 and the steady trend reflects either a continuous contribution of these radionuclides to the environment or the later decay of former releases.

4.4.2 Freshwater fish

4.4.2.1 Cesium-137

Freshwater fish form part of a traditional diet and so represent a pathway for the transfer of ^{137}Cs to man. Some species are also popular for sports fishing. Although there are differences between species, all measurements reported are far below the limit set by the authorities for ^{137}Cs in fish for commercial retail (Norway: 3000 Bq/kg; EU (Finland, Denmark): 600 Bq/kg). A comparison of ^{137}Cs activity concentrations over time in fish from three lakes in Scandinavia (Fig. 4.14) shows predatory species such as pike and perch to have higher concentrations than omnivorous species. Concentration also varies with size in Arctic char from Lake Fjellfroskvatn in northern Norway (Fig. 4.15), with larger, and thus older, fish having higher levels of ^{137}Cs. This could be due to a shift in diet, and thereby trophic level, as the fish age, or could reflect a longer biological half-life for ^{137}Cs in larger fish (Rowan and Rasmussen 1995). Lake Apukkajärvi is a small oligotrophic lake in Rovaniemi, Finland

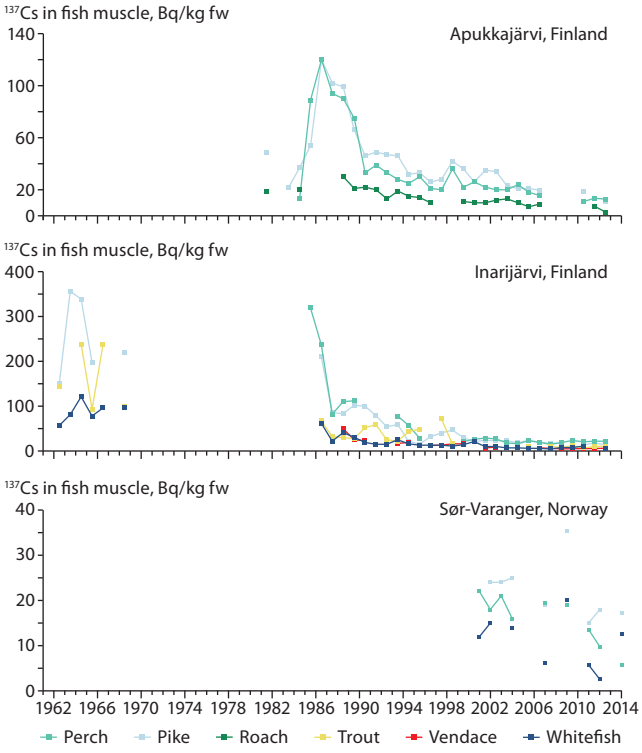

Figure 4.14 Activity concentrations of ^{137}Cs in muscle from fish species in Lake Apukkajärvi in northern Finland, Lake Inarijärvi in northern Finland, and Finnmark (Sør-Varanger, Norway; Gjelsvik et al., 2014).

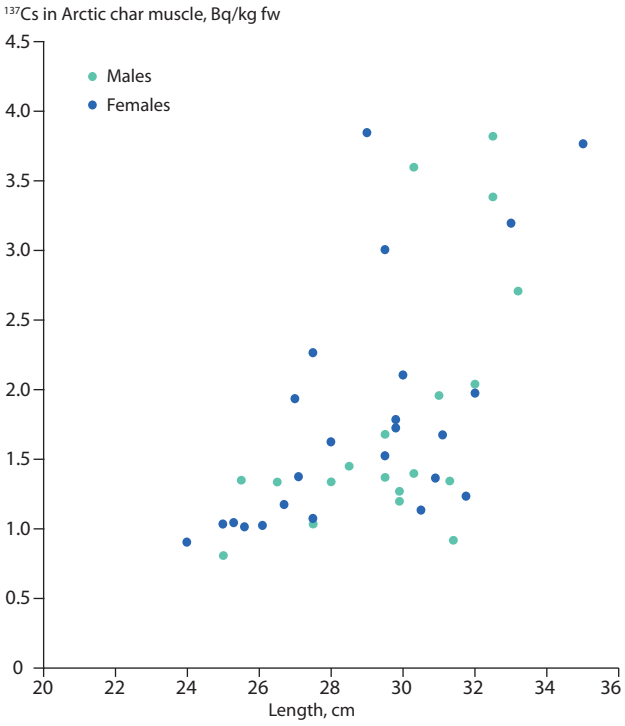

Figure 4.15 Activity concentration of ^{137}Cs in Arctic char muscle versus length of fish from Lake Fjellfroskvatn in northern Norway. Combined data from 2010 and 2011 (NRPA unpubl. data).

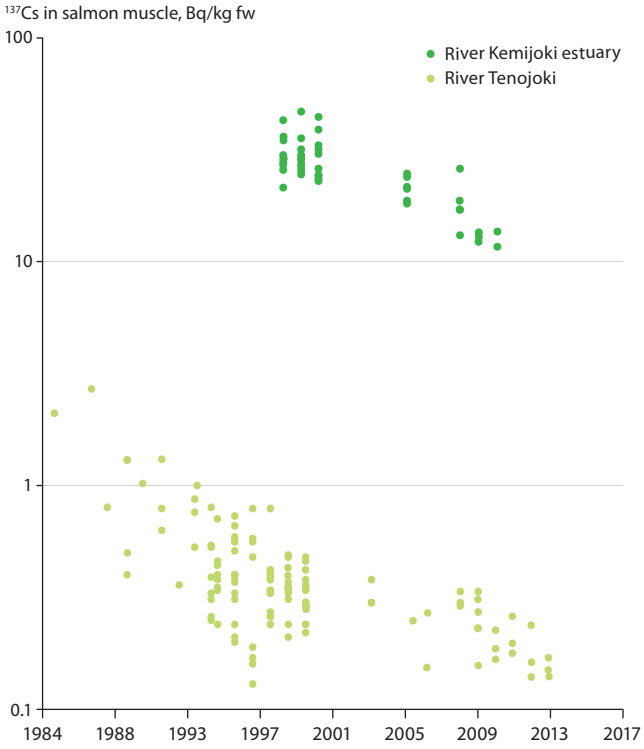

Figure 4.16 Activity concentrations of ^{137}Cs in muscle from salmon caught in the river Tenojoki (and its tributary river Inarijoki) and the Kemijoki river estuary.

while Lake Inarijärvi is a large lake important for fishing in Inari, Finland. Today there is little difference between the lakes in terms of ^{137}Cs levels in fish, with activity concentrations ranging from a few Bq/kg fw to a few tens of Bq/kg fw and the general trend over the past ten years has been a slow decline. However, levels are slightly higher compared to those of commercial fish species from, for example, the Barents Sea (see Sect. 4.2.4, Fig. 4.5).

Activity concentrations of ^{137}Cs in salmon in the rivers Kemijoki and Tenojoki have been measured since the mid-1980s (Fig. 4.16). The Tenojoki river marks the border between Finland and Norway and discharges into the Barents Sea, and the salmon caught in this river and its tributary the Inarijoki river originate from the Barents Sea and are there to mate. In contrast, the river Kemijoki discharges into the Baltic Sea. ^{137}Cs concentrations in Baltic Sea seawater (HELCOM, 2013) are an order of magnitude higher than in the Barents Sea (see Fig. 4.3), and this is also reflected in the salmon data (Fig. 4.16) where ^{137}Cs concentrations in Kemijoki river salmon are around 100 times higher than in Tenojoki river salmon.

4.4.2.2 Polonium-210

As part of Health Canada's study on background radiation levels in country foods, a total of 125 fish samples were collected from three lakes in the Experimental Lakes Area during summer 2014: Lake 305 (11 northern pike, 20 lake whitefish), Lake 302 (20 white sucker, 40 lake whitefish), and Lake 226 (34 lake whitefish). Naturally-occurring radionuclides (such as ^{226}Ra, ^{210}Pb, and ^{210}Po) were analyzed (Chen et al. 2015). While concentrations of all other radionuclides were below conventional detection limits, ^{210}Po was measured in most of the fish samples collected. Average ^{210}Po concentrations in fish muscle were similar at 1.7±1.5 Bq/kg fw (pike), 1.7±0.6 Bq/kg fw (white sucker), and 1.4±1.1 Bq/kg fw (whitefish). The overall average ^{210}Po concentration in freshwater fish muscle is estimated at 1.5 Bq/kg fw. Consuming 1 kg of fish with 1.5 Bq of ^{210}Po in muscle corresponds to a radiation dose of 2 µSv, which is a very small fraction (<1/1000) of the annual dose (about 3 mSv) from exposure to natural background radiation in Canada.

4.4.3 Soil

Soil sampling campaigns with different aims and spatial resolutions took place in Troms and Finnmark in 1986, 1995, 1998, 2004, 2005, 2010 and 2011 (Fig. 4.17). After the Chernobyl accident, the National Institute of Radiation Hygiene (now Norwegian Radiation Protection Authority) initiated a programme of coordinated soil sampling in each Norwegian municipality to document the total deposition of ^{137}Cs. The inventory of ^{137}Cs in soil depends on the initial deposition but also on soil type because ^{137}Cs binds differently to different types of soil particle. The amount and chemical composition of precipitation in the area determines the half-life of ^{137}Cs in the environment (Gjelsvik and Steinnes 2013). The current inventory of ^{137}Cs acts as a reservoir for the local plants and mushrooms that accumulate radionuclides through their respective root and mycelium systems.

4.4.4 Plants and plant-like organisms

STUK has collected vegetation samples throughout Finnish Lapland with a main interest in levels of ^{137}Cs. The species most studied are those important as winter reindeer forage (e.g. lichens) and for local populations in northern Finland

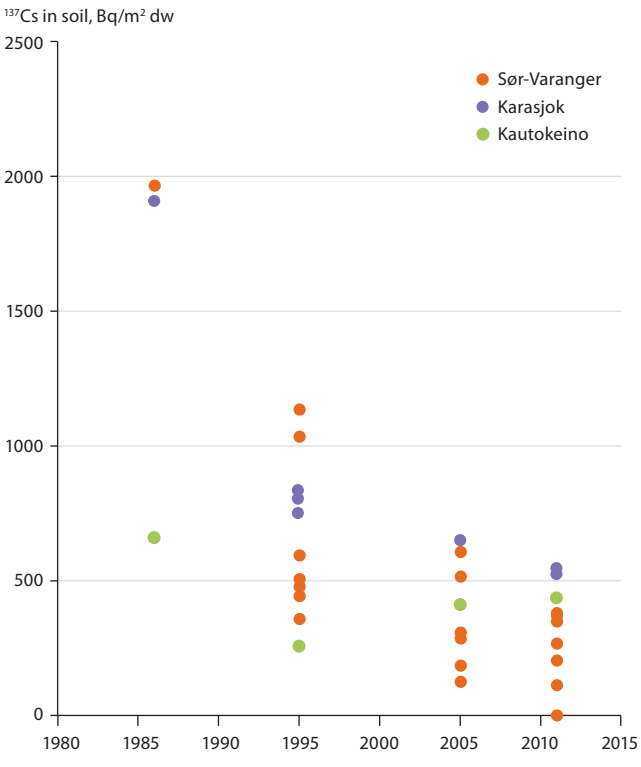

Figure 4.17 Inventories of ^{137}Cs in soil (0 to 3–4 cm) from three municipalities in Finnmark from sampling campaigns in 1986 (Backe et al. 1986), 1995 and 2005 (Gjelsvik and Steinnes 2013) and 2011 (Jensen et al. 2012).

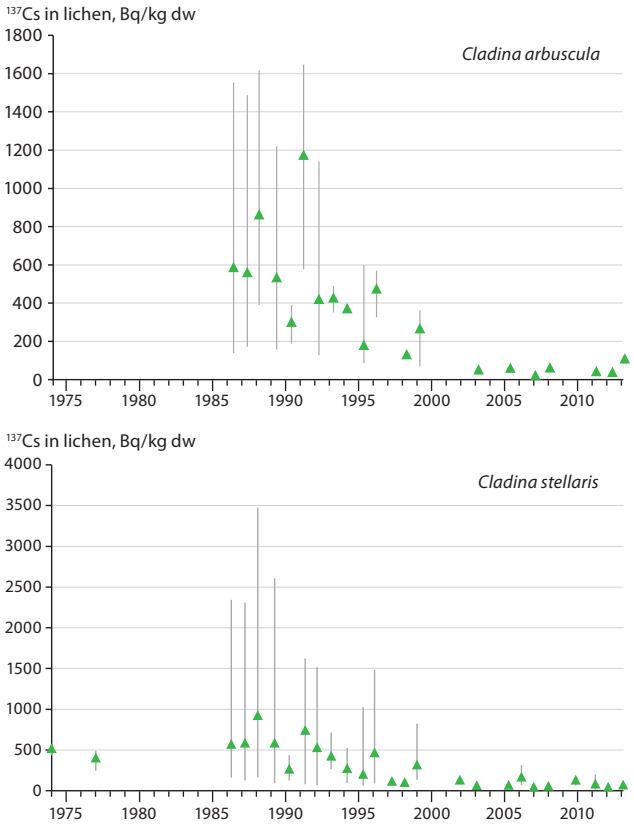

Figure 4.18 ^{137}Cs activity concentrations in the lichens *Cladina arbuscula* and *C. stellaris.* The data show annual means with vertical bars showing the range in observed concentrations (STUK unpubl. data).

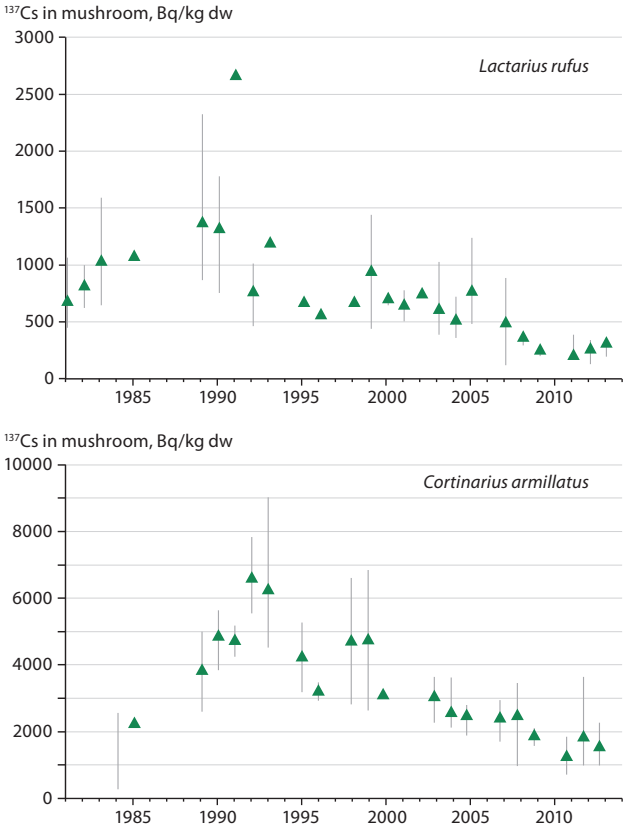

Figure 4.19. Activity concentrations of ^{137}Cs in the mushrooms *Lactarius rufus* and *Cortinarius armillatus* (Ylipieti and Rissanen 2012).

(e.g. berries and mushrooms). Data from across the whole of Finnish Lapland for the lichens *Cladina arbuscula* and *C. stellaris* show that levels of ^{137}Cs in these species have declined since the Chernobyl accident (Fig. 4.18). Lichens are formed by symbiotic association between fungi and algae or cyanobacteria. Most lichens do not absorb nutrients from their substrate but depend on atmospheric sources for nutrition. This means that the concentration of ^{137}Cs in lichens is directly proportional to the amount in deposition, which is why the high ^{137}Cs concentrations found in lichens immediately after the Chernobyl accident are no longer seen (as the older lichens are grazed by reindeer the source of ^{137}Cs for the newer lichens has declined considerably).

Activity concentrations of ^{137}Cs in two mushrooms species, *Lactarius rufus* and *Cortinarius armillatus*, in Finnish Lapland are now below the values observed in the mid-1980s before the Chernobyl accident (Fig. 4.19). The peak ^{137}Cs levels was found about 5–6 years after the Chernobyl accident when the Chernobyl fallout had migrated from the surface to soil layers at which the mycelium occur.

Collection of wild edible berries is popular in Scandinavia and Russia and their consumption may act as a pathway for the transfer of radioactive contaminants to man. ^{137}Cs levels in various edible berries from Troms (Norway) in 2010 (Table 4.6), were all below the limit set by the Norwegian Food Authority of 600 Bq/kg for ^{137}Cs in berries for commercial retail and are far below the levels in berries from central Norway that received higher levels of fallout following the Chernobyl accident (NRPA unpubl.).

Table 4.6 Activity concentrations of ^{137}Cs in various species of berry from Dividalen, northern Norway, 2010 (Gwynn et al. 2013).

Berry species	Number of samples	^{137}Cs in Bq/kg dw Geometric mean ±1σ (min–max)
Bilberry *Vaccinium myrtillus*	8	20.8 ± 2.4 (5.2–64.4)
Lingonberry *Vaccinium vitis-idaea*	6	13.5 ± 1.9 (5.8–29.0)
Crowberry *Empetrum nigrum*	10	7.9 ± 2.1 (2.5–27.7)
Cloudberry *Rubus chamaemorus*	6	13.5 ± 2.8 (3.8–46.8)
Bunchberry *Chamaepericlymenum suecicum*	5	35.1 ± 1.9 (16.1–77.8)
Juniper berry *Juniperus communis*	2	5.8 ± 2.4 (3.2–10.7)

4.4.5 Red fox

The NRPA has analyzed activity concentrations of ^{137}Cs in the muscle of red fox (*Vulpes vulpes*) from various locations in Finnmark, northern Norway (Fig. 4.20). The sources of anthropogenic radionuclides in the terrestrial environment in Finnmark are global fallout and to a limited extent the Chernobyl accident. The marine environment receives a continuous contribution from the nuclear fuel reprocessing plants at Sellafield (UK) and Cap de la Hague (France), and through outflow from the Baltic Sea. However, dilution in the marine environment means that ^{137}Cs levels in the marine

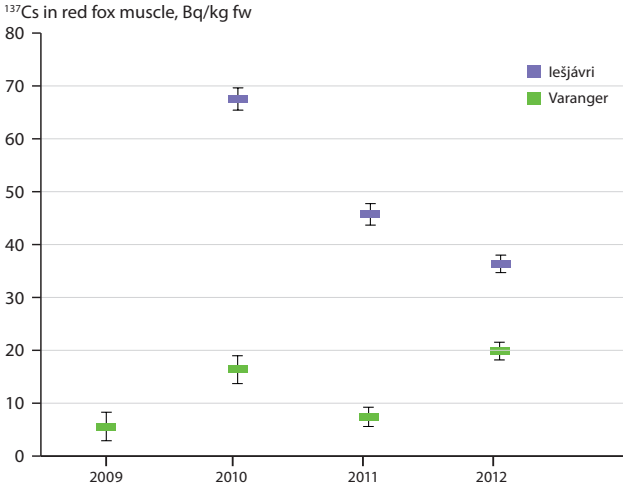

Figure 4.20 Activity concentrations of ^{137}Cs in red fox (*Vulpes vulpes*) muscle (geometric mean ± standard deviation) (NRPA unpubl. data).

environment in northern Norway are reduced compared to the terrestrial environment.

It is clear that foxes from Iesjavri have a higher ^{137}Cs concentration than foxes from the Varanger Peninsula. Red foxes are highly opportunistic feeders and will feed on available food items in their territory (Killengreen et al. 2011). The uptake of radionuclides in fauna is via the diet and the current concentrations of radionuclides in a specimen will reflect the concentrations in their specific food items. Iesjavri is an inland area while foxes living on the Varanger Peninsula may have access to marine food items. The results shown here support the findings of earlier studies that foxes living closer to the shore have a higher consumption of marine food items (Killengreen et al. 2011).

