# AMAP Assessment 2009:

## Radioactivity in the Arctic

**Arctic Monitoring and Assessment Programme (AMAP)**

AMAP

## Abbreviations

| | |
|---|---|
| ACIA | Arctic Climate Impact Assessment |
| AMAP | Arctic Monitoring and Assessment Programme |
| AMOC | Atlantic meridional overturning circulation |
| Bq | Becquerel |
| Ci | Curie |
| $CO_2$ | Carbon dioxide |
| CR | Concentration ratio |
| DCC | Dose conversion coefficient |
| DL | Detection limit |
| DOC | Dissolved organic carbon |
| dw | Dry weight |
| EARP | Enhanced Actinide Removal Plant |
| EBRD | European Bank for Reconstruction and Development |
| EPIC | Environmental Protection from Ionizing Contaminants |
| ERICA | Environmental Risk from Ionising Contaminants: Assessment and Management |
| FMBA | Federal Medical-Biological Agency (Russia) |
| Gy | Gray |
| HAL | Highly active liquor |
| HLW | Vitrified high-level waste |
| IAEA | International Atomic Energy Agency |
| ICRP | International Commission on Radiological Protection |
| IPCC | Intergovernmental Panel on Climate Change |
| LLW | Low-level waste |
| MLW | Medium-level waste |
| MOX | Mixed oxide |
| NRPA | Norwegian Radiation Protection Authority |
| $O_2$ | Oxygen |
| PA | Production Association (Russia) |
| POP | Persistent organic pollutant |
| RAPS | Reference Animals and Plants |
| RTG | Radioisotope thermoelectric generator |
| Sev RAO | Federal State Unitary Enterprise (Russia) |
| SMP | Strategic Master Plan (Russia) |
| STS | Sites of Temporary Storage |
| STUK | Finnish Radiation and Nuclear Safety Authority |
| Sv | Sievert |
| TENORM | Technologically enhanced naturally-occurring radioactive material |
| UNSCEAR | United Nations Scientific Committee on the Effects of Atomic Radiation |
| ww | Wet weight |

## Main radionuclides discussed

| | |
|---|---|
| Am | Americium |
| Be | Beryllium |
| Cs | Cesium |
| I | Iodine |
| Pb | Lead |
| Po | Polonium |
| Pu | Plutonium |
| Ra | Radium |
| Sr | Strontium |
| Tc | Technetium |

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# Radioactivity in the Arctic

# Contents

# Preface

This assessment report details the results of the 2009 AMAP assessment of Radioactivity in the Arctic. It builds upon the previous AMAP radioactivity assessments that were presented in 1998* and 2002**.

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;
- to identify possible causes for the changing conditions;
- to detect emerging problems, their possible causes, and the potential risk to Arctic ecosystems including indigenous peoples and other Arctic residents; and
- to recommend actions required to reduce risks to Arctic ecosystems.

This report is one of the detailed assessment reports that provide the accessible scientific basis and validation for the statements and recommendations made in the AMAP State of the Arctic Environment report, 'Arctic Pollution 2009' that was delivered to Arctic Council Ministers at their meeting in Tromsø, Norway in April 2009. It includes extensive background data and references to the scientific literature, and details the sources for figures reproduced in the 'Arctic Pollution 2009'*** report. Whereas the 'Arctic Pollution 2009' report contains recommendations that specifically focus on actions aimed at improving the Arctic environment, 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, etc.

To allow readers of this report to see how AMAP interprets and develops its scientifically-based assessment product in terms of more action-orientated conclusions and recommendations, the 'Executive Summary of the Arctic Pollution 2009 Ministerial Report', which also covers other priority issues (Persistent Organic Pollutants, and Radioactivity), is reproduced in this report on pages vii to xii.

The AMAP assessment is not a formal environmental risk assessment. Rather, it constitutes a compilation of current knowledge about the Arctic region, an evaluation of this information in relation to agreed criteria of environmental quality, and a statement of the prevailing conditions in the area. The assessment presented in this report was prepared in a systematic and uniform manner to provide a comparable knowledge base that builds on earlier work and can be extended through continuing work in the future.

The AMAP scientific assessments are prepared under the direction of the AMAP Assessment Steering Group. The product is the responsibility of the scientific experts involved in the preparation of the assessment. Lead countries for this AMAP Radioactivity Assessment were Norway and Russia. The assessment is based on work conducted by a large number of scientists and experts from the Arctic countries (Canada, Denmark/Greenland/Faroe Islands, Finland, Iceland, Norway, Russia, Sweden, and the United States), together with contributions from indigenous peoples organizations, from other organizations, and from experts in other countries.