4.4.6 Reindeer/Caribou

Issues regarding reindeer and their forage as well as radioactive contamination in population groups that depend on reindeer and caribou for food have been covered by previous AMAP assessments. AMAP (2010b) presented information on this topic that is still relevant and factually correct. Nevertheless, some additional information and updates are available.

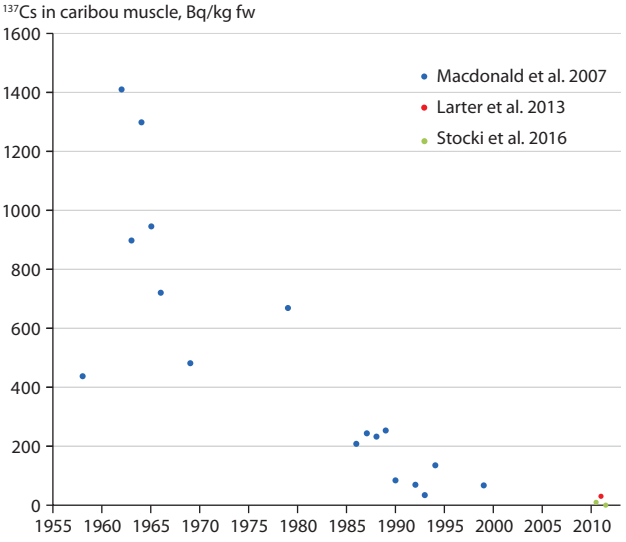

Figure 4.21 Activity concentrations of ^{137}Cs in caribou muscle from the Canadian Arctic between 1958 and 2012.

4.4.6.1 Canada

Activity concentrations of ^{137}Cs in caribou from the Canadian Arctic have been monitored for several decades (Thomas et al. 2005; Macdonald et al. 2007). Ongoing monitoring of major Canadian herds has confirmed a continuing decline in ^{137}Cs in all regions of Canada (Macdonald et al. 2007). Measurements made shortly before and after the FDNPP accident (Larter et al. 2013; Stocki et al. 2016) showed no significant increase of ^{137}Cs level in caribou (Fig. 4.21).

4.4.6.2 Norway

Reindeer cohorts and their herders have been monitored in northern Norway (Finnmark County) since 1967 and mid-Norway since 1987. In Finnmark, the radioactive contamination largely originates from the global fallout of the 1960s, whereas the Chernobyl accident is responsible for a significant amount of the radioactive contamination in mid-Norway.

AMAP (2010b) reported a slowdown in the decline in ^{37}Cs activity concentrations in reindeer meat in the Chernobyl-affected areas of Norway (Fig 4.22). Levels in early autumn (September) from 1987 onwards decreased with a half-life of about 9 years. In early winter (November–January) levels declined more rapidly (half-life about 6 years). However, from the early 2000s there has been a tendency towards a slower decline with a half-life approaching 30 years, the physical half-life of ^{137}Cs. This indicates the radioactive contamination is maintained in the system, probably bound in the soil, and is then decaying at the rate of the physical half-life. In autumn 2014, some areas of southern Norway experienced high levels of ^{137}Cs in reindeer cohorts. This sudden rise was caused by an unusually large number of mushrooms on the feeding grounds of the reindeer. The season was particularly good for gypsy mushroom (*Rozites caperatus*) which is known to be an efficient accumulator of radioactive cesium. Such events highlight the need to monitor both biota and reindeer in affected areas, and reindeer herders in the Chernobyl-affected areas of central Norway are still recommended to follow dietary advice to avoid internal doses from ^{137}Cs exceeding 1 mSv/y (Skuterud and Thørring 2012).

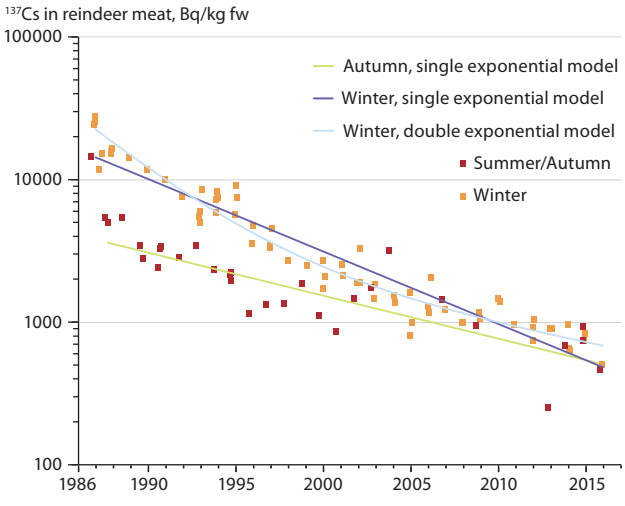

Figure 4.22 Activity concentrations of ^{137}Cs in reindeer meat from a Chernobyl-affected area of mid-Norway (update of data presented by Skuterud et al. 2005).

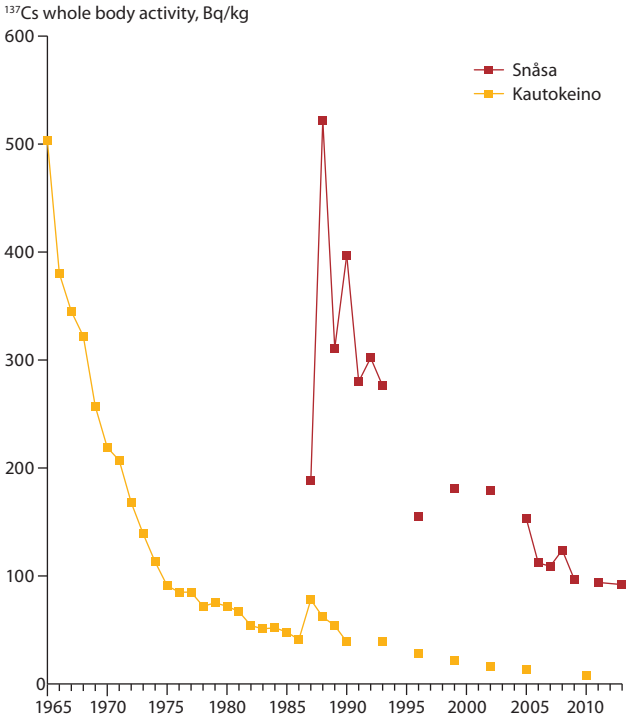

Figure 4.23 Whole body activity concentrations of ^{137}Cs in reindeer herders from Finnmark (Kautokeino) and mid-Norway (Snåsa).

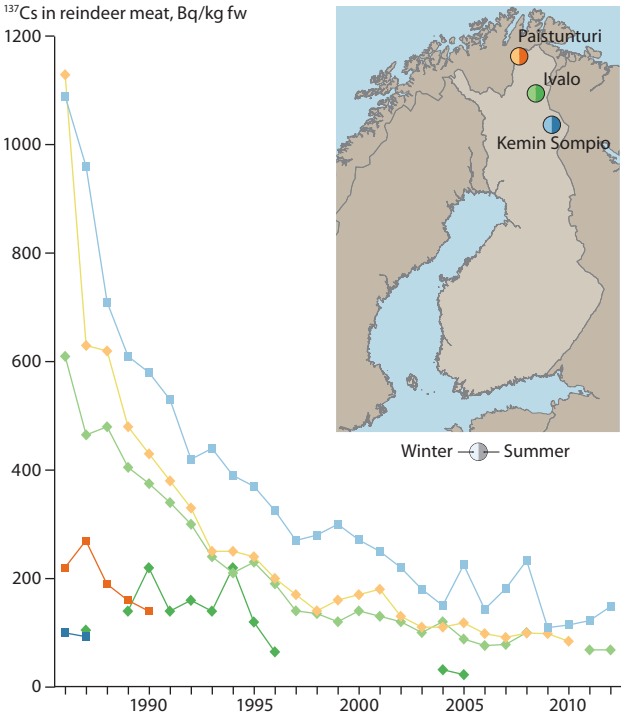

Figure 4.24 Change in ^{137}Cs activity concentrations in reindeer meat in three reindeer herding cooperatives in northern Finland since the Chernobyl accident in 1986.

Whole body counts of ^{137}Cs in reindeer herders have been undertaken by the NRPA since 1967 in Finnmark and 1987 in mid-Norway (see Fig. 4.23). Activity concentrations measured in Finnmark approached the levels detected in the general public around year 2000 and the monitoring program for this area was ended in 2010 (Thørring and Skuterud 2012). However, reindeer and their herders from mid-Norway are still affected by radiation released during the Chernobyl accident and owing to the slowdown in ^{137}Cs levels in reindeer meat, there have been only small reductions in ^{137}Cs levels in reindeer meat consumers over recent years. As a consequence, the monitoring of whole body counts in the mid-Norwegian area is still ongoing.

The FDNPP accident was responsible for negligible inputs of radioactive cesium to the European Arctic, and thereby responsible for only a small percentage increase to the contamination levels already present (adding about 10 Bq/kg to reindeer meat; NRPA unpublished data and CEEPRA).

4.4.6.3 Finland

Figure 4.24 shows the change in mean ^{137}Cs activity concentrations in three reindeer herding cooperatives in Finland. By number of reindeer, the Kemin-Sompio is the largest and the Paistunturi the second largest reindeer herding cooperative in Finland. Activity concentrations have declined steadily since their peak in 1986 and in 2013 were typically below 100 Bq/kg fw.

The atmospheric nuclear weapons tests of the 1960s have clearly been the main contributor to the ^{137}Cs burden of people living in the Inari region of Finland (Fig. 4.25). In fact, the profile of the change in whole body ^{137}Cs is very similar to that observed in Kautokeino, Norway (see Fig. 4.23). In 2011, the whole body ^{137}Cs content of people living in the Inari region had declined to a fraction of that observed in the 1960s.

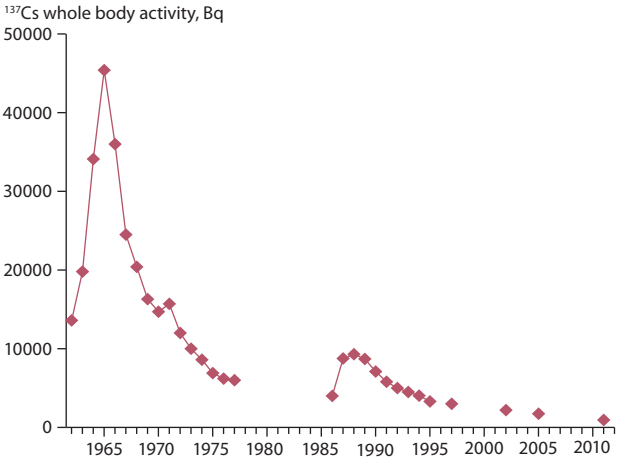

Figure 4.25 Whole body counts of ^{137}Cs for people living in the Inari region of Finland (Leppänen et al. 2011; STUK unpubl. data).

4.4.7 Anthropogenic radioactivity in milk

Milk is an important matrix for environmental radioactivity assessment since many radionuclides of health concern exhibit efficient soil-to-milk uptake pathways. As such, the surveillance of milk provides a very good indication as to the severity of environmental impact following a nuclear event. ^{90}Sr is a radionuclide of particular concern because it has a long radiological and biological half-life coupled with a high affinity for incorporation into human bone and teeth.

4.4.7.1 Canada

From 1984 to 1993, Health Canada's Canadian Radiological Monitoring Network (CRMN) collected data for ^{90}Sr activity in milk samples originating from 20 locations across Canada. After

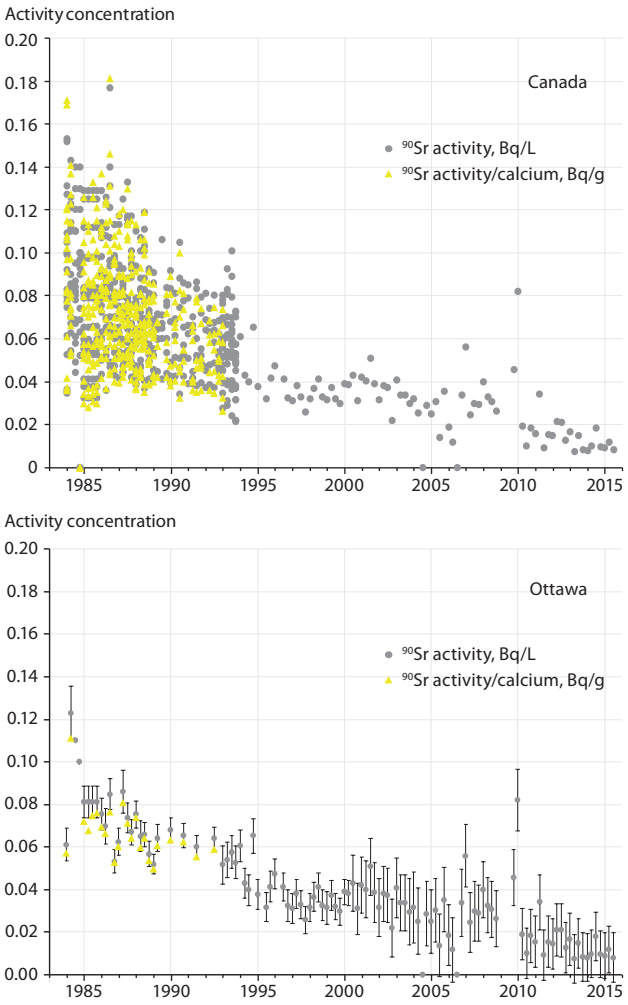

Figure 4.26 Activity concentrations of ^{90}Sr in whole milk from 20 sampling locations in Canada, and from the city of Ottawa, Ontario Canada.

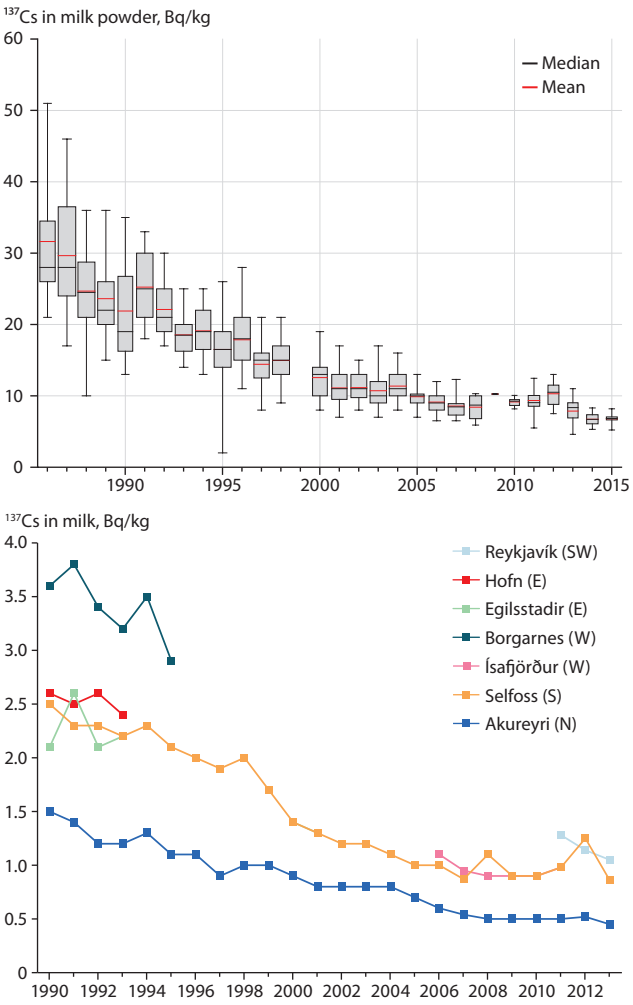

Figure 4.27 Activity concentrations of ^{137}Cs in milk powder from southern Iceland (Selfoss) and northern Iceland (Blönduós) (upper) and in milk from various dairies across Iceland (lower) since the Chernobyl accident.

1993, routine milk analysis for ^{90}Sr activity was discontinued for all monitoring stations, except the Ottawa location. The results are available via the Health Canada website[4]. Figure 4.26 shows a steady decline in the activity concentration of ^{90}Sr in milk since the end of the nuclear weapon tests in the 1960s.

4.4.7.2 Iceland

In Iceland, activity concentrations of ^{137}Cs are measured regularly in cow's milk and milk powder. Current values are very low and a slight decline in concentration can be seen over time (Fig. 4.27). The results are available via the Icelandic Radiation Safety Authority website (www.gr.is/fraedsla/skyrslur).

4.4.7.3 Finland

STUK has monitored ^{137}Cs activity concentrations in milk produced in Finnish Lapland since the early 1960s. Several different farms and dairies have participated in this monitoring. However, the most consistent monitoring has been undertaken in Rovaniemi and Salla (Fig. 4.28). In 2013, the measured concentrations ranged from 0.05 to 0.2 Bq/kg of fresh milk which is 100 times lower than the values observed in the mid-1960s (30 to 35 Bq/kg of fresh milk). Current levels of ^{137}Cs in milk are negligible.

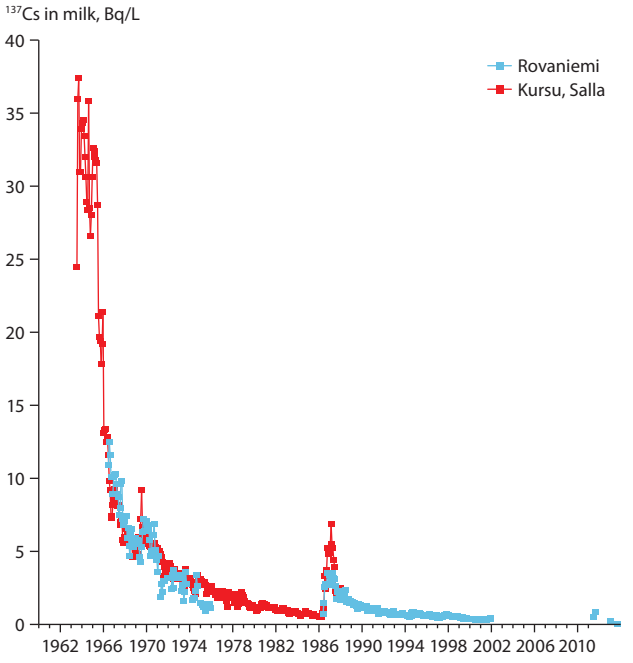

Figure 4.28 Activity concentrations of ^{137}Cs in milk produced in Rovaniemi and Salla in northern Finland (Solatie et al. 2009).

[4] http://data.gc.ca/data/en/dataset/12acd145-626a-49eb-b850-0a59c9bc7506/

4.5 Targeted assessments in areas of known radioactive contamination

The national monitoring programs summarized in this chapter are designed to establish long-term trends in anthropogenic radionuclides in particular areas and/or foodstuffs. However, it is also necessary to assess the current status of radioactive contamination in specific areas where human activities have increased the risk of contamination. This section reports on two targeted assessments. The first concerns Stepovogo Fjord on Novaya Zemlya off northern Russia, where numerous objects containing radioactive waste were dumped by the former Soviet Union and later Russia until the early 1990s. The second is Port Radium, a mining area in the Northwest Territories, Canada, which has elevated levels of naturally-occurring radionuclides due to the former mining and milling of radium and uranium.

4.5.1 Case study 1: Monitoring of dumped material in the Stepovogo Fjord

This section presents the outcome of the 2012 investigation by the Joint Norwegian-Russian Expert Group (JNREG) on the radioecological status of Stepovogo Fjord on the eastern coast of Novaya Zemlya. Stepovogo Fjord is characterized by a distinct inner and outer part that are partially separated from the open Kara Sea by an underwater sill at the entrance to the fjord. The inner and outer parts are themselves subdivided by an underwater sill at about 20 m. The inner part is up to 60 m deep, whereas the outer part reaches maximum depths of 30 to 40 m.

4.5.1.1 Dumped nuclear waste

The nuclear submarine *K-27* suffered a reactor accident on 24 May 1968 while on naval exercises. Following the accident, the *K-27* remained at Gremikha Fjord while the reactors cooled before being decommissioned in 1979. Following procedures to limit the potential for radioactive leakage from the vessel, the *K-27* was towed to Novaya Zemlya in September 1981 and sunk at a depth of 33 m in the outer part of Stepovogo Fjord. The total activity within the two reactors at the time of dumping was 2018 TBq (Sivintsev et al. 2005; Table 4.7).

Table 4.7 Selected radionuclides in the *K-27* reactors at the time of dumping (Sivintsev et al. 2005).

Radionuclide	Activity, TBq
^{239}Pu	0.49
^{240}Pu	0.015
^{241}Am	0.0069
^{137}Cs	486
^{90}Sr	496
^{60}Co	658

Within the inner part of Stepovogo Fjord, 1917 containers have been dumped at various depths with a reported total activity of 106 TBq (Table 4.8). The waste in these containers has been described as either unknown in origin or as containing a range of operational waste such as clothing and components (Sivintsev et al. 2005). Four reactor lids with a combined activity of 3.7 TBq have also been dumped in the inner part of Stepovogo Fjord (Sivintsev et al. 2005).

4.5.1.2 Past investigations of radioactivity in Stepovogo Fjord

A series of joint Norwegian-Russian investigations into the status of the dumped nuclear waste took place in the 1990s, with a cruise to the Kara Sea in 1992 followed by cruises to Stepovogo Fjord in 1993 and 1994 (JNREG 1996). The main objectives of the studies were to locate and identify dumped objects as well as to identify any leakage via the collection of environmental samples. These cruises were followed by other Norwegian-Russian cruises in 2003 and 2004 that focused on radionuclide contamination of sediments in Stepovogo Fjord (Dahle et al. 2009). A series of Russian-led investigations took place in the intervening years up to and including 2012.

Overall trends from the 1993 and 1994 studies showed activity concentrations of radionuclides in surface seawater in the fjord to be broadly similar to those observed in the open Kara Sea, but with elevated ^{137}Cs and ^{90}Sr levels in bottom water from the inner part of the fjord. For sediments, activity concentrations in cores from the fjord were similar to those taken in the open Kara Sea. However, single sediment samples collected by an ROV (remotely operated vehicle) close to *K-27* showed ^{137}Cs activity concentrations more than two orders of magnitude higher than in other samples. ^{137}Cs, ^{90}Sr, ^{134}Cs and ^{60}Co activities in surface sediments from the inner part of Stepovogo Fjord were highly variable but elevated in some samples taken close to dumped containers. Vertical sediment profiles showed higher ^{137}Cs and ^{239,240}Pu activities in the top 5 cm of sediment cores with decreasing activity concentrations down core.

The JNREG concluded that elevated levels of radionuclides in sediments collected close to dumped containers in the inner part of Stepovogo Fjord in 1993/1994 indicated that leakage

Table 4.8 Low- and intermediate-level solid radioactive waste dumped in Stepovogo Fjord (Sivintsev et al. 2005).

Year of dumping	No. containers	No. unpacked items	Activity at the time of dumping, TBq	Activity in 2000, TBq
1968	465	3	15.8	3.8
1970	243	-	31.7	8.1
1972	242	-	18.1	4.9
1973	517	-	24.4	6.7
1975	450	-	16.0	4.7
Total	1917	3	106.0	28.2

Figure 4.29 Sampling locations in Stepovogo Fjord in August/September 2012 (JNREG 2014).

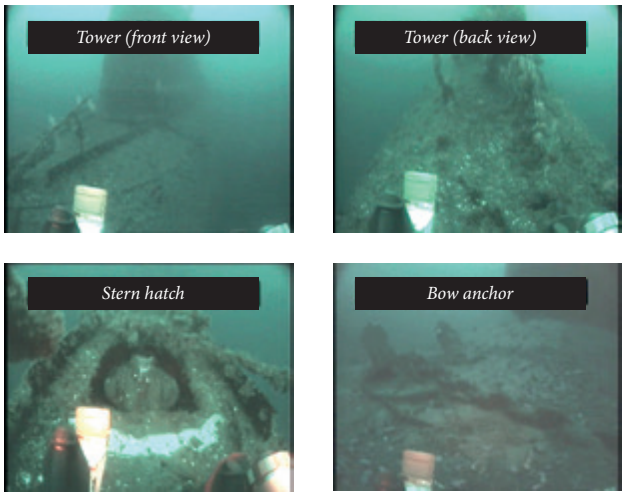

Figure 4.30 The nuclear submarine *K-27* in the outer part of Stepovogo Fjord (JNREG 2014).

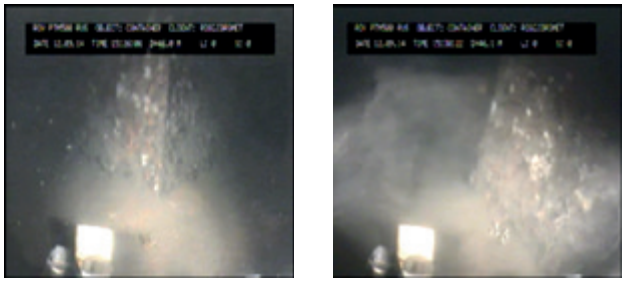

Figure 4.31 Dumped containers on the seafloor in the inner part of Stepovogo Fjord (JNREG 2014).

had occurred. However, follow-up investigations in 2002–2004 found no evidence of additional leakage from the dumped waste in the inner part of the fjord, nor any leakage from the reactors in *K-27* in the outer part of the fjord. Measurements showed sediment activity concentrations in Stepovogo Fjord in the early 2000s to have declined appreciably since the early 1990s.

4.5.1.3 Latest investigations of radioactivity in Stepovogo Fjord

The material presented here is from the results of the Joint Norwegian-Russian investigation aboard the RV *Ivan Petrov* of the Federal Service for Hydrometeorology and Environmental Monitoring (Roshydromet) in August/September 2012. This section presents a concise summary of the more detailed information available in the cruise report (JNREG 2014). Samples were collected in the Kara Sea near the entrance to Stepovogo Fjord, and from the inner and outer parts of the fjord (Fig. 4.29).

In 2012, *K-27* was observed lying upright and clear of bottom sediments at a depth of around 30 m in the outer part of the fjord with no obvious corrosion damage of the outer hull (Fig. 4.30). The deck was covered by a 3–5 cm layer of sediment that had been colonized by bivalve mollusks and other benthic organisms.

In the inner part of Stepovogo Fjord, only a limited number of containers were inspected during the 2012 investigation. These were observed at depths of up to 50 m embedded in soft sediments (Fig. 4.31). The containers examined seemed intact and had benthic organisms attached to the sides.

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

However, the activity concentrations of ^{137}Cs and, to a lesser extent, ^{90}Sr remained elevated in bottom water from the inner part of the fjord compared with surface water and the outer part of the fjord (Fig. 4.32). This is probably due to a combination of leakage from the dumped containers, the subsequent remobilization 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 ^{137}Cs in sediment cores in 2012 were slightly deeper (about 3 to 4 cm) than observed in 1993 (top 2 cm), indicating the slow burial of contaminated layers (Fig. 4.33). Activity concentrations of ^{137}Cs, ^{90}Sr, ^{239,240}Pu and ^{241}Am in all biota sampled in 2012 were low and, in the case of ^{137}Cs, often below the detection limit.

As activity concentrations of ^{137}Cs in surface sediments from the inner part of the fjord remain elevated, it is likely that sediments will continue to act as a diffuse source of ^{137}Cs to bottom water in this part of the fjord. Due to the bathymetry and physical geography of the inner fjord, future releases from dumped containers lying in the deeper areas are likely to have only limited impacts on the wider marine environment. However, future releases from dumped containers in shallower areas of the inner fjord may be more readily transported to the outer 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. The situation with regard to the nuclear submarine *K-27* in the outer part of Stepovogo Fjord should also be kept under review, especially in connection with any plans involving its recovery.

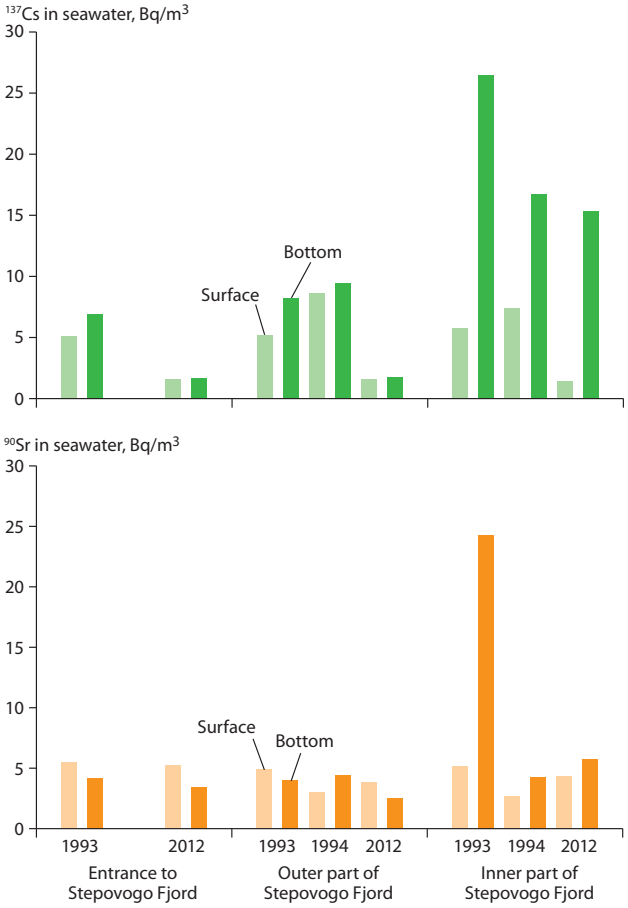

Figure 4.32 Comparison of mean activity concentrations of ^{137}Cs (upper) and ^{90}Sr (lower) in filtered seawater from Stepovogo Fjord in 2012 with results from 1993 and 1994 (JNREG 2014).

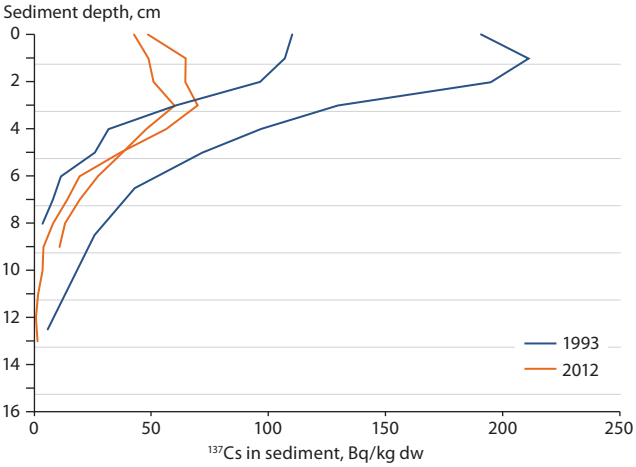

Figure 4.33 Comparison of activity concentrations of ^{137}Cs profiles in sediment cores taken from the inner part of Stepovogo Fjord in 2012 and 1993. Uncertainties on individual measurements were typically less than 10%.

4.5.2 Case study 2: Environmental monitoring at the Port Radium Site

This section provides a summary of environmental radioactivity data collected at the Port Radium Site over the recent five-year period (2008–2012) following site remediation in 2007 (SENES 2013).

Port Radium is located near Echo Bay, on the eastern shore of Great Bear Lake in the Northwest Territories. The site comprises several different areas: Port Radium Mine, Echo Bay Mine, Cross Fault Lake Mine, Cross Fault Lake Camp, and Glacier Bay Airstrip. Following the discovery of a pitchblende seam at the Port Radium Site in 1930, some form of mining and milling activity was undertaken almost continuously between 1932 and 1982. From the time of its initial discovery until 1940, the site was mined for radium-containing pitchblende. From 1942 to 1960, the site was operated as a uranium mine. The underground uranium mine was closed in 1962 in accordance with the standards at the time. The site was re-opened for silver mining in 1964. Upon depletion of the reserves in 1982, the site was shutdown according to the mine safety and land use requirements of the day. The Port Radium Mine remained effectively unchanged from 1982 to 2007 when the site was remediated.

Owing to past activities at the site and in association with ore/concentrate transportation, the local community expressed significant concern regarding existing contamination of the environment and the potential for future exposure to radiation through the traditional use of the land. Remediation activities began in May 2007, and continued to June 2008. The findings of environmental investigations on radiological status that were completed at the Port Radium Site during the pre-remediation (2001–2006) and pose-remediation (2008–2012) periods are briefly summarized here.

4.5.2.1 Pre-remediation period (2001-2006)

Land-locked tailings areas were covered to reduce gamma radiation and in these areas radiation levels were near the upper end of the range in natural background levels. Elevated gamma radiation levels (0.20–0.74 μSv/h) existed in some areas of the site including exposed tailings, and areas with waste rock, or other miscellaneous materials. There was no concern about impacts of radioactivity in surface water runoff in either the McDonough Tailings Containment Area (TCA) (previously known as Garbage Lake) or Great Bear Lake, and air quality, including levels of particulate matter and radon, were not of concern.

Quantitative human health and ecological risk assessments were undertaken to assess the risks associated with the site in its condition at that time. These indicated the limited environmental impacts associated with the site. From a human health perspective, the most significant potential exposure at the site would result from exposure to elevated levels of external gamma radiation. The exposure analysis demonstrated that under current land use conditions, the incremental dose associated with this pathway would be below the regulatory limit of 1 mSv per year for members of the public.

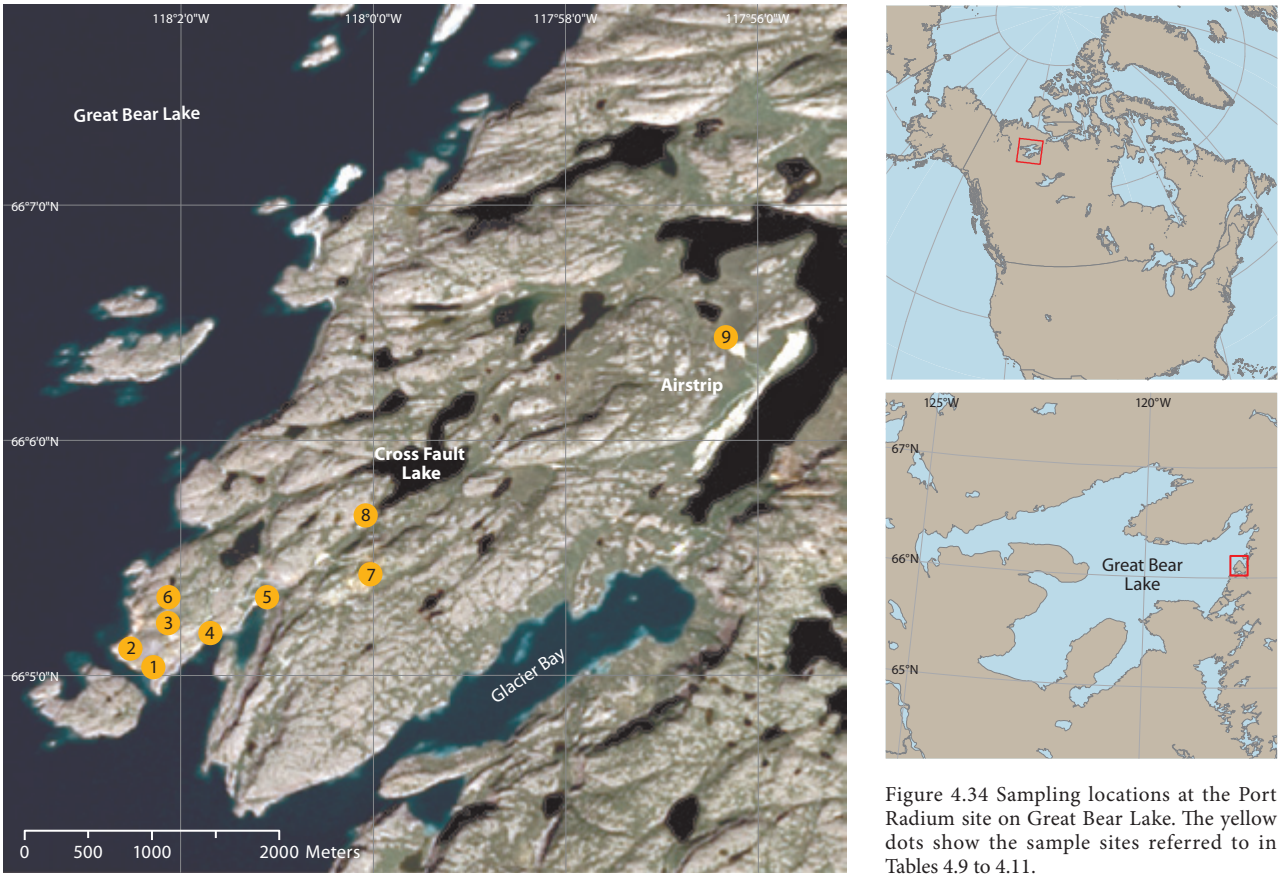

Figure 4.34 Sampling locations at the Port Radium site on Great Bear Lake. The yellow dots show the sample sites referred to in Tables 4.9 to 4.11.

4.5.2.2 Post-remediation period (2008–2012)

Spot gamma radiation measurements for the 2008–2012 period showed consistency from one year to the next and were comparable to the survey results obtained during the remediation period. The 2012 roving gamma radiation survey shows that activity levels were generally similar to or slightly below those measured during the remediation period in 2007. The results were also well below the remedial action level of 2.5 µS/h. There was no evidence of an increase in gamma radiation exposure rate as measured in 2012. Overall, monitoring confirms that the management measures in these areas have been effective in reducing residual radiological exposures.

To date, environmental criteria for radionuclides in water have not been developed for the protection of aquatic species. To provide some perspective, the measured activity concentrations of radionuclides in water were compared to Guidelines for Canadian Drinking Water Quality (Health Canada 2010). The 2008–2012 monitoring results demonstrate that with the exception of two samples all levels were below available guideline values for drinking water. The results of the 2008–2012 water quality monitoring program indicate that radionuclides in the vicinity of the former Port Radium mine are not a cause for concern for aquatic or human life.

Collection and analysis of fish has been an integral part of assessing the potential effects of the Port Radium site on the local environment. Lake trout (*Salvelinus namaycush*), lake whitefish (*Coregonus clupeaformis*) and lake herring or cisco (*C. artedi*) are a significant part of the traditional diet in the community of Déline and monitoring the quality of fish is an important component of the environmental monitoring program at Port Radium. Five composite lake trout samples comprising two to three muscle samples each were analyzed for ^{226}Ra and ^{210}Pb, major radionuclides in the ^{238}U decay chain. Both nuclides were at, or below, detection limits (^{226}Ra, 0.005 Bq/g fw; ^{210}Pb, 0.02 Bq/g fw). These levels correspond with values reported in lake trout collected in 2001 and 2006. Activity levels of ^{226}Ra and ^{210}Pb in whitefish from Cameron Bay (Great Bear Lake, about 13 km from the Port Radium Mine) in 2001 were also at the detection limit. All the data from the Port Radium monitoring program show ^{226}Ra and ^{210}Pb are at background levels. There is no evidence that radionuclides from the Port Radium site are affecting lake trout in this region of Great Bear Lake.