AMAP would like to express its appreciation to all of these experts, who have contributed their time, effort, and data; and especially to the lead experts who coordinated the production of this report, and to referees who provided valuable comments and helped ensure the quality of the report. A list of the main contributors is included in the acknowledgements on page vi of this report. The list is not comprehensive. Specifically, it does not include the many national institutes, laboratories and organizations, and their staff, which have been involved in the various countries. Apologies, and no lesser thanks, are given to any individuals unintentionally omitted from the list. Special thanks are due to the lead authors responsible for the preparation of the various chapters of this report.

The support of the Arctic countries is vital to the success of AMAP. AMAP work is essentially based on ongoing activities within the Arctic countries, and the countries also provide the necessary support for most of the experts involved in the preparation of the assessments. In particular, AMAP would like to express its appreciation to Norway and Russia for undertaking a lead role in supporting the Radioactivity assessment. Special thanks are also offered to the Nordic Council of Ministers for their financial support to the work of AMAP, and to sponsors of projects that have delivered data for use in this assessment.

The AMAP Working Group that was established to oversee this work, and the AMAP radioactivity expert group are pleased to present its assessment.

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Oslo, August 2010

\* AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xii+859 pp.

\*\* AMAP, 2004. AMAP Assessment 2002: Radioactivity in the Arctic. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xi+100 pp.

\*\*\* AMAP, 2009. Arctic Pollution 2009. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xi+83 pp.

# Acknowledgements

# Executive Summary to the *Arctic Pollution 2009* Ministerial Report

## Preamble

The Arctic Monitoring and Assessment Programme (AMAP) was established in 1991 to monitor identified pollution risks and their impacts on Arctic ecosystems. The first AMAP report, Arctic Pollution Issues: A State of the Arctic Environment Report[1] and its update Arctic Pollution 2002[2] were published in 1997 and 2002, respectively. Three further reports have been published on specific topics: the Arctic Climate Impact Assessment[3] (produced by AMAP in cooperation with the Conservation of Arctic Flora and Fauna working group and the International Arctic Science Committee in 2004), and reports on Acidification and Arctic Haze[4] (2006) and Arctic Oil and Gas[5] (2008).

These assessments show that the Arctic is closely connected to the rest of the world. The Arctic receives contaminants from sources far outside the Arctic region; Arctic climate influences the global climate and vice versa. The AMAP assessment reports have been welcomed by the Arctic governments, who have agreed to increase their efforts to limit and reduce emissions of contaminants into the environment and to promote international cooperation in order to address the serious pollution risks and adverse effects of Arctic climate change reported by AMAP.

AMAP information assisted in the establishment, and continues to assist the further evaluation and development of the protocols on persistent organic pollutants (POPs) and heavy metals to the United Nations Economic Commission for Europe's (UN ECE) Convention on Long-range Transboundary Air Pollution (LRTAP Convention) and the Stockholm Convention on Persistent Organic Pollutants. Information from AMAP is useful in documenting trends and in showing whether persistent substances are accumulating in the Arctic, which is relevant with respect to the screening criteria for persistence, long-range transport, and bioaccumulation that are applied to proposals to add substances to the above international agreements.

The Arctic Council's Arctic Contaminants Action Program (ACAP) was established to undertake cooperative actions to reduce pollution of the Arctic as a direct follow-up to address the concerns raised by AMAP. AMAP information is also used in establishing priorities for the Arctic Council/ PAME Regional Programme of Action for the Protection of the Arctic Marine Environment from Land-based Activities (RPA). A number of activities have been initiated to follow-up on the Arctic Climate Impact Assessment.

The current assessment report updates to the information presented in the AMAP 1997 and 2002 assessment reports with respect to three subject areas: persistent organic pollutants, contaminants and human health, and radioactivity. The POPs update has a particular emphasis on 'emerging' and current use POPs. The human health update addresses health effects of POPs, mercury, and lead exposure.

The information presented in the Arctic Pollution 2009 report is based on scientific information compiled for AMAP by scientists and experts, as listed on page 83. The background documents to this assessment have been subject to peer review and are in the process of being published in AMAP scientific assessment reports or scientific journals. All of these documents are made available on the AMAP website, www.amap.no.

This Executive Summary provides the main conclusions and recommendations of the 2009 AMAP assessments.