An integral component of the environmental protection program at Port Radium is the collection of surface soils and plants, which are subsequently analyzed for chemicals of potential concern. Soil monitoring was conducted to ensure that significant amounts of contaminants are not entering the terrestrial food web. This ensures the protection of species, such as moose, that form part of the traditional diet in this area and which may be exposed to contamination by foraging on species such as alder and aquatic plants. In 2012, soil samples were collected at nine sites (Fig. 4.34). Pooled soil samples were analyzed by gamma spectrometry to determine the concentrations of major radionuclides in the ^{238}U decay chain (Table 4.9). ^{226}Ra and ^{210}Pb reached a maximum level at Site 3 (Radium Lake), followed by Site 8, at which activity concentrations were two orders of magnitude lower than at Site 3. In general, ^{226}Ra and ^{210}Pb were close to secular equilibrium at all sites. Sites 4, 5, and 7 were very close in concentration to the two background Sites 6 and 9. The 2012 program was also designed to test plants near disturbed areas to ensure that contamination

Table 4.9 Activity concentrations of ^{226}Ra and ^{210}Pb in soils at Port Radium disturbed and control sites in 2012.

Sample site	^{226}Ra, Bq/g dw	^{210}Pb, Bq/g dw
Site 3	10	10
Site 4	0.13	0.12
Site 5	0.11	0.05
Site 7	0.05	0.09
Site 8	0.69	0.51
Site 6 (control)	0.13	0.24
Site 9 (control)	0.08	0.06

was not exceeding safe levels if ingested by wildlife, and not hazardous to human health. Table 4.10 provides a summary of the vegetation sampling conducted. The concentrations of radionuclides in plant species followed the same general trends as the non-radioactive elements. The highest concentrations were observed in sedge and alder at Site 2, and fell to close to or below detection limits at the background sites. The results are summarized in Table 4.11.

The change over time in activity concentrations of the major radioactive contaminants in plants reflects the same broad

Table 4.10 Site locations and samples collected in 2012 for multi-element scans and radionuclide analysis.

Sample site	Site from 2003	Date of collection	GPS location		Samples collected in 2012
Site 1	Site PR #1 (Silver Point)	16 Sept	N66 05.036	W118 02.285	Birch, willow, soil
Site 2	Site PR #8 (West Adit)	16 Sept	N66 05.116	W118 02.526	Birch, willow, alder, Labrador tea, sedge, soil
Site 3	Site PR #3 (Radium Lake)	16 Sept	N66 05.224	W118 02.133	Birch, willow, alder, Labrador tea, soil
Site 4	Site PR #7 (Roadway)	16/17 Sept	N66 05.181	W118 01.694	Birch, willow, alder, Labrador tea, soil
Site 5	Site PR #5 (Echo Bay Seep)	14&17 Sept	N66 05.334	W118 01.103	Birch, willow, alder, Labrador tea, soil
Site 6	Site 12 (background)	17 Sept	N66 05.334	W118 02.131	Birch, willow, alder, Labrador tea, soil
Site 7	Upper Echo Bay (2004)	17 Sept	N66 05.432	W118 00.028	Birch, willow, alder, Labrador tea, soil
Site 8	Cross Fault Lake west (2004)	17 Sept	N66 05.683	W118 00.081	Birch, willow, alder, sedge, Labrador tea, soil
Site 9	Background (Airstrip 2004)	18 Sept	N66 06.440	W117 56.332	Birch, willow, alder, Labrador tea, soil

Table 4.11 Activity concentrations of ^{226}Ra and ^{210}Pb in plant species from Port Radium in September 2012.

Location	Site name	Species	^{210}Pb, Bq/g dw	^{226}Ra, Bq/g dw
Site 1	Silver Point	Birch	0.22	0.056
Site 2	West Adit	Birch	0.10	0.25
		Willow	0.17	0.038
		Alder	0.19	0.011
		Sedge	0.20	0.60
Site 3	Radium Lake	Birch	0.12	0.043
		Alder	0.21	0.018
		Labrador tea	0.12	0.43
Site 4	Roadway	Birch	0.16	0.03
		Willow	0.14	0.008
		Alder	0.04	0.008
		Labrador tea	0.03	0.22
Site 5	Echo Bay Seep	Birch	0.07	0.044
		Willow	0.13	0.013
		Alder	0.12	0.017
		Labrador tea	0.06	0.043
Site 6	Background (Dumpy Lake)	Birch	0.08	0.029
		Willow	0.08	<0.005
		Alder	0.06	0.01
		Labrador tea	0.06	0.1
Site 7	Upper Echo Bay	Birch	0.07	<0.005
		Willow	0.06	<0.005
		Alder	0.07	<0.005
		Labrador tea	0.03	0.01
Site 8	Crossfault Lake Laydown area	Birch	0.08	0.012
		Willow	0.06	0.009
		Alder	0.07	<0.005
		Sedge	0.02	0.042
		Labrador tea	0.03	0.025
Site 9	Background (Airstrip)	Willow	0.04	<0.005
		Alder	0.06	<0.005
		Labrador tea	0.05	0.008

patterns as in soils, and provides an important link between elevated levels in soils and uptake into the food web. Analysis of plants at selected Port Radium sites indicates the potential for elevated levels of contaminants of concern, particularly arsenic and uranium. These elevated levels represent a risk to birds and mammals (i.e. wildlife) at the site. However, it is important to note that current levels of radioactive contamination are generally lower than those measured in prior sampling campaigns.

The findings of the 2008–2012 long-term monitoring program at Port Radium show that human health risks associated with being present at the Port Radium site were low before remediation, and are now even lower following remediation. Risks of adverse effects on wildlife are also low.

4.6 Conclusions

All eight Arctic countries have ongoing national monitoring programs to establish long-term trends in anthropogenic radionuclides.

Measurements in the Canadian Pacific, North Atlantic, Icelandic waters and Barents Sea all indicate that current levels of anthropogenic radionuclides are low compared to levels measured between the 1970s and 1990s, and are still decreasing. The major sources of anthropogenic radionuclides in the Arctic seas are global fallout in the 1960s. For the Atlantic side, a more recent source of radioactivity is the discharge of radioactive contaminants from the reprocessing plants at Sellafield and Cap de la Hague, and on the Pacific side there has been concern about the possible contribution of radioactivity from the accident at the Fukushima Daiichi Nuclear Power Plant (FDNPP). Dilution and sedimentation of the radioactive material mean contributions from the earlier nuclear weapons tests are now diminishing and national monitoring confirms that the reductions in authorized discharges from Sellafield and Cap de la Hague have resulted in lower activity concentrations of radionuclides in the marine environment.

Under normal circumstances the activity concentration of each of the naturally-occurring radionuclides (i.e. ^{7}Be and ^{210}Pb) is 100-fold higher than for the anthropogenic radionuclides (i.e. ^{134}Cs, ^{137}Cs, ^{131}I and ^{90}Sr) in air samples. Nevertheless, by employing sensitive analytical instruments it is still possible to detect changes in anthropogenic radionuclides. Measurements made at atmospheric monitoring stations across the Arctic have identified accidental releases such as those from the FDNPP accident or the possible accidental melt-down of a cesium-containing object at a smelter in Elektrostal, as detected by Norwegian monitoring. Natural events such as the redistribution of radionuclides during forest fires are also detected.

Monitoring in the freshwater environment reveals the same decreasing trend in activity concentrations of anthropogenic radionuclides (mainly ^{137}Cs) as seen in the marine compartment. The main source of ^{137}Cs contamination in the terrestrial environment is the Chernobyl accident and local differences in current concentrations of ^{137}Cs are, among others, caused by differences in fallout in 1986. Such variability is evident in, for example, geographical differences in activity concentrations in reindeer meat. However, as activity concentrations in fish depend on trophic level as well as on radionuclide biological half-life, present-day concentrations are determined by more than just the physical decay of radionuclides.

Targeted assessments of radioactive contamination in specific areas of the Arctic where human activities have increased the risk of contamination are also undertaken. For example, monitoring at Stepovogo Fjord on Novaya Zemlya off northern Russia identified limited elevated activity concentrations of anthropogenic radionuclides. Although the levels detected represent no immediate cause for concern, continuous follow-up is important because significant point sources remain in the area. At Port Radium in the Northwest Territories Canada, slightly elevated background radiation will not cause human exposure above recommended levels but may represent a risk to birds and mammals at the site.

Appendix: Overview of marine monitoring efforts

Area	Compartment	Radionuclides	Period covered by the monitoring
Canada (Pacific)	Fish	^{137}Cs	2013
		^{134}Cs	2013
		^{210}Po	2013
North Atlantic (Greenland and Faroe Islands)	Seawater	^{137}Cs	1970–2013
Icelandic waters	Seawater	^{137}Cs	1995–2012
	Fish	^{137}Cs	1996–2012
Barents Sea	Seawater	^{137}Cs, ^{90}Sr	2006–2013
		^{99}Tc	1997–2013
	Sediment	^{137}Cs	2006–2013
		Pu isotopes	2000–2009 (plus information from the 1990s)
	Fish	^{137}Cs	1979–2013
Kara Sea	Seawater	^{137}Cs	1993–1994 + 2012
		^{134}Cs	1993–1994
		^{90}Sr	1993–1994 + 2012
		^{239,240}Pu	1993–1994 + 2012
		^{241}Am	1993–1994 + 2012
	Sediment	^{137}Cs	1993–1994 + 2012
		^{134}Cs	1993–1994 + 2012
		^{90}Sr	1993–1994 + 2012
		^{238}Pu	2012
		^{239,240}Pu	1993–1994 + 2012
		^{241}Am	1993–1994 + 2012
		^{60}Co	2012

5. The Fukushima Daiichi accident and potential future accidents

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Key findings

- *The accidental release of radioactivity from the Fukushima Daiichi nuclear power plant in 2011 has caused only very low levels of additional anthropogenic radionuclides throughout the Arctic area*
- *Fukushima-derived radioactivity is not expected to result in adverse impacts on Arctic biota*
- *Monitoring indicates minimal risk posed by FDNPP-derived radioactivity in traditional (country) foods, and thus a negligible risk to Arctic residents through a traditional diet*
- *The Fukushima Daiichi accident has highlighted the potential for many nuclear accident scenarios, including those created by geohazards, to affect the Arctic region and emphasizes the importance of good communication between scientists, the general public and policy makers on associated risks to Arctic residents and wildlife*
- *The potential for an accidental release of radioactivity from existing, and planned, sources indicates the need for regular updates of the existing AMAP assessments of radioactivity in the Arctic*

5.1 Introduction

The Arctic is vulnerable to radioactive pollution transported from distant sources, whether by ocean currents or via the atmosphere. These sources include the atmospheric tests conducted in the 1950s and 1960s, nuclear fuel reprocessing, historical dumping and, more recently, accidents such as those at nuclear power plants in Chernobyl in 1986 and Fukushima in 2011. The latter serve as a reminder that accidents of considerable scale can and do occur at nuclear facilities. Given the number and age of many nuclear power plants located closer to the Arctic than those at Chernobyl (Ukraine) and Fukushima (Japan), there is reasonable cause for concern that an accidental release much closer to the AMAP area could have considerable impacts on the region. There is also potential for nuclear-related accidents within the Arctic region itself, such as through the use of floating nuclear power plants or nuclear-powered icebreakers. Atmospheric monitoring stations in the AMAP region will continue to provide a means for detecting routine emissions from nuclear facilities and for providing early warnings of major nuclear events.

5.2 The Fukushima Daiichi accident

On 11 March 2011, the Fukushima Daiichi Nuclear Power Plant (FDNPP) in Fukushima, Japan experienced two unprecedented natural disasters resulting in a severe failure of three on-site nuclear reactor units (Fig. 5.1). During the Great East Japan Earthquake the plant had executed an emergency shutdown but when the following tsunami waves hit the facility, the backup electric generators used for cooling were lost. Without cooling, the reactor cores overheated due to the decay heat in the nuclear fuel and there was a meltdown in three reactors. The accident culminated in significant releases of radioactive contaminants to the air due to hydrogen explosions in reactor buildings 1, 3 and 4, and to the marine environment beginning 12 March 2011. Estimates of the released activity vary between sources. According to Povinec et al. (2013a), approximately 159 PBq of ^{131}I (8-day half-life) and 15.3 PBq of ^{137}Cs (30-year half-life) were released into the atmosphere. The emissions were of significant domestic concern in Japan. The radioactivity released into the atmosphere travelled east across the Pacific Ocean to North America (Bowyer et al. 2011; Diaz et al. 2011), and so across to Europe (Masson et al. 2011) and then over to Central Asia (Bolsunovsky and Dementyev 2011). On day 15 after the initial releases, traces of radioactivity originating from the FDNPP could be detected across the entire northern hemisphere (Thakur et al. 2013).

It is important to note, however, that all radiation levels measured outside Japan have been very low and represent a negligible public and environmental hazard. This chapter details measurements of radionuclide concentrations in the Arctic environment. The first registrations of FDNPP-derived radionuclides were at the air monitoring stations and later transfer to terrestrial biota is evident.

Figure 5.1 Devastation at the Fukushima Daiichi nuclear power plant following meltdown at three on-site nuclear reactor units in March 2011. The inset shows the location of the Fukushima Daiichi NPP.

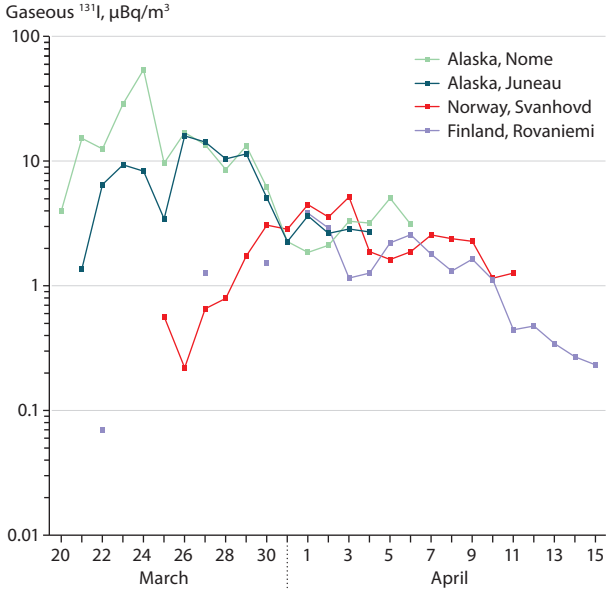


Figure 5.2 Activity concentrations of gaseous ^{131}I collected on activated charcoal filters in Alaska, Finland and Norway following the FDNPP accident on 11 March 2011. Note the logarithmic scale on the y-axis.

5.2.1 National monitoring responses – atmosphere

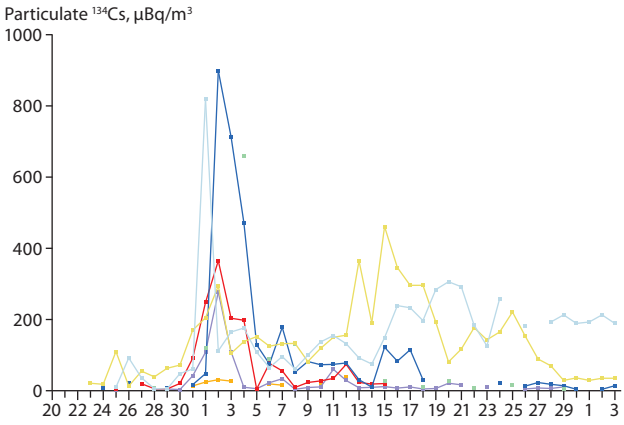
There are several atmospheric monitoring stations in the AMAP region that continuously monitor the activity concentrations of radionuclides in the lower atmosphere and at ground level. These are operated primarily for detecting routine emissions from nuclear facilities and for providing early warnings of major nuclear events. Following the FDNPP accident, sampling increased at many of these stations.

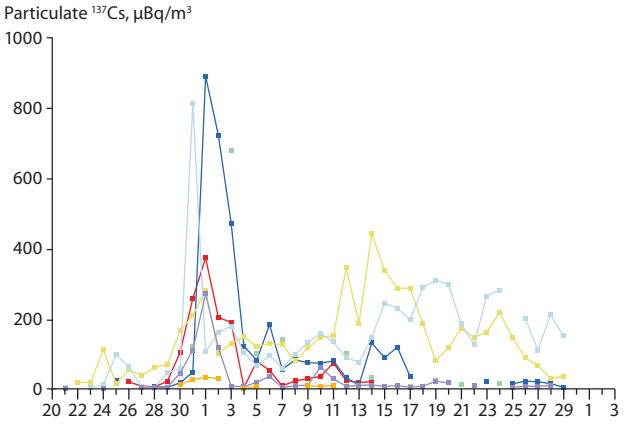
5.2.1.1 Alaska (USA)

The nationwide RadNet system monitors air, precipitation, drinking water, and pasteurized milk to track radiation within the U.S. environment. Immediately after the FDNPP accident, the RadNet system began operating on an emergency schedule, with expanded and accelerated sampling and analysis. The most robust data set for FDNPP-derived radionuclides in Alaska was obtained using charcoal filters to detect gaseous ^{131}I. Data for two representative stations – Nome and Juneau – are shown in Fig. 5.2 together with data from Finland (Rovaniemi) and Norway (Svanhovd). ^{131}I was detected at Nome and Juneau from 20 March, with the peak activity at Nome reached on 24 March (54 mBq/m^3) and at Juneau on 26 March (17 mBq/m^3). Data collection ceased in early April 2011.

5.2.1.2 Canada

Canada has an extensive network of monitoring stations across the country. For those in Arctic areas, the Yellowknife and the Resolute Comprehensive Nuclear-Test-Ban Treaty (CTBT) sampler stations are equipped to detect and measure radionuclides in particulate form. From these stations daily reports exist from the period after the FDNPP accident, and the data for ^{137}Cs, ^{134}Cs and ^{131}I are shown in Fig. 5.3. The most abundant radionuclide detected at Canadian sites was ^{131}I. Levels were low, amounting to no more than a few





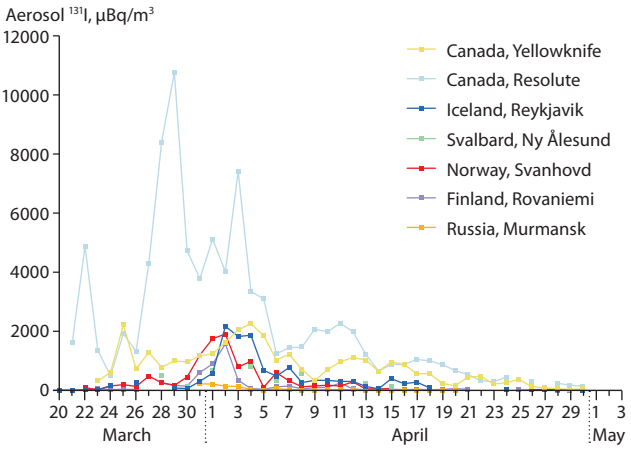


Figure 5.3 Activity concentrations of ^{134}Cs, ^{137}Cs and ^{131}I bound to aerosol particles in the Arctic, as detected at various Arctic air filter stations immediately after the FDNPP accident.

millibecquerels per cubic meter of air (mBq/m^3). The highest activity concentrations of ^{137}Cs and ^{134}Cs were an order of magnitude lower than those for ^{131}I and were more scattered. Arrival times were comparable to those of ^{131}I, plus or minus a day. Although ^{137}Cs has a relatively long half-life (30 years), activity concentrations decreased to non-detectable levels by the end of May (data not shown) owing to washout and dry deposition from the atmosphere.

5.2.1.3 Iceland

The Icelandic Radiation Safety Authority runs a CTBT station in Reykjavik. A total air volume of about 14,500–15,000 m^3 per day is filtered at this station with the filters being changed on a daily basis. The reported data are built on CTBT data, but as the CTBT's automated analysis system is designed to detect signs of nuclear explosions, it does not provide an accurate quantitative evaluation of the concentration of substances in the atmosphere. So the data presented here were revised by Michael Mooring at the Radiation and Nuclear Safety Authority (STUK) in Finland, who applied diagnostic tools to improve the data, as described by Guðnason et al. (2012). The first European evidence of the FDNPP accident was observed at the CTBT station in Reykjavik on the filter operated from 19–20 March. ^{131}I was detected, while ^{134}Cs and ^{137}Cs were not detected until four days later (see Fig. 5.3).

5.2.1.4 Norway

The Norwegian Radiation Protection Authority (NRPA) has three sampling stations that monitor atmospheric radioactivity in the Arctic region. These are located at Svanhovd (Pasvikdalen), Skibotn (in Storfjord municipality, Troms County) and at Viksjøfjell (east of Kirkenes, near the Russian border). Immediately after the FDNPP accident, the intensity of monitoring was increased at the Svanhovd station; these data are reported here. In addition, a sampler located at Svanhovd was also used to measure ^{131}I in the gaseous phase (Møller et al. 2013). Of the radionuclides analyzed, the ^{131}I in gaseous form was observed in by far the highest concentrations thus underlining the importance of this isotope with regard to possible health effects from accidents such as that at the FDNPP. The highest activity concentrations of atmospheric radionuclides at Norwegian stations were observed in the samples collected during 1–2 April (Figs. 5.2 and 5.3).

5.2.1.5 Finland

In Finland, STUK has three sampling stations located in the Arctic region: Ivalo, Sodankylä and Rovaniemi. In addition, the Finnish Meteorological Institute also has three aerosol sampling stations in the Finnish Lapland at Kevo (municipality of Utsjoki), at Sodankylä and at Rovaniemi. The southernmost station, in Rovaniemi, is located on the Arctic Circle. This station was sampled for aerosols and radionuclides in gaseous form on a daily basis immediately after the FDNPP accident. The first observations of the FDNPP-derived plume were observed in samples collected during 18–21 March. Highest concentrations were observed in samples collected during 1–2 April (Figs. 5.2 and 5.3) (Leppänen et al. 2013b).

5.2.1.6 Svalbard (Norway)

The Finnish Meteorological Institute (FMI) has monitored activity concentrations of ^{210}Pb in air at the Mt Zeppelin Global Atmosphere Watch (GAW) station, Ny Ålesund, Svalbard, since 2000 in collaboration with the Norwegian Polar Institute (NPI). After the FDNPP accident, samples were analyzed for, among others, ^{131}I and ^{137}Cs to determine whether the radionuclide emissions could also be detected in the atmosphere of the High Arctic. Owing to a fixed sampling regime with three filter changes per week, measurements at the Zeppelin station represent 2–3 day periods. The peak concentrations of ^{131}I and ^{137}Cs were found in the days up to 4 April 2011 (Fig 5.3) (Paatero et al. 2012).

5.2.1.7 Russia

Monitoring of environmental radioactive contamination is conducted by subdivisions of the Federal Service of Russia on Hydrometeorology and Environmental Monitoring located north of the Arctic Circle. Figure 5.3 presents data from the aerosol filtering station in Murmansk. Monitoring at this station is daily and the first signs of radioactivity from the FDNPP accident were observed on 31 March. Activity concentrations for both ^{131}I and ^{137}Cs are considerably lower at the Murmansk station that at other stations in the nearby area (Rovaniemi and Svanhovd) although the reason for this difference is not clear. The radionuclides of FDNPP origin detected in the Russian Arctic were present at levels indicating negligible hazard to the general population (see Box 5.1).

5.2.1.8 Atmospheric transport of FDNPP-derived radionuclides

Routine and emergency monitoring of atmospheric radioactivity levels by the eight Arctic nations following the significant release of radioactive contaminants from the FDNPP, shows ^{134}Cs and ^{137}Cs to have exhibited similar distributions and concentrations over the following days. The data presented in Fig. 5.3 show a peak in concentration (31 March – 4 April) recorded at Resolute (Canada: 74.6975°N, 94.8322°W) which did not occur to the same extent further south at Yellowknife (Canada: 62.4422°N, 114.3975°W). The peak at Resolute occurred one day ahead of a peak of similar magnitude recorded in Iceland indicating rapid eastward transport of radioactive cesium at this latitude. Stations at Yellowknife, Rovaniemi (Finland) and Svanhovd (Norway) all detected a peak on the same day as Iceland (2 April) but at less than half the activity concentration, indicating that most radioactive cesium was transported in a plume at a higher latitude. The peak detected in Ny Ålesund (Spitsbergen) was recorded three days later, probably due to the interrupted sampling regime at this station. A second plume of lower activity air was transported along a lower latitude, recorded at Canadian stations in mid-April, but these releases were not detected at any of the other Arctic stations except for Iceland.

Both particulate and gaseous ^{131}I was found in higher activity concentrations on the North American continent in late March 2011, than at the European stations. The peaks in ^{131}I at the European stations were considerably lower compared to the American and Canadian data due to dilution in the air masses. Activity concentrations in all areas gradually decreased during April and most stations reduced their monitoring efforts in May.

Box 5.1 Radionuclides of 'FDNPP origin' well below Russian norms

Radioactive iodine of 'FDNPP origin' was detected for the first time on 23 March 2011 in the European part of Russia (Moscow region) and was detectable across the Russian Arctic in low volume activities <0.3 μBq/m³ (Roshydromet 2012) from 27 March. During the period 28–30 March, the activity concentration of FDNPP-derived ^{131}I measured in air over the entire European Russian territory increased suddenly and ^{134}Cs, ^{136}Cs, ^{137}Cs, ^{132}I, ^{132}Te were also recorded in air. The peak activity concentration of FDNPP-derived radionuclides in the central part of the European Russian territory was observed during 3–4 April: for ^{131}I up to 4 μBq/m³ and for ^{137}Cs up to 1.2 mBq/m³. Figure 5.4 presents the mean monthly activity concentrations of ^{137}Cs in surface air over the Russian Arctic compared with similar data for the central part of the European Russian territory. The graphic shows that the volume activity of FDNPP-derived radionuclides measured in air in the Russian Arctic was considerably lower than in the central areas of the European Russian territory (Roshydromet 2012). The arrival of ^{137}Cs in air masses contaminated by radioactive products of the FDNPP accident in Japan is also evident in the elevated ^{137}Cs activity concentrations in surface air over the Russian Arctic in 2011 compared to previous years (Table 5.1).

Although radioactive products from the hydrogen explosions at the FDNPP could be measured in surface air across the Russian Arctic from the end of March until June 2011, the volume activities of the FDNPP-derived radionuclides observed in the Russian Arctic atmosphere over that period were still several orders of magnitude below the norms currently in force in Russia where permissible atmospheric mean annual volume activities of 7.3 Bq/m³ for ^{131}I, 19 Bq/m³ for ^{134}Cs and 27 Bq/m³ for ^{137}Cs (NRS 2009) are judged to cause no hazard for the general population.

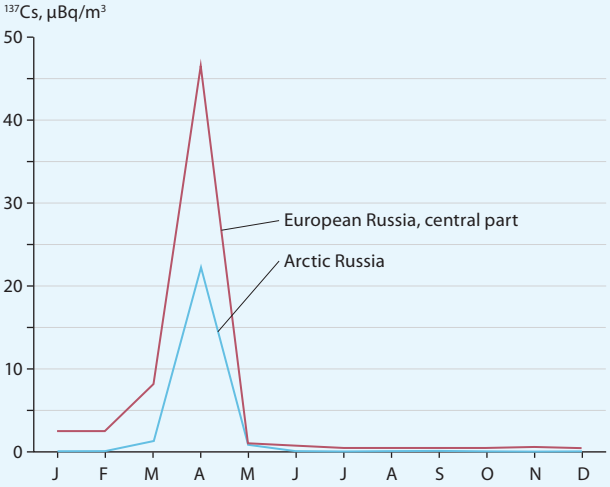


Figure 5.4 Atmospheric activity concentration of ^{137}Cs in the Russian Arctic and in the central part of the European Russian territory in 2011 (Roshydromet 2012).

Table 5.1 Mean annual activity concentrations of ^{137}Cs in the surface atmospheric layer and total ^{137}Cs atmospheric deposition, averaged across the Russian Arctic (Roshydromet 2008, 2009, 2010, 2011, 2012, 2013). Volume activity data from sites at Norilsk (not currently operational), Dixon, Turukhansk, Salekhard, Narjan-Mar, Murmansk, Kandalaksha, Zasheek (not currently operational), Amderma.

Date	Volume activity ^{137}Cs, μBq/m³	Deposition ^{137}Cs, Bq/m²/y
2006	0.10	<0.4
2007	0.12	<0.4
2008	0.09	<0.2
2009	0.11	<0.2
2010	0.09	<0.2
2011	2.12	2.0
2012	0.16	0.2

5.2.2 National monitoring responses – biota

In addition to monitoring atmospheric radioactivity, many Arctic nations have also measured activity levels in biota following the FDNPP accident. The findings from monitoring activities in Alaska, Canada, Norway, Finland and Russia are presented in the following sections.

5.2.2.1 Alaska (USA)

As a follow up on the use of Amchitka Island as a nuclear test site in 1965–1971, the US Department of Energy monitors radioactivity there and on the nearby reference island Adak. In the 2011 sampling season, the decision was made to include ^{134}Cs in the analyses of reindeer lichens (*Cladina* spp.) to look for any FDNPP-derived contribution to the cesium levels at both sites. Table 5.2 reports the levels of ^{134}Cs and ^{137}Cs in lichens from Amchitka and Adak, before (1997) and after (2011) the FDNPP incident. Levels in 2011 are much higher than in 1997, but given that the ratio of ^{134}Cs:^{137}Cs is approximately 1 (0.99±0.10, Leppänen et al. 2013b) from the FDNPP fallout some of the ^{137}Cs detected in 2011 may derive from detonations at Lop Nor, China (1964–1996) (USDoE 2013) or a local source. In general, activity concentrations at Adak are higher than at the Amchitka test site, which is explained by the higher precipitation at Adak compared to Amchitka (USDoE 2013).

5.2.2.2 Canada

There is no regular monitoring of radioactivity levels in biota in Canada, but to address growing public concern about radioactive contamination of the Pacific Ocean fish stocks, samples of commonly consumed salmon and groundfish were obtained from the Canadian west coast in 2013. None of the fish samples analyzed in that study contained ^{134}Cs or ^{137}Cs activity concentrations that exceeded the average detection limit of ~2 Bq/kg (Chen 2014).

Following the FDNPP accident in March 2011, northern Canadians expressed concern about possible radioactive contaminants in traditional country foods. As a result, a study funded by the Northern Contaminants Program was undertaken to measure levels of radionuclides in Arctic caribou and beluga. Samples of lichens, mushrooms, caribou and beluga taken before and after the FDNPP accident were freeze dried, homogenized, and measured using gamma ray spectroscopy to identify the radionuclides present and

Table 5.2 Activity concentrations of ^{134}Cs and ^{137}Cs in reindeer lichen (*Cladina* spp.) sampled at Amchitka and Adak islands before the FDNPP accident (1997) and after the FDNPP accident (2011) (USDoE 2013).

	^{134}Cs, Bq/kg dw		^{137}Cs, Bq/kg dw	
	No. of samples	Median (min–max)	No. of samples	Median (min–max)
Amchitka				
Pre-FDNPP (1997)	-	-	4	- (2.3–2.7)
Post-FDNPP (2011)	3	79.6 (49.6–222)	3	95.1 (69.9–263)
Adak				
Pre-FDNPP (1997)	-	-	3	- (0.9–0.9)
Post-FDNPP (2011)	3	463 (392–925)	3	559 (492–1077)

Table 5.3 Activity concentrations of ^{134}Cs and ^{137}Cs in plant and animal samples collected in the Canadian Arctic before and after the FDNPP accident (Stocki et al. 2016). Detection limits are listed for each sample type as less than value.

Sample type	Sample dates	No. of samples	^{134}Cs, Bq/kg ww	^{137}Cs, Bq/kg ww Mean (min–max)
Pre-FDNPP				
Lichens	n/a	n/a	n/a	n/a
Mushrooms	7 Aug 2010	6	<0.6	10.2[a] (2.2–35)
Caribou	11 Sep – 29 Nov 2009	20	<2.8	18.8[a] (9.4–46)
Beluga	5–27 Jul 2010	19	<1.7	-
Post-FDNPP				
Lichens	5 Aug 2011	11	<3.8	3.92[b] (0–13.8)
Mushrooms	5 Aug 2011	6	<0.6	9.2[b] (0–19)
Caribou	17 Aug – 25 Sep 2011	14	<2.8	9.1[a] (3.6– 20.2)
Beluga	6–24 Jul 2011	22	<1.7	-

[a]Geometric mean; [b]arithmetic mean.

determine the radioactivity concentration in the samples. To determine the efficiency of the detectors for the different-sized samples, physical calibration standards were used and virtual simulations were also performed. A comparison of ^{137}Cs in the caribou samples before and after the accident (see Table 5.3) indicates no observable increase in activity. The amount of ^{137}Cs measured in the mushrooms and lichens was low, and the pre-accident mushrooms had the same level of ^{137}Cs as the post-accident mushrooms. No ^{137}Cs was detected in the pre-accident beluga samples, even when combining all ^{137}Cs measurement peaks into one spectrum. There was also no detection of ^{137}Cs in the individual post-accident beluga samples. But when the individual post-accident beluga measurement peaks for ^{137}Cs were combined, a negligible amount of radioactive ^{137}Cs was detectable (0.63±0.23 Bq/kg ww), although no ^{134}Cs. This implies that any ^{137}Cs present in beluga samples is probably due to fallout from the atmospheric weapons tests of the 1960s (Stocki et al. 2016). Both the caribou and beluga results have been communicated to the communities and stakeholders in the region such that they are aware of the minimal risk posed by FDNPP-derived radioactivity in these foodstuffs.

5.2.2.3 Norway

The NRPA has analyzed a number of environmental samples since the FDNPP accident. In northern Norway all terrestrial samples of mushrooms, lichens, reindeer and red fox contain detectable levels of ^{137}Cs. After the FDNPP accident a proportion of the samples also contained ^{134}Cs. Table 5.4 shows the samples where both cesium isotopes were detected. Levels are

Table 5.4 Activity concentrations of ^{134}Cs and ^{137}Cs and the mean increase in ^{137}Cs originating from the FDNPP accident in environmental samples from Finnmark and Troms, Norway during 2012–2013 (NRPA unpublished data).

Sample type	No. of samples	^{134}Cs, Bq/kg Median (min–max)	^{137}Cs, Bq/kg Median (min–max)	Mean increase in ^{137}Cs concentration, %
Mushrooms[a]	6	2.3 (0.5–7.1)	810 (600–1340)	0.3
Lichens[a]	2	2.6 (1.4–3.9)	80 (65–96)	3.3
Reindeer[b]	2	0.38 (0.35–0.41)	87 (86–87)	0.4
Red fox[b]	23	0.44 (0.21–0.78)	22 (13–71)	1.5

[a]Dry weight; [b]wet weight.

similar to those observed in Finnish Lapland. Section 5.2.2.4 explains the method for assessing the additional contribution of radiocesium derived from the FDNNP accident in biota. Applying this method on the Norwegian data shows that the contribution from the FDNPP accident to the anthropogenic radioactivity concentrations was very small, of the order of a few percent.

5.2.2.4 Finland

The Radiation and Nuclear Safety Authority (STUK) routinely monitors radioactivity in environmental samples. After the FDNPP accident, particular attention was paid to the detection of ^{134}Cs. Because Chernobyl-derived ^{134}Cs has virtually all decayed, almost all ^{134}Cs measured in biota today is considered to originate from the FDNPP accident. Figure 5.5 presents observed mean ^{134}Cs concentrations in the star reindeer lichen *Cladonia stellaris* in Lapland, Finland. The trend line shows the decay in ^{134}Cs over time – the more recent ^{134}Cs observations clearly deviate from the trend observed for Chernobyl-derived ^{134}Cs. Table 5.5 summarizes the results for ^{134}Cs and ^{137}Cs detected during 2011–2013. The ^{134}Cs:^{137}Cs ratio in the FDNPP release was approximately 1 (0.99±0.10, Leppänen et al. 2013b). Several studies have found ^{134}Cs:^{137}Cs ratios in agreement with this observation; for aerosol filters, fallout, and in seawater collected close to the FDNPP (e.g. Masson et al. 2011; Aoyama et al. 2012; Beresford et al. 2012; Ioannidou et al. 2012; Paatero et al. 2012). Using a ^{134}Cs:^{137}Cs ratio of 1 the measured ^{134}Cs value indicates the proportion of the measured ^{137}Cs derived from the FDNPP accident alone. Table 5.5 indicates the influence of the FDNPP accident on radioactivity levels in Lapland to be very small; activity concentrations of ^{137}Cs typically increased by less than 1%.

5.2.2.5 Russia

The Murmansk Marine Biological Institute collected and analyzed environmental samples from Murmansk Oblast during 2011–2012. The FDNPP release was only observed in one sample of star reindeer lichen – collected outside the town of Apatity on 18 October 2011. The ^{134}Cs concentration in this sample was 1.9±1.0 Bq/kg and the ^{137}Cs concentration 33±8 Bq/kg. Although only based on one sample, the additional contribution of the FDNPP accident to the existing environmental ^{137}Cs level could be about 6%. This is comparable to the increase observed in lichen samples in northern Norway.

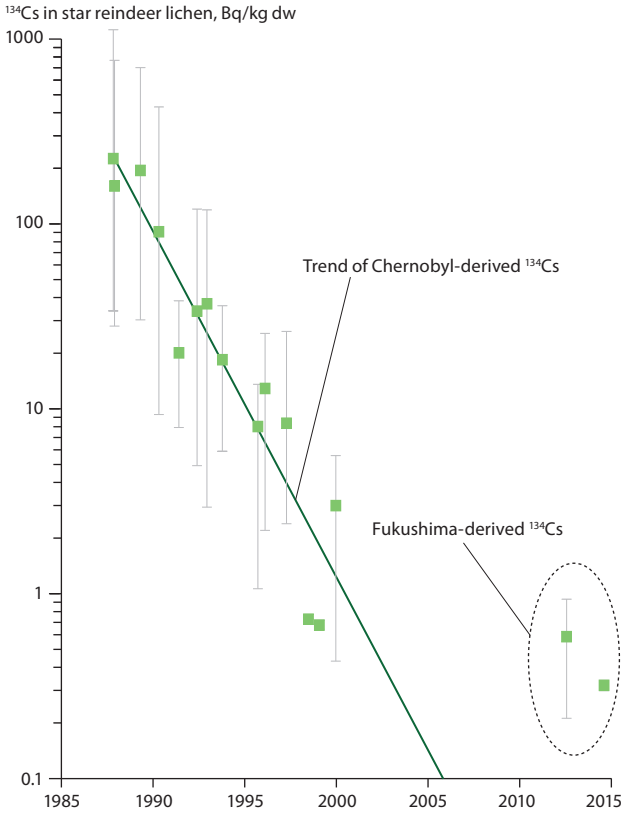


Figure 5.5 Average annual ^{134}Cs concentrations observed in star reindeer lichen *Cladonia stellaris* in northern Finland (Koivurova et al. 2015).The trend line is fitted to the data observed after the Chernobyl accident.

Table 5.5 Activity concentrations of ^{134}Cs and ^{137}Cs and the mean increase in ^{137}Cs originating from the FDNPP accident in environmental samples from northern Finland during 2011–2013 (Koivurova et al. 2015).