## Persistent Organic Pollutants (POPS)

### Legacy POPS

P1. Levels of many POPs have declined in the Arctic environment. This is a consequence of past bans and restrictions on uses and emissions in Arctic and other countries. 'Legacy' POPs that contaminate the Arctic mainly as a result of past use and emissions include PCBs, DDTs, HCB, chlordane, dieldrin, toxaphene, and dioxins.

P2. National policy efforts to reduce the use and emissions of these POPs have been extended regionally and globally through the UN ECE LRTAP POPs Protocol and Stockholm Convention, respectively. These initiatives made extensive use of the information presented in AMAP assessments. The Stockholm Convention on POPs explicitly acknowledges that "... Arctic ecosystems and indigenous communities are particularly at risk." The occurrence of chemicals in the Arctic can be evidence of their ability for long-range transport and environmental persistence.

P3. Firm conclusion about the impact of policy decision on environmental levels will require continued monitoring of 'legacy POPs' in both abiotic environments and in key biota. AMAP information on temporal trends in the Arctic has contributed to the evaluation of the 'effectiveness and sufficiency' of the UN ECE LRTAP Convention Protocol on POPs, and the Stockholm Convention.

P4. Additional years of monitoring are needed to increase statistical power of existing time series in order to verify temporal trends. This will allow examination of the response to efforts to reduce global emissions and how this may be affected by climate variability and possible changes in contaminant pathways.

P5. Despite these reductions, concentrations of some legacy POPs, such as PCBs in some top predators in the marine food web, are still high enough to affect the health of wildlife and humans.

### Emerging and current-use POPS

P6. Many chemicals in commercial use today have the potential to transport to and accumulate in the Arctic but are not yet regulated by international agreements. Although knowledge about these chemicals in the Arctic remains much more limited than for legacy POPs, new monitoring efforts have extended the information concerning their presence in the Arctic. This information is relevant to ongoing consideration of new chemicals for inclusion under existing national, regional and global agreements to regulate use and emissions of POPs.

P7. Many of these compounds transport over long distances and accumulate in Arctic food webs. New knowledge highlights the potential importance of ocean transport pathways. In contrast to atmospheric pathways ocean currents are slow. This may delay the environmental response to regulations.

P8. Compounds that have some POP characteristics and that are documented in the current AMAP assessment include:

- Brominated flame retardants (BFRs)

The current AMAP assessment includes new information on three groups of chemicals used as flame retardants: polybrominated diphenyl ethers (PBDEs) (including Penta-, Octa- and Deca-BDEs), Hexabromocyclododecane (HCBD) and tetrabromobisphenol-A (TBBPA). The assessment shows that:

Penta-BDE transports over long distances and bioaccumulates in biota. Penta-BDE and Octa-BDEs have been banned/restricted in Europe, parts of North America. They are no longer produced in Russia and use there is very limited. Penta-BDE and Octa-BDEs are under consideration for inclusion under the international Conventions regulating POPs; Deca-BDEs are now restricted in the EU.

HBCD is ubiquitous in the Arctic. It undergoes long-range transport and accumulates in animals. It has also been proposed as a candidate for inclusion under international regulations.

There is some evidence that environmental levels of Penta-BDE are now starting to level off or decline due to national regulations and reductions in use and production. TBPPA is present at low levels in several Arctic animals and plants, but more data are needed to assess its potential to undergo long-range transport.

Some BFRs that are used as substitutes for phased-out substances have been detected in occasional Arctic samples. Their presence in the Arctic is a warning sign that they may have some POP characteristics.

- Fluorinated compounds

Fluorinated compounds reach the Arctic both via the atmosphere and via ocean currents. They are extremely persistent and accumulate in animals that are high in the marine food web.

Production of products containing perfluorooctane sulfonate (PFOS) was substantially reduced in 2001, but PFOS continues to be produced in China. Products that contain PFOS and other fluorinated compounds can still serve as sources to the environment. PFOS and related compounds are currently subject to review for both international and national regulation.

Perfluorooctanate (PFOA) and other perfluorocarboxylates (PFCAs) continue to be produced. Fluorinated substances can also degrade to PFOA and other PFCAs. Canada is the only Arctic country so far to ban some import and manufacture of several products that are suspected to break down to PFOA and PFCAs.

Precursors of PFOS and PFCAs have been detected in Arctic air and may be a source of PFOS and PFCAs in Arctic wildlife. Concentrations in Arctic air are one order of magnitude lower than in more southern, urban regions.

Time trends of PFOS in wildlife show an initial increase starting in the mid-1980s. In recent years, some studies show a continuing increase while others show a sharp decline. The declines follow reduction in PFOS production. PFCAs have increased in Arctic wildlife since the 1990s, reflecting continued production of their precursors.