Sample type	No. of samples	^{134}Cs, Bq/kg ww Mean (min–max)	^{137}Cs, Bq/kg ww Mean (min–max)	Mean increase in ^{137}Cs concentration, %
Reindeer	37	0.32 (0.04–1.2)	100 (1–260)	0.6
Moose	3	0.02 (0.02–0.03)	16(15–18)	0.3
Wolf	9	0.7 (0.2–1.1)	200 (150–380)	0.4
Fish	20	0.09 (0.03–0.27)	14 (5–46)	1.6
Milk	1	0.013	0.24	5.3
Cloudberry	1	0.12	64	0.06
Lingonberry	2	0.019 (0.018–0.021)	3.8 (3.6–3.9)	0.9
Blueberry	2	0.020 (0.017–0.23)	3.5 (3.3–3.7)	0.9
Mushrooms	28	0.07 (0.01–0.24)	44 (4.4–150)	0.4
Lichens[a]	16	0.67 (0.16–1.2)	61 (3.7–28)	1.7
Beard lichens[a]	24	0.56 (0.24–1.3)	13 (3.7–28)	6.9

[a]Beard lichens grow on tree trunks and branches where they obtain all nutrients from the air and rainwater whereas 'Lichens' grow on the ground and so absorb nutrients from the soil as well. Beard lichens are also an important reindeer winter fodder and provide a pathway for cesium to enter reindeer and ultimately man.

Table 5.6 Effective half-lives for ^{137}Cs and ^{134}Cs in years calculated for Arctic sample types after the FDNPP accident (Koivurova et al. 2015) and the Chernobyl accident (Paller et al. 1999; Morita and Yoshida 2005; Puhakainen et al. 2007; Franic et al. 2008; Skuterud et al. 2009; Iurian et al. 2010; Leppänen et al. 2011; Taylor et al. 2011; Skrkal et al. 2013). The uncertainties correspond to 1σ uncertainty. The R^2 is the coefficient of determination.

Sample type	Post-FDNPP			Post-Chernobyl	
	^{137}Cs	^{134}Cs	R^2	^{137}Cs	^{134}Cs
Reindeer	2.8 ± 0.7	1.2 ± 0.3	0.38	3–4	1.40
Fish	4.7 ± 1.9	1.5 ± 0.6	0.26	3.2–16.7	1.93
Mushrooms	3.5 ± 1.3	1.4 ± 0.5	0.21	5.5–16	1.59
Lichen	19 ± 7	2.1 ± 0.8	0.33	2.7–3.4/10–12	1.82
Beard lichen	1.6 ± 0.2	0.91 ± 0.18	0.55	-	-

5.2.2.6 FDNPP-derived radionuclides observed in Arctic biota

The results from atmospheric measurements and the analyses of radioactivity in Arctic biota presented here are consistent with other observations and agree well with results reported in the scientific literature (Bolsunovsky and Dementyev 2011; Bowyer et al. 2011; Diaz et al. 2011; Masson et al. 2011; Thakur et al. 2013; Leppänen et al. 2013b). Several plumes of radioactivity dispersed across the Arctic following the FDNPP accident and the fallout levels varied geographically. From an AMAP perspective, it is important to note that an accidental release of radioactivity from an NPP situated far from the Arctic can, over a relatively short period (15 days in the case of the FDNPP accident), cause measurable fallout across the Arctic region. The environmental consequences of the additional radioactivity in the Arctic region originating from the FDNPP accident are expected to be minimal. However, the Chernobyl and FDNPP incidents serve as a reminder that accidents of considerable scale can and do occur at nuclear facilities, often arising from a chain of events that are only clearly visible with hindsight. Given the number and age of many NPPs situated much closer to the Arctic than the FDNPP there is reasonable cause for concern that an accidental release closer to the AMAP area could have considerable impacts on the region.

5.2.2.7 Transfer factors calculated from FDNPP fallout data

This section on transfer factors calculated from the FDNPP fallout and their comparison with transfer factors calculated after the Chernobyl accident in 1986 is part of a study by Koivurova et al. (2015). The effective half-lives and aggregated transfer factors reported here were based on ^{134}Cs and ^{137}Cs in environmental samples collected in northern Finland during 2011–2013 (see Table 5.5). The calculations are based on the assumption that new ^{134}Cs and ^{137}Cs fallout, around 0.5–1 Bq/m^2, has originated from the FDNPP accident. ^{137}Cs is detected regularly in environmental samples but ^{134}Cs has not been observed for over a decade. Thus Koivurova et al. (2015) were able to determine effective half-lives (Table 5.6) and aggregated transfer factors (Table 5.7) based on FDNPP fallout. The aggregated transfer factors and effective half-lives for ^{134}Cs and ^{137}Cs determined by Koivurova et al. (2015) after the FDNPP accident agree well with those reported in the literature that were determined after the Chernobyl accident.

Table 5.7 Ranges in aggregated transfer coefficient for cesium calculated for various Arctic sample types after the FDNPP accident based on ^{134}Cs fallout in northern Finland (Koivurova et al. 2015) and the Chernobyl accident (Rantavaara and Haukka 1987; Saxén and Koskelainen 1992; IAEA 1996, 2010; Lehto et al. 2013).

Sample type	Post-FDNPP	Post-Chernobyl
Wet weight measurements		
Reindeer	0.80–1.8	0.6–1.1
Moose	0.038–0.15	0.018–0.024
Wolf	1.0–2.2	–
Fish	0.03–0.60	0.03–0.12
Milk	0.038	0.0008–0.0015
Dry weight measurements		
Cloudberry	0.08	0.002–0.23
Lingonberry	0–0.10	0.032–0.04
Blueberry	0–0.13	0.017–0.12
Mushrooms	0.11–0.19	0.01–3.22
Lichen	0.8–2.3	1.4
Beard lichen	1.9–3.5	–

Reindeer accumulate cesium efficiently due to a diet based on large quantities of lichens and it is also no surprise that wolf also accumulate cesium efficiently being one of the top predators. One surprising result was the relatively high accumulation of cesium by epiphytic beard lichens. Beard lichens are important winter fodder for reindeer and provide an important pathway for cesium into reindeer meat and to people consuming reindeer meat. However, the effective half-life for ^{134}Cs in beard lichen is relatively short at about 0.91 years (see Fig. 5.6), and just over half that of ^{137}Cs (1.6 years). The importance of regional variation in the relative increase in ^{137}Cs originating from the FDNPP accident is currently unclear (Fig. 5.6 right). The post-FDNPP determined aggregated transfer factors for mushrooms were lower than the post-Chernobyl values. This is understandable because there is a time lag of typically 3–5 years between fallout on the soil surface and the migration of radiocesium to the soil layers at which it is available to the fungi mycelium. This suggests the post-FDNPP sampling could have taken place too soon.

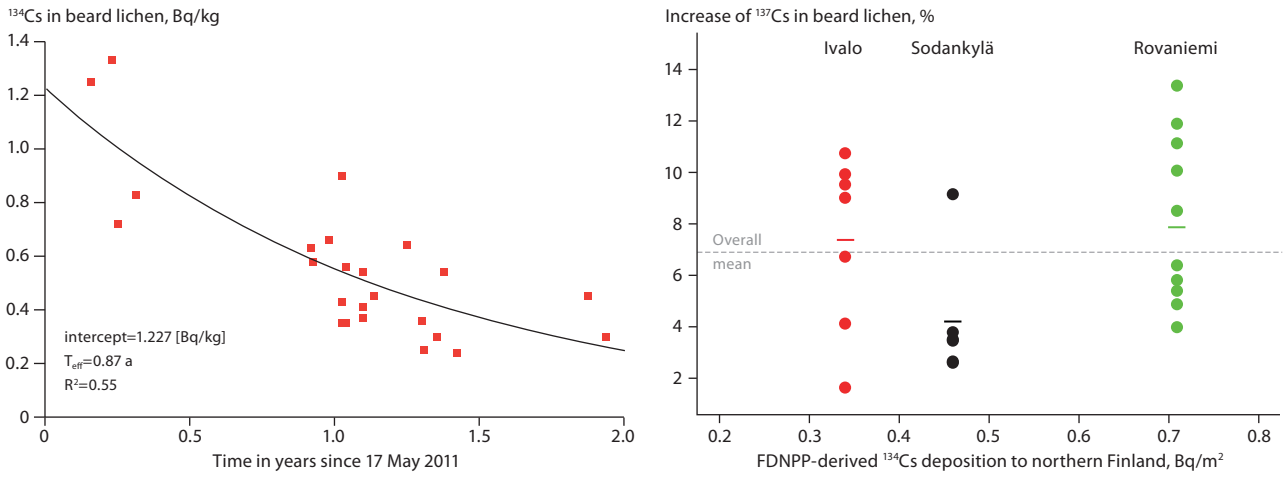


Figure 5.6 Effective half-life of ^{134}Cs in beard lichen (left) and the percentage increase of ^{137}Cs in beard lichen in northern Finland owing to FDNPP-derived fallout for three different areas. The horizontal bars show the mean percentage ^{137}Cs increase for each area and the grey line shows the overall mean (right) (Koivurova et al. 2015).

5.2.3 Ocean transport of FDNPP-derived radionuclides

Marine studies near the Japanese coast a few months after the FDNPP accident measured levels of ^{131}I, ^{137}Cs, ^{3}H, ^{14}C, ^{129}I and ^{90}Sr in ocean water (Povinec et al. 2012, 2013b; Casacuberta et al. 2013). With a half-life of about 8 days, ^{131}I does not contribute to the long-term contamination of the Pacific Ocean, whereas ^{137}Cs, with a half-life of 30 years, is widely dispersed by ocean currents. Inputs to the North Pacific Ocean from atmospheric deposition and direct discharges from the FDNPP have been estimated by many studies since the accident. On the basis of these studies, the International Atomic Energy Agency (IAEA) has estimated the total input to the ocean to be 10.5 PBq (log-normal mean; range 6.4–17.3 PBq) (IAEA 2015b).

Ocean transport of FDNPP-derived radionuclides has been investigated through direct sampling in 2011 and 2012 (Buesseler et al. 2012; Ramzaev et al. 2014). Roshydromet performed radioecological surveys in areas relevant for the transport of radioactivity to the Russian coastline. The surveys to assess radioactive contamination in seawater and air took place in the Sea of Japan and in the Kurile-Kamchatka region of the Pacific Ocean (see Fig. 5.7 for sample sites). The first survey was completed in April-May 2011 onboard the R/V *Pavel Gordienko* and the second took place in August-September 2012 onboard the R/V *Akademik Shokalsky* (Ramzaev et al. 2014).

The accidental releases were observed to have minimal influence on radioactivity levels in the area towards the Russian coast of the Sea of Japan. Only trace amounts of ^{134}Cs were detected in seawater (maximum 0.3 Bq/m^3) while ^{137}Cs was in the range typical for the local background level (1–3 Bq/m^3), with mean ^{137}Cs activity concentrations in surface waters of 2.8 Bq/m^3 (Sea of Japan), 2.0 Bq/m^3 (Pacific Ocean Kure-Kamchatka region) and 2.4 Bq/m^3 (northwestern Pacific Ocean, to the west of Japan) (IAEA 2005b; Ramzaev et al. 2014).

However, east of Japan, elevated ^{137}Cs activity concentrations were found in 2011 and 2012. In 2011, the Russian expedition measured up to 31.2 Bq/m^3 at Station 4. This is in the lower range of levels detected by Buesseler et al. (2012) who conducted an extensive expedition to the area east of Japan in June 2011. They measured surface water levels of FDNPP-derived radionuclides to a distance of 600 km from the NPP, and found the eastward Kuroshio Current to act as a southern boundary to their spread. In 2012, the Russian expedition observed ^{134}Cs and ^{137}Cs activity concentrations in the upper 100 m layer to be still much higher than pre-accident background levels, in the range 20–30 Bq/m^3 for both radionuclides (Ramzaev et al. 2014: their Table 1). The maximum levels of ^{134}Cs and ^{137}Cs were detected in the North-Western Pacific at the northern periphery of the Kuroshio Current, about 400 km from the FDNPP. Most importantly, the 2012 data show a downward mixing of ^{137}Cs in the water column (Fig. 5.8).

The downward mixing of ^{137}Cs documented by Ramzaev et al. (2014) is in accordance with the predictions of basin-scale models applied to assess the spread of ^{137}Cs from the FDNPP (Behrens et al. 2012; Nakano and Povinec 2012; Rossi et al. 2013, 2014).

The arrival of the FDNPP-derived ocean transport plume at the west coast of Canada was reported by Smith et al. (2015). By June 2013, measurements of ^{134}Cs and ^{137}Cs in seawater showed the spread of the FDNPP signal onto the Canadian continental shelf. By February 2014, activity levels had increased to 2 Bq/m^3 throughout the upper 150 m of the water column. These observations were consistent with projections by Rossi and co-workers concerning the timing of arrival of the initial FDNPP ^{137}Cs signal, and the revised modelling simulations of Rossi and colleagues (2013, 2014) indicating a maximum FDNPP ^{137}Cs level of 2.8 Bq/m^3 on the west coast of Canada in 2015. Total levels of ^{137}Cs (FDNPP-derived plus fallout ^{137}Cs) off the North American coast are expected to attain maximum values in the 3–5 Bq/m^3 range by 2015–2016 before declining to levels closer to the fallout background of about 1 Bq/m^3 by 2021.

None of these model studies focused on if and when the elevated levels may reach Arctic areas but they all include the Bering Sea, which is defined as a part of the Arctic area by AMAP (AMAP 1998). Whereas the models of Behrens et al. (2012) and Nakano and Povinec (2012) suggested a small (0.3–0.4 Bq/m^3) intrusion of FDNPP-derived ^{137}Cs north of the Aleutian chain after two to three years, Rossi et al. (2013, 2014) suggested a higher contribution of up to 2 Bq/m^3 by 2016. The difference in source term definition (10 PBq by Behrens et al. 2012; and 22 PBq by Rossi et al. 2013, 2014) does not fully account for the differences in the estimates, meaning that the inherent distinctions of the models applied are significant (Buesseler 2014).

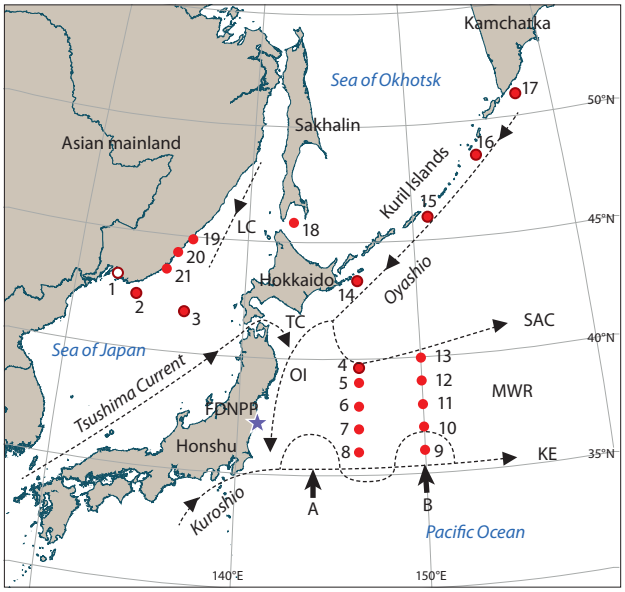


Figure 5.7 Study area with water sampling stations. Station No. 1 in 2011 is shown with an open circle, stations in 2012 are shown with closed circles, and repeat stations visited in 2011 and 2012 are shown with double circles. Dashed lines show major currents directions: Kuroshio (warm current), Oyashio (cold current), LC (Liman cold current); OI (Oyashio Intrusion), TC (Tsugaru current), SAC (Sub-Arctic current), KE (Kuroshio Extension), MWR (mixed water region), FDNPP (Fukushima Daiichi nuclear power plant). Arrows A and B indicate positions of the ridges for the 144°E and 150°E quasi-stationary meanders of the Kuroshio Extension.

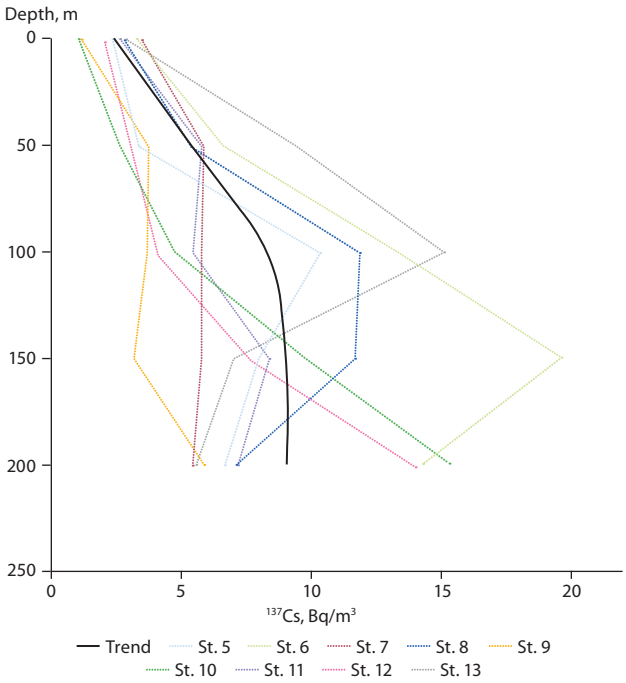


Figure 5.8 Vertical distribution of ^{137}Cs activity concentrations in the upper 200 m of the water column at nine stations sampled in the period 21–26 August 2012 to the south of the Oyashio (SubArctic) Front in the western North Pacific Ocean. For station code and location see Fig 5.7. The solid line represents average ^{137}Cs activity concentrations for the nine stations and the activities are given on the date of sampling (Ramzaev et al. 2014).

Future projections thus estimate a moderate increase in ^{137}Cs level in the Bering Sea of 0.3–2 Bq/m^3. This increase will probably not cause harm to either wildlife or human health (Smith et al. 2015). As a comparison, the Baltic Sea in northern Europe is densely populated and one of the most radiologically contaminated marine areas in the world, with ^{137}Cs levels well above the projected increase for the Bering Sea (IAEA 2005b). The average doses received by coastal inhabitants around the Baltic Sea are 4–4.5 mSv/y and the calculated additional dose to coastal residents eating contaminated fish is a maximum of 0.04–0.2 mSv/y depending on year and area (Nielsen et al. 1999). Based on maximum ^{137}Cs activity concentrations in fish species sampled on the west coast of Canada in 2013, Chen et al. (2015) estimated a worst-case annual dose from ^{134}Cs and ^{137}Cs of 2 μSv for an adult consuming 50 kg of fish per year. This is one thousandth of the annual natural background radiation dose in Canada, and of no radiological health concern.

5.3 Potential for nuclear-related accidents in the Arctic region

There are many nuclear power plants located much closer to the Arctic than the FDNPP (see Ch. 2) which could, in the event of an accident, potentially cause significant risk of harm to Arctic residents and wildlife. Several of these have had their planned lifetimes extended and/or their capacity increased. In addition, many new plants are planned over the coming decade. Increased shipping in the Arctic area could also increase the use of the existing fleet of nuclear-powered icebreakers.

5.3.1 Floating nuclear power plants: history and status

The Russian Federation initiated the development of floating nuclear power plants (FNPPs) about 25 years ago. Even during the 1980s, Russia developed plans for low capacity power plants for the provision of power and heat in remote Arctic areas (AMAP 2010b). The plans were revised many times until in 2009, OKBM Afrikantov (www.okbm.nnov.ru/english) completed the assembly of two prototype reactors based on low-enriched uranium fuel (<20% ^{235}U). The hull of the world's first FNPP, *Akademik Lomonosov*, was launched in June 2010 at the Baltiysky Zavod shipyard, St. Petersburg.

5.3.1.1 *Akademik Lomonosov*

The *Akademik Lomonosov* is equipped with two KLT-40S reactors, which are a modified version of the KLT-40 reactor on the nuclear ice-breaker *Taymyr*. Box 5.2 reviews the technical aspects of the *Akademik Lomonosov* FNPP.

In addition to site safety, concerns have been raised about the KLT-40S reactor itself. For example: inadequate decommissioning plans; lack of published dismantlement plans or sites for long-term post-dismantlement reactor storage; the need to build a new storage site for decommissioning waste, adding to back-end costs; and little discussion of security – security measures for an FNPP differ from those for a traditional NPP in terms of transport and installing a traditional perimeter control zone around the plant.

The two KLT-40S reactors and a steam generating unit were installed in the hull during the last quarter of 2013 at the Baltiysky Zavod shipyard in St Petersburg. The operation to install the reactors, carried out using a special floating crane,

Box 5.2 Technical aspects of the *Akademik Lomonosov* FNPP

The KLT-40S reactor is of a compact modular design, i.e., the reactor and other heat transport components are connected with short nozzles. The reactor working conditions are similar to conventional PWR design (IAEA 2004) and although the reactor has been developed as barge-mounted it can also be based on land (OECD-NEA 2011). The capacity of the KLT-40S reactor is 38.5 MW electric and 150 MW thermal.

The core design will make use of LEU fuel. Although the exact configuration of the fuel is unknown, it is likely that UO2 granules in the silumin matrix will be used with fuel cladding made up of zircalloy (Egnatuk 2013). Refueling of the whole core would be completed at the end of each fuel cycle. However, fuel bundles are expected to be shuffled within the core every two years (OECD-NEA 2011).

Reactor safety is ensured by a combination of both active and passive decay heat removal systems. Five radioactive release barriers are incorporated: fuel matrix, fuel cladding, primary circuit, containment, and reactor compartment protective enclosure. The core damage frequencies (CDF) with respect to internal events are 10^{-7} per reactor year; however, the cumulative CDF for combined internal and external events is calculated to be 10^{-5} per reactor year (IAEA 2009). These probabilistic safety assessment results are in accordance with the IAEA recommendations for land-based NPPs.

was supervised by Rosenergoatom and the Russian Maritime Register of Shipping (IAEA 2009). The plant is now scheduled for delivery in September 2016 and will be deployed near the port of Pevek on the Chukotka peninsula in the East Siberian Sea (IAEA 2009).

5.3.1.2 Accident scenario in the Barents Sea involving an FNPP

The *Shtokman* field, one of the world's largest natural gas fields, lies in the northwestern part of the South Barents Basin in the Russian sector of the Barents Sea, 600 km north of the Kola Peninsula. Drilling operations in the *Shtokman* field have created an urgent need for a reliable local energy source. Owing to the remote location of the field this poses some challenges. An FNPP could be a good solution. Under the umbrella of the CEEPRA (Collaboration Network on EuroArctic Environmental Radiation Protection and Research) network a hypothetical accident was simulated for an FNPP located in the *Shtokman* field. The Finnish Meteorological Institute (FMI) and the Norwegian Meteorological Institute (MET) used atmospheric modelling to simulate the ^{131}I activity concentration in air and the ^{137}Cs activity concentration at ground level resulting from such an accident. The main aim was to quantify the radionuclide distributions that could reach northern Europe. The results of these simulations were published by CEEPRA (2014).

Model runs were performed separately in both institutes, using similar source terms and other conditions. The meteorological conditions however could vary per model run. The Murmansk Marine Biological Institute based their radioecological marine modeling in the case of an FNPP accident on the FMI atmospheric modelling results (CEEPRA 2014). The hypothetical FNPP accident was situated in the *Shtokman* gas field. The reactor was thought to be an icebreaker-type reactor KLT-40 with an average burn-up of 78,000 mega-watt days (MWd), about 466,000 MWd/ton of heavy metal. The typical fuel for a KLT-40 type of reactor is 150 kg ^{235}U with an enrichment level of 90%. The duration of release after shutdown was set at two hours, amounting to a discharge of 1% of the radionuclides in the plant. The effective release height was set 100 m above sea level.

The model predicted the ^{131}I concentration in ambient air and the activity concentration of ^{137}Cs at ground level. The results were used to assess an employee's total dose from both external and internal exposure. The impacts studied were the effects of the accident on the marine environment and on the nearest human settlements on the mainland. Numerical weather forecasts for the simulation were obtained from the European Centre for Medium-Range Weather Forecasts (ECMWF). The dispersion model used was the SILAM and dose calculations were based on the Technical Research Centre of Finland's (VTT's) dose calculation model (VALMA).

For certain meteorological conditions, model results showed that radionuclides released from the hypothetical accident at the FNPP could reach northern Scandinavia. The results of the FMI and MET dispersion models were compared for similar meteorological conditions, for an accident starting on 4 November 2010. The distributions of ^{137}Cs activity concentrations, as calculated by the SILAM model from the FMI and the SNAP model from the MET are shown in Fig. 5.9.

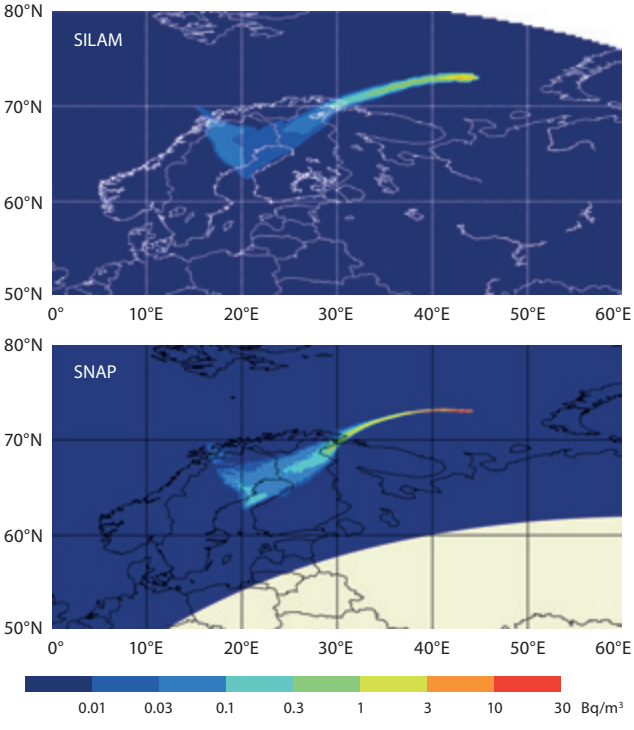


Figure 5.9 Average ^{137}Cs activity concentration, 2 days after a hypothetical floating nuclear power plant reactor accident in the Barents Sea on 4 November 2010. Finnish Meteorological Institute results (SILAM simulation) and Norwegian Meteorological Institute results (SNAP simulation).

The concentration patterns are very similar for both models. This is also the case for other isotopes. In general, the shape of the radioactive plume resulting from the accident is similar for both models. Small, local differences can be observed in the magnitudes of calculated concentration and deposition.

The results of both models indicate that the effects of the FNPP accident would be mainly local. Workers at the plant would be affected, but the activity concentrations in the emission plume reaching the mainland appear to be very low under this scenario. These activity concentrations would contribute only a fraction of the annual radiation exposure for people living in the area (CEEPRA 2014).

5.3.2 Nuclear-powered icebreakers

Russia currently has the biggest icebreaker fleet in the world, and the only one that includes nuclear-powered icebreakers. An advantage of nuclear-powered ships in the Arctic is that like nuclear-powered submarines, they are able to operate for long periods without the need to return to port to refuel. Recent reports have also noted renewed interest in the United States for utilizing nuclear power for icebreakers[5].

However, the Russian icebreaker fleet is aging with most of its icebreakers now more than 20 years old. According to some reports, if new shipbuilding projects are not realized, the *50 Let Pobedy*, built in 2007, will be the only Russian nuclear icebreaker still in operation by 2021. Under a new build plan instigated by *Atomflot*, the United Shipbuilding Company started building what is to become the biggest and most powerful icebreaker in the world in November 2013 at the Baltiysky Zavod shipyard near St. Petersburg. Referred to as Project 22220 and named *Arctica*, the vessel will be 173m long with a beam of 34m, and powered by two nuclear engines yielding 60 MW of power, enabling the vessel to navigate ice up to 3m thick. The ship is due to be ready to enter service in 2017–2019. At least one more ship in this class has been proposed.

Although largely self-reliant, a nuclear powered icebreaker or any other form of FNPP for use in the Arctic will require additional infrastructure capable of handling fuel, spent fuel and radioactive wastes. Increasing the number of such sources in the area will increase the risk of contamination events over time.

5.4 Conclusions and recommendations

That traces of radioactivity originating from the FDNPP accident were detected across the entire northern hemisphere after only fifteen days, emphasizes how an accidental release of radioactivity from a nuclear power plant located far from the Arctic can still result in measurable fallout across the Arctic region. Although all radiation levels measured outside Japan have been very low and represent a negligible public and environmental hazard, the number and age of nuclear power plants situated much closer to the Arctic give cause for concern that an accidental release closer to the AMAP area could result in considerable impacts on the region. This highlights the importance of atmospheric monitoring stations in the AMAP region continuing to **monitor radioactivity in the lower atmosphere and at ground level, in order to detect routine emissions from nuclear facilities and so provide early warnings of major nuclear events**.

Although environmental monitoring indicates that FDNPP-derived radioactivity in the Arctic environment currently poses a negligible risk to Arctic residents, including those following a traditional diet, there is clearly a need to **ensure good communication between scientists, the general public and policy makers should this situation change**.

The potential for an accidental release of radioactivity from existing sources, as well as new sources planned for the coming decade, highlights the need to **update the existing AMAP assessments of radioactivity in the Arctic at regular intervals and to maintain and improve monitoring programs in order to establish reliable baseline data.**

[5] https://www.foreignaffairs.com/articles/united-states/2014-12-11/breaking-ice

6. Naturally-occurring radioactive material

Authors: William Standring, Frits Steenhuisen, Jing Chen, Ari-Pekka Leppänen, Louise Kiel Jensen, Kristine Thrane

Key findings

- *Increased exploitation of resources in the Arctic is expected to enhance the risk of releasing NORM/TENORM to the environment within the AMAP region*
- *Environmental impact assessments prior to the start of any undertaking that could lead to radioactive contamination of the Arctic environment are very important from a radiological perspective*
- *Climate effects (principally warming) on the behavior of NORM/TENORM in the Arctic that could affect human health should be a subject for further research*

6.1 Introduction

Naturally-occurring radioactive material (NORM) and technologically-enhanced naturally-occurring radioactive material (TENORM) have been topics of interest in previous AMAP assessments (AMAP 1997, 2010b; see also Box 6.1) especially regarding the oil and gas industries and their development within the Arctic. Although the information reported in previous AMAP assessments is still largely relevant oil and gas exploitation in Arctic areas is expected to increase with the opening up of the Arctic under a warmer climate. Future oil and gas activity in the Arctic region is also highly dependent on developments in the world energy market. Uranium mining is another source of TENORM. Although there is no uranium mining in the Arctic at present, the potential exists for future activity in Canada and Greenland. As a consequence, it is extremely difficult to predict the geographical areas likely to be affected by TENORM over the coming decades and the magnitude of the associated effects.

Box 6.1 NORM and TENORM

NORM refers to any naturally-occurring radioactive material. The main radionuclide of concern in NORM is ^{226}Ra, including subsequent daughter radionuclides. Naturally-occurring radioactive materials are referred to as TENORM – *technologically-enhanced* NORM – when the radionuclides that occur naturally in soils and rock formations and consequently in the water that comes into contact with them are concentrated or exposed to the environment by activities such as oil and gas extraction and uranium mining (Heaton and Lambley 1995). Radionuclide concentrations in TENORM are often orders of magnitude higher than in the parent materials. The majority of radionuclides in TENORM are found in the uranium and thorium decay chains.

6.2 Sources and environmental impacts of TENORM

6.2.1 Oil and gas activities in the Arctic

Naturally-occurring radioactive elements in bedrock leach into the water present in underground hydrocarbon reservoirs and some of this is extracted together with the oil and gas and becomes part of the well stream processed on board installations. This is known as 'produced water'. Produced water has been described as the largest volume waste stream in the exploration and production process for oil and gas (Stephenson 1992) and concerns over the potential radiological impacts on marine biota have been raised. Activity concentrations of TENORM in produced water can be up to a thousand times higher than in seawater (NRPA 2004a). Hydraulic fracturing ('fracking') used to extract terrestrial shale oil and gas also produces large amounts of liquid wastes containing TENORM (Brown 2014). In addition to liquid wastes, radioactive substances can also accumulate in the solid wastes produced from both petroleum drilling and fracking. These include the substances incorporated within the scale that accumulates on process equipment, as well as those in sand, muds or gravel, potentially leading to additional radioactive contamination issues with regard to the storage and handling of such waste. ^{226}Ra, ^{228}Ra and ^{210}Pb, and their associated daughter radionuclides, are the most commonly occurring radionuclides in produced water and solid wastes.

The Arctic region is currently seeing an expansion in oil and gas exploitation activities. In addition to the increasing amount of production water extracted from existing fields as they age (AMAP 2010a), this increase in exploitation will also lead to larger volumes of produced water and other wastes, resulting in an enhanced source of naturally-occurring radioactivity in the Arctic. Arctic areas where significant resources have been identified and further development may occur include the Kara and Barents Sea regions (Kontorovich et al. 2011), the Eastern Arctic and the Alaskan North Slope (Houseknecht et al. 2012) (see also Box 6.2). Of the 16 petroleum fields in production in the Norwegian Sea, five discharge produced water into the surrounding waters. The Norwegian Government has adopted a zero discharge policy for harmful wastes from the petroleum industry, though some discharges of produced water are allowed. Field-specific discharges of radionuclides are available for all fields and in the Norwegian and Barents Seas, 22.7 million Sm^3 (standard cubic meter at 15°C and 101.325 kPa) of produced water were discharged in 2014, containing 43.6 GBq ^{226}Ra and 45.8 GBq ^{228}Ra.

A review of possible impacts on marine biota following exposure to produced waters was undertaken by Hosseini et al. (2012). They assessed some of the few studies available on this topic but could draw no firm conclusions about detrimental effects on biota from TENORM released in produced water. In general, findings appear to indicate that

Box 6.2 Potential for increased oil and gas activity in the Arctic

The first Arctic field to be developed was *Norman Wells* in the Mackenzie Valley (Canada), where oil has been produced commercially since the 1920s. It was not until the late 1960s that production started in other Arctic regions. By 2010, Arctic production accounted for as much as 10% and 25% of the world's total oil and gas production, respectively (numbers from AMAP 2010a). Four countries currently produce oil and gas from their Arctic territories: USA (Alaska), Canada, Norway and Russia. Each of these countries could show increased oil and gas activities over the coming decades.

USA (Alaska): In 2012, the U.S. Geological Survey published estimates of Alaska's North Slope shales inventory – potentially 80 trillion cubic feet of shale gas and as much as 2 billion barrels of shale oil (Houseknecht et al. 2012). This value was based on estimates from oil and gas extraction from similar formations in other US territory areas.

Canada: Three fields produced oil and/or gas in the Northwest Territories in 2012: *Norman Wells* in the central Mackenzie Valley, *Ikhil* on the Mackenzie Delta, and *Cameron Hills* southwest of Hay River in the southern Northwest Territories. The total aggregate oil production in 2012 was 4.8 million barrels of oil (a 24% increase on 2011). Total aggregate natural gas production in the Northwest Territories in 2012 was 162.8 million cubic meters, about the same as in 2011. As of 2015, about 25% of Canada's discovered resources of conventional petroleum are in the North and remain undeveloped. The Mackenzie Valley gas pipeline, scheduled for completion in 2014 and expected to increase production, ran into funding difficulties and is yet to be completed.

Norway: Norway has extensive marine oil and gas production within the AMAP area, with the northernmost fields being *Snøhvit* (gas) and *Goliat* (oil) located near the southwestern entrance to the Barents Sea. *Snøhvit* has been in operation since 2007, while production at *Goliat* started in 2016. Several other large gas fields have been discovered in the Barents Sea but none are currently being developed. A further 16 petroleum fields are in production in the Norwegian Sea.

Russia: The Russian Arctic contains huge hydrocarbon and mineral reserves. Oil- and gas-bearing regions cover about 90% of the total shelf area of Russia; some 5.2–6.2 million km^2. Approximately 2 million km^2 of this area is located in the western Arctic on the shelves of the Barents and Kara Seas, where there are reported potential hydrocarbon resources of 335 BBOE (billion barrels oil equivalent) (Kontorovich et al. 2011). For example, Rosneft (www.rosneft.com) launched projects in the Kara and Barents Seas in 2010 after obtaining four licenses to explore Russia's Arctic shelf. Three relate to designated areas in the Kara Sea (East Prinovozemelsky 1, 2 and 3) and the fourth is for the South-Russky block in the Pechora Sea. These areas are estimated to hold 21.5 billion tons of oil equivalent (=150 BBOE). Also known to contain exploitable oil and gas reserves are the Laptev Sea, the East Siberian and Chukchi seas and the Eastern Arctic, as well as the Timan-Pechora, Yenisei-Laptev, Barents-Kara and Indigirka-Chukotka provinces, and the South Yamal, Lena-Anabar and Anadyr regions. As of August 2014, Rosneft and Statoil Petroleum AS started exploration operations at the Pingvin License PL713 prospect on the Norwegian continental shelf in the Barents Sea. The Pechora and Barents Sea shelves contains 11 fields, including four oil fields (*Prirazlomnoe*, *Dolginskoe*, *Varandeyskoe*, *Medinskoe*), three gas fields (*Murmanskoe*, *Ludlovskoe*, *Severo-Kildinskoe*), three gas condensate fields (*Shtokman*, *Pomorskoe*, *Ledovoe*) and one oil and gas condensate field (*Severo-Gulyaevskoe*) (www.arctic-info.com). Exploration of the Nenets Autonomous Region has also revealed considerable reserves of oil, gas and gas condensate.

risk to the environment from TENORM is negligible, although the uncertainties regarding the nature and magnitude of different sources of produced water indicate that further study on this topic is needed. Especially because it is clear that the Arctic will probably become a more important part of the supply of oil and gas reserves for several Arctic countries in the future, with a considerable increase in inputs of TENORM thus likely. Another issue concerning produced waters that requires further study is the mixture of chemicals often present in the produced waters from the additives used during oil/gas production. Previous AMAP assessments (AMAP 1997, 2002, 2010) recommended robust environmental assessment when planning any undertaking that could potentially lead to radioactive contamination of the Arctic environment. Such assessments are essential for predicting the environmental consequences of any development project involving discharges of TENORM and ensuring that potential problems are foreseen and addressed at an early stage in project planning and design.

6.2.2 Uranium mining in the Arctic

Other extraction industries that may be a source of TENORM in the Arctic environment are mining for uranium, other metals and phosphate (Kola Peninsula). When mining for uranium, the ore is generally extracted from the ground via either open-pit or underground mining operations. Once mined, the uranium ore will undergo a range of processing activities to produce a physical or chemical concentrate known as 'yellow cake'. These activities include uranium milling, where the ore is crushed and ground to a uniform particle size, followed by chemical leaching to extract and concentrate the uranium. The uranium milling process results in 'tailings'. The tailings, a general term for the waste products from mining extraction, contain several naturally-occurring radioactive elements, including uranium, thorium, radium, lead and polonium, and are also a source of radon in the atmosphere, which is released following the decay of ^{226}Ra. Tailings from uranium mining may contain other chemically hazardous elements, including arsenic and can be acidic as a result of leaching processes. Tailings are stored in specially designed waste management facilities. Leakage from such facilities could become a source of TENORM in the Arctic. Leakage is likely to have a greater impact on terrestrial environments than than marine environments due to the greater dilution capacity of the marine environment. The effects of discharges from uranium mining and milling have been studied in non-Arctic areas (e.g. Hierro et al. 2013; Bister et al. 2015), but whether the accumulated knowledge is

directly applicable to Arctic conditions is not known. Multiple stressor effects from uranium mining and milling activities in the Arctic should also be considered.