- Polychlorinated naphthalenes

Polychlorinated naphthalenes (PCNs) are no longer manufactured and levels in the environment peaked almost half a century ago. However, PCNs are still present in the Arctic with indications of further input from a combination of combustion sources and emission from old products. There are no studies to assess their temporal trends in the Arctic. They contribute to dioxin-like toxicity in Arctic animals but are generally much less important than PCBs.

- Endosulfan

Endosulfan is a pesticide that is still in use in many parts of the world. Endosulfan and its breakdown products appear to be persistent in the environment. The presence of endosulfan in the Arctic confirms its ability to transport over long distances. There is clear indication of bioaccumulation in fish but there is no evidence for biomagnification by marine mammals.

Long-term trend analysis of samples taken at Alert (Ellesmere Island, Canada) indicates that endosulfan concentrations have remained unchanged in the remote Arctic atmosphere, unlike most legacy POPs. Calculations based on air and seawater concentrations suggest that endosulfan enters open (i.e. ice-free) waters of the Arctic Ocean.

The limited information available in wildlife indicates that concentrations of endosulfan and its breakdown product endosulfan sulphate in blubber of marine mammals are an order of magnitude lower than those of major legacy POPs such as DDT and chlordane.

Endosulfan is currently under discussion for inclusion under the UN-ECE LRTAP POPs Protocol and the Stockholm Convention.

- Other current-use pesticides

Previous AMAP assessments have highlighted lindane (gamma-hexachlorocyclohexane [HCH]) as a current-use pesticide that is ubiquitously present in the Arctic. Several other current use pesticides (including chlorpyrifos, chlorothalonil, dacthal, diazinon, diclofol, methoxychlor, and trifluralin) have been detected in the Arctic. The levels are often low, but their presence shows that they can transport over long distances and accumulate in the food web.

## Biological effects

P9. Recent studies of biological effects of POPs have been able to confirm the causal link between POPs and observations of adverse health effects in Arctic top predators. These controlled experiments on sled-dogs and captive Arctic foxes show effects on hormone, immune and reproductive systems.

P10. The observed effects are mainly due to the breakdown products, indicating that these may be more important than the original POP compounds.

# Contaminants and Human Health

## Population health and effects of contaminants

H1. In light of current studies, many indigenous populations in the Arctic region have poorer health than national averages. While socioeconomic conditions and lifestyle choices are major determinants of health, contaminants may also have a contributing effect. Toxicological studies show that contaminants, at the levels found in some parts of the Arctic, have the potential for adverse health effects in people. Epidemiological studies, looking at Arctic residents directly, provide evidence for subtle immunological, cardiovascular, and reproductive effects due to contaminants in some Arctic populations. These results indicate that POPs, mercury, and lead can affect health of people and especially children at lower levels of exposure than previously thought. Genetic characteristics of the various Arctic populations also affect their response to contaminants and susceptibility to certain diseases.

H2. A major dietary shift from traditional to store-bought food is underway in most of the Arctic, with important health implications. In addition to environmental concentrations of the contaminants in traditional foods, lifestyle factors and social and cultural practices play a large role in determining human exposure to contaminants in Arctic areas. Despite changes in lifestyle and diet that are resulting in increasing consumption of store-bought foods, traditional foods remain important to Arctic indigenous peoples for social, cultural, nutritional, economic, and spiritual reasons. Store-bought foods are increasingly the main source of dietary energy, but traditional foods provide many nutrients and are still a major contributor to healthy diets in many communities. Some traditional foods can also carry potential risks from contaminants. The combination of high prices for store-bought foods and the work, risks, and costs associated with obtaining traditional foods has made food security a large concern for many Arctic residents.

H3. Recent studies have found a number of mechanisms by which contaminants can affect metabolism. Obesity is associated with an increased risk of cardiovascular disease and of developing diabetes; as in other parts of the world, obesity is increasing in Arctic communities. POPs, even at low concentrations, also increase the risk of diabetes. These new findings emphasize the need to consider the interactions between contaminants and other health conditions.

## Trends in exposure and contaminant levels

H4. Human exposure to most legacy POPs and mercury is decreasing in many Arctic populations. This reflects changes in diet, changing levels of environmental contamination, and health advice to critical groups in some areas concerning consumption of certain foods; however, exposure remains high in some populations. The proportion of women of childbearing age who exceed blood level guidelines for PCBs, mercury, and lead is decreasing. For PCBs and lead, in particular, there is evidence that this reflects the declines in environmental