6.2.2.1 Canada

The only uranium mine in the Canadian Arctic, Port Radium, stopped operating in 1982. This site is subject to long-term monitoring and the results are regularly published. Further information is available in Chap. 4. Although no uranium mining activity currently takes place in the Canadian Arctic, a recent policy statement by the Government of Nunavut (Government of Nunavut 2012) opens the door to possible future development in that it "*…recognizes that uranium mining is of concern to some Nunavutummiut, but that, when properly managed, can be of benefit to our territory.*"

6.2.2.2 Greenland

Uranium exploration campaigns took place in many areas of Greenland between the early 1950s and early 1980s. Several uranium anomalies were identified and nearly 30 uranium deposits are known today, some with large potential. However in 1985, the Danish Government shifted its energy policy strategy from one based on nuclear power to one based on conventional fossil fuels and in 1988 imposed a ban on the issue of uranium exploration or exploitation licenses in Greenland. This was later formulated as the 'zero-tolerance' policy. In 2013, after gaining full control over its natural resources under the Act on Greenland Self-Government, the Greenland parliament lifted the ban on uranium exploration (Naalakkersuisut 2014). Although there has been no uranium mining in Greenland to date, an application for an exploration license is likely to be submitted to the Greenlandic Government in the near future (see Box 6.3).

6.3 Human exposure to Arctic residents from NORM/TENORM

6.3.1 Radon and its decay products

Radon (^{222}Rn) is a naturally-occurring radioactive gas generated by the decay of uranium-bearing minerals in rocks and soils. Exposure to indoor radon has been identified as the second leading cause of lung cancer after tobacco smoking. Radon also plays an important role in NORM levels found in the Arctic. Activity concentrations of uranium and its decay products (also known as uranium daughters) in the overlying soils and bedrock vary widely from place to place depending on geological characteristics. As these radionuclides decay radon is emitted. Uranium is ubiquitous in the environment and so radon is continuously released from the ground. The radon (half-life of 3.8 days) released is then transported into the surrounding atmosphere via diffusion and advection and as radon is chemically inert only a small proportion of the radon present in the air column is removed by chemical processes. Because radon is naturally present in the air, simply by breathing, people everywhere are exposed to radiation from both the radon itself and from the short-lived radon progeny[6].

Box 6.3 Uranium exploration at Kvanefjeld (*Kuannersuit*)

In 1955, systematic uranium exploration was initiated in southern Greenland by a consortium comprising the Grønlands Geologiske Undersøgelser (Greenland Geological Surveys), the Atomic Energy Commission, and Kryolitselskabet Øresund A/S. Investigations focused on the Mesoproterozoic Ilímaussaq alkaline complex in the Gardar Province. In 1956, a highly radioactive zone was defined in part of the intrusion referred to as Kvanefjeld (in Greenlandic, *Kuannersuit*). Kvanefjeld is located 7 km from the coastal town of Narsaq in southwest Greenland. Kvanefjeld is a low-grade uranium occurrence, but spans a very large area making it the largest of the known uranium occurrences in Greenland. It is a unique type of uranium deposit with the majority of uranium hosted by the phosphor-silicate mineral steenstrupine, which contains 0.2–0.5% UO2. Prior to the introduction of the zero tolerance policy by the Danish Government, detailed geological mapping and radiometric surveys were carried out, about 11 km of core were drilled and a 1 km long adit was constructed for bulk-sampling to supply metallurgical test-work undertaken by the National Laboratory Risø.

Since 2007, the area has been explored by Greenland Minerals and Energy Ltd for rare earth elements (REE) and by-products of uranium, zinc and fluorine, as well as for metallurgical routes. Since the renewed exploration, additional REE (and uranium) resources have been assessed in two other bodies of the Ilímaussaq complex; the Sørensen Zone and Zone 3. Greenland Minerals and Energy Ltd reports that the inferred tonnage implies that it is the fifth largest uranium occurrence in the world. However, the company's intention is to start the operation based on the Kvanefjeld occurrence, consisting of 437 million tonnes of ore grading 274 ppm uranium. If the mine becomes a reality, the project will comprise an open pit mine, a processing plant, a tailings facility, a port, mine accommodation and roads connecting the different parts of the project. The concept for the processing plant is not yet decided, but may include facilities for production of RE-carbonates and yellow-cake. A number of alternative locations for tailings storage (e.g. valley deposition, lake deposition and deposition in the fjord system/deep sea) are being evaluated. Greenland Minerals and Energy Ltd has conducted baseline studies in the Kvanefjeld and Narsaq area since 2007 to establish the natural levels of specific metals and radioactivity. Environmental impact and social impact assessments are underway and expected to be submitted to the Greenlandic government, along with an application for an exploration license during 2014/2015.

[6] The term radon is used generically to indicate both radon and its short-lived decay products (^{218}Po, ^{214}Po, ^{214}Pb and ^{214}Bi). The progeny contribute most of the dose to lung tissue when radon is inhaled.

The decay of radon also produces longer-lived radionuclides, including one with more environmental significance: ^{210}Pb (half-life 22 years). Radon daughter radionuclides are all solid reactive particles which can readily attach to other particles in the air and be transported long distances before deposition back to the earth's surface. For this reason, ^{210}Pb is widely distributed and accumulates in the Arctic as elsewhere. Lead-210 decays via the β-emitting radionuclide ^{210}Bi to an α-emitting radionuclide ^{210}Po (half-life 138 days). The latter is of radiological significance because the effective dose to man from an uptake of the same amount of both radionuclides is approximately twice as much for ^{210}Po. Indeed, ^{210}Po is considered one of the most hazardous radioactive materials for internal exposure (ingestion or inhalation). As ^{210}Po has a much shorter half-life than ^{210}Pb, it is continually produced from the grandparent radionuclide and so, because ^{210}Pb can be atmospherically transported over long distances before being deposited and thereby accumulated in the Arctic, the same is also true for ^{210}Po. Accumulation of ^{210}Po in biota is an important link in the transfer of this highly toxic radionuclide to Arctic residents: lichen → reindeer/caribou → human.

Levels of natural background radiation vary geographically. Concerning radon – a significant source of NORM in the environment – low levels could be expected in the Arctic due to the extensive areas of glaciers and permafrost preventing its release from the ground. But monitoring data instead indicate high levels of natural background radiation in the Arctic. However, these elevated levels are not due to local geology but to the accumulation of NORM transported in from areas outside the Arctic.

This long-range atmospheric transport of naturally-occurring radionuclides has created a strong and reversed seasonal pattern in aerosol radioactivity concentration in the Arctic, generally characterized by a winter maximum and a summer minimum. Since the long-lived radon decay product ^{210}Pb is the most common particulate monitored at almost all aerosol monitoring stations worldwide, the significance of natural radionuclides transported to the Arctic can be quantified by the ratio of ^{210}Pb in winter over ^{210}Pb in summer (Chen 2014). A recent study analyzed long-term atmospheric monitoring data at the Canadian High Arctic, sub-Arctic and mid-latitude sites, and demonstrated that the annual average atmospheric ^{210}Pb concentration could be up to four-fold higher than the local background level in the low Arctic and up to six-fold higher in the High Arctic (Chen 2014).

The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) has estimated that the radiation dose from indoor radon exposure represents about half the dose from exposure to all natural sources of ionizing radiation (UNSCEAR 2009). In an indoor environment, there are many factors affecting radon concentrations and these could differ in the Arctic compared to other world regions. A survey of indoor radon characteristics and associated radiation doses in all health regions across the Canadian Arctic also showed average radon concentrations, such as percentages of homes above 200 Bq/m^3 to vary widely from one Arctic region to another (Health Canada 2012). For example, among the five Arctic health regions in Canada, indoor radon exposure is lower in Nunavut and higher in Yukon Territory. The population weighted average percentage of homes above 200 Bq/m^3 in the Arctic health regions was 7.1%, which was comparable to the Canadian national average of 6.9%, however. The local level of radon in the air can be increased by industrial activities, such as mining and fracking, often due to dumped waste products.

As is the case globally, the Arctic has experienced a warming trend over the past century. A full account of temperature changes in the Arctic region was published recently by AMAP (Walsh et al. 2011) and the Intergovernmental Panel on Climate Change (IPCC 2013). The increase in average temperature since 1980 has been twice as high over the Arctic as it has been over the rest of the world, with most of the warming occurring in autumn and early winter. Arctic air temperature is expected to continue to rise, with an increase in average autumn and winter temperatures of 3–6°C projected for the late 21st century (even when modelling scenarios use lower greenhouse gas emissions than have been recorded over the past ten years). Climate-driven changes in the behavior of NORM/TENORM that could affect human health should be a subject for further research. For example, permafrost degradation under a warmer climate could lead to enhanced radon release.

6.3.2 Consumption of local foods

When considering the radiation doses received by the general public following consumption of foods from Arctic regions, naturally-occurring radionuclides make a greater contribution than anthropogenic radionuclides. As shown in previous chapters, activity concentrations of anthropogenic radionuclides and thus the dose received by the general public were very small in 2013. For example, activity concentrations of ^{137}Cs (an anthropogenic radionuclide) in Barents Sea fish were typically below 0.5 Bq/kg fw (fresh weight), whereas those of naturally-occurring ^{210}Po were typically 0.2–0.8 Bq/kg fw and those of naturally-occurring ^{210}Pb typically 0.08–0.12 Bq/kg fw. Although the activity concentrations are very similar, the dose conversion factors are very different due to the type of radiation emitted by these nuclides (ICRP 1996). Figure 6.1 shows the effective dose to man obtained from Barents Sea fish as a function of consumption – by far the highest dose was caused by naturally-occurring ^{210}Po.

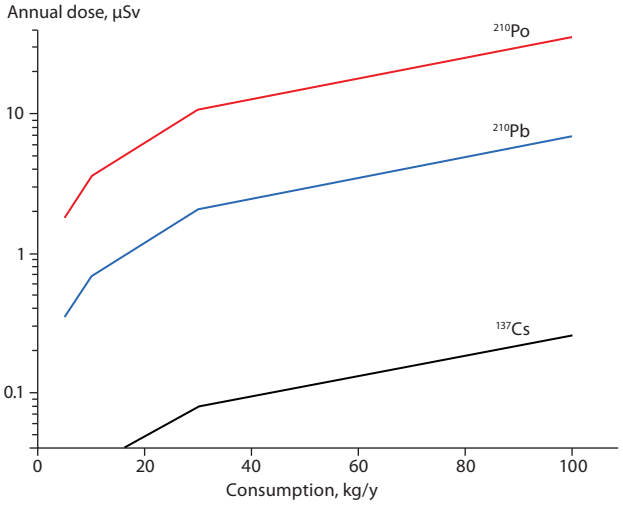


Figure 6.1 Effective dose to man from Barents Sea fish as function of fish consumption (Leppänen et al. 2013a).

Comparing doses from (anthropogenic) ^{137}Cs and (naturally-occurring) ^{210}Pb and ^{210}Po gives a ratio of 0.006–0.007. This means that less than 1% of the total ingestion dose obtained from Barents Sea fish comes from the anthropogenic ^{137}Cs. Even for a person consuming Barents Sea fish on a daily basis, it is clear from Fig. 6.1 that the total ingestion dose is still below 20 μSv/y and that a very large annual fish consumption would only contribute a negligible amount to the typical annual dose in a human in the region. It should also be noted that the activity concentrations of naturally-occurring ^{210}Po and ^{210}Pb in Barents Sea cod and haddock are very low compared, for example, to sardine from the North-East Atlantic where the observed ^{210}Po level of 66 Bq/kg fw is roughly 100 times higher than for Barents Sea cod. There is, however, potential for the concentration of naturally-occurring radionuclides in Arctic foods to rise as a result of human activities, including the expansion of the oil and gas industry and the start of other activities including uranium mining and fracking. This could, in turn, increase the radiation doses from NORM to people in the Arctic regions.

6.4 Conclusions and recommendations

TENORM – *technologically-enhanced* naturally-occurring radioactive materials – occurs when radionuclides that occur naturally in soils and rock formations and consequently in the water that comes into contact with them are concentrated or exposed to the environment. Within the Arctic, activities such as oil and gas extraction and uranium mining result in TENORM. Radionuclide concentrations in TENORM are often orders of magnitude higher than in the parent materials.

Although oil and gas exploitation in Arctic areas is expected to increase with the opening up of the Arctic under a warmer climate, predicting the location and extent of Arctic areas likely to be affected by TENORM over the coming decades is extremely difficult, not least because future oil and gas activity in the Arctic region is highly dependent on developments in the world energy market. Produced water is the largest volume waste stream in the exploration and production process for oil and gas and concerns over the potential radiological impacts of TENORM on marine biota have been raised. **Although findings suggest that risk to the environment from TENORM is negligible, uncertainties regarding the nature and magnitude of the different sources of produced water suggest further study is needed.**

Another extraction industry that may be a future source of TENORM in the Arctic environment is mining for uranium. Despite no uranium mining activity in either Greenland or the Canadian Arctic at present, this could change in the future. Tailings are the main source of TENORM from uranium mining. Tailings are stored in specially designed waste management facilities. Leakage from such facilities could become a source of TENORM in the terrestrial Arctic. **Potential impacts from uranium mining and milling are not well understood in the Arctic and more research is needed**.

The local level of radon in air, a naturally-occurring radioactive gas generated by the decay of uranium-bearing minerals in rocks and soils and the second leading cause of lung cancer after tobacco smoking, can be increased by industrial activities such as mining and fracking, often due to dumped waste products. Long-range transport is also an important source of naturally-occurring radionuclides in the Arctic. In fact, a recent study found the annual average atmospheric concentration of ^{210}Pb (a decay product of radon) to be up to four-fold higher than local background levels in the Low Arctic and up to six-fold higher in the High Arctic. The effective dose to man following the consumption of local foods is greatest from the relatively short-lived but highly toxic ^{210}Po (a decay product of ^{210}Pb). **Climate effects (principally warming) on the behavior of NORM/TENORM in the Arctic could have implications for human and environmental health and this requires further research.**

7. Conclusions

This assessment provides new insights and updated information concerning actual and potential sources of radioactive contamination in the Arctic. Levels of anthropogenic radioactivity in the Arctic attributable to previously identified releases are low and generally declining. This concerns releases associated with atmospheric nuclear tests conducted in the 1950s and 1960s, nuclear fuel reprocessing, historical dumping of radioactive waste and, more recently, accidents at nuclear power plants such as in Chernobyl in 1986 and Fukushima in 2011. The risks posed to health from radioactivity in the Arctic are therefore generally decreasing. This is predominantly due to the natural decay of radionuclides previously released into the environment.

Nevertheless, the Arctic remains vulnerable to radioactive pollution transported from distant sources, whether by ocean currents or movement of air masses in the atmosphere. Studies indicate that specific sources (such as radioactive waste dumped in the Barents and Kara seas) have the potential to increase contamination levels at a local and possibly regional level.

Transfer pathways for certain radionuclides in the Arctic terrestrial environment can result in elevated human exposures. The Arctic continues to have a high density of sources of radioactive contamination, due to historical dumping of radioactive waste in some areas of the Russian Arctic and incomplete decommissioning of nuclear equipment and facilities, with inadequate storage conditions for radioactive waste at some sites.

Monitoring the levels of radionuclides within the Arctic environment is a core component of the AMAP program and all eight Arctic countries have ongoing national monitoring programs to establish long-term trends in levels of anthropogenic radionuclides. Monitoring provides valuable baseline information as well as early warning of new releases of radionuclides. The most recent results from national monitoring programs across the Arctic indicate that activity concentrations of anthropogenic radionuclides in the Arctic Ocean are low compared to levels measured between the 1970s and 1990s and are still decreasing, and that the same decreasing trend in activity concentrations is observed in the terrestrial and freshwater environments.

Some of these monitoring systems registered an increase in radioactive contamination associated with releases following the catastrophic accident at the Fukushima Daiichi nuclear power plant in 2011. Although the radioactive impact of this event on the Arctic has so far proved minimal and of no concern to human health, the accident serves as a reminder that the Arctic is not isolated from the rest of the world – and that a nuclear accident thousands of miles away can pose a risk to the region. ***This highlights the importance of continuing to monitor radioactivity in the lower atmosphere and at ground level, in order to detect routine emissions from nuclear facilities and so provide early warnings of major nuclear events.***

Within the Arctic, activities such as oil and gas extraction and uranium mining result in technologically-enhanced naturally-occurring radioactive materials (TENORM). Although findings suggest that risk to the environment from TENORM associated with the oil and gas industry is negligible, ***further study is needed on the nature and magnitude of the different sources of produced water.*** Although there is no uranium mining activity in either Greenland or the Canadian Arctic at present, ***potential impacts from uranium mining and milling are not well understood in the Arctic and more research is needed.***

An additional and growing concern is the extent to which climate change – mainly through changes in the hydrological cycle (such as thawing permafrost and declining snow cover) and increasing frequency of wildfires – could affect the remobilization of radioactivity within the Arctic. Climate change could also influence the extent of releases of technologically-enhanced naturally-occurring radioactive material (TENORM), given the expected increase in Arctic resource exploitation under a warming climate. Warming could also result in a widespread and substantially increased radiation dose to Arctic residents from exposure to the radioactive gas radon and its daughter radionuclides.

Although several of the risks associated with radioactive contamination in the Arctic have been reduced in the past five to ten years, ***there is still a clear need for continued monitoring of radioactivity in the Arctic, especially in relation to the large amounts of radioactivity previously dumped at sea, the potential for future accidents at nuclear facilities, and the possible effects of climate change on the remobilization of radioactivity.***

The present assessment also documents the good progress that has been made in mitigating risk from potential radioactive contamination in northwestern Russia that have been highlighted in previous AMAP assessments. This includes decommissioning radioisotope thermoelectric generators (RTGs) and nuclear submarines; the nuclear waste vessel *Lepse;* and managing and remediating sites of temporary storage for radioactive wastes at Gremikha and Andreeva Bay. Nearly all the spent nuclear fuel has now been removed from Gremikha and although the decommissioning work at Andreeva Bay is not as advanced, with most of the spent nuclear fuel and radioactive waste still on-site, decommissioning work at Andreeva is progressing, albeit slowly. The long-term consequences of the various decommissioning activities in northwestern Russia for Mayak PA and its surrounding environment remain uncertain, however.

The accidental release of radioactivity from the Fukushima Daiichi nuclear power plant caused only very low levels of additional radionuclides throughout the Arctic region and is not expected to result in adverse impacts on Arctic biota or Arctic residents consuming a traditional diet. However, it has highlighted the potential for many nuclear accident scenarios, including those created by geohazards to affect the Arctic region. The possibility of a future accidental release of radioactivity from existing sources, as well as new sources

planned for the coming decades, ***highlights the need to update existing AMAP assessments of radioactivity in the Arctic at regular intervals and to maintain and improve monitoring programs in order to establish reliable baseline data.***

Continued cooperation between the Arctic nations on radioactivity issues is essential and should be strengthened, especially in relation to environmental monitoring and remediation activities.

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Acronyms

AMAP	Arctic Monitoring and Assessment Programme
CA	Controlled Area
CEEPRA	Collaboration Network on EuroArctic Environmental Radiation Protection and Research
CTBTO	Comprehensive Nuclear-Test-Ban Treaty Organization
dw	Dry weight
EBRD	European Bank for Reconstruction and Development
FDNPP	Fukushima Daiichi nuclear power plant
FMBC	Federal Medical-Biological Center (Russia)
FMI	Finnish Meteorological Institute
FNPP	Floating nuclear power plant
fw	Fresh (wet) weight
HAL	Highly Active Liquor
HAST	Highly Active Storage Tank
HPZ	Health Protection Zone
IAEA	International Atomic Energy Agency
JNREG	Joint Norwegian-Russian Expert Group
MCIC	Mining and Chemical Industrial Combine, Zheleznogorsk
MDA	Minimum detectable activity
NORM	Naturally-occurring radioactive material
NPP	Nuclear power plant
NRPA	Norwegian Radiation Protection Authority
PNE	Peaceful nuclear explosion
ROV	Remotely operated vehicle
RTG	Radioisotope thermoelectric generator
SA	Supervision Area
SCC	Siberian Chemical Combine
SNF	Spent nuclear fuel
STS	Site of Temporary Storage
STUK	Radiation and Nuclear Safety Authority (Finland)
TENORM	Technologically-enhanced naturally-occurring radioactive material

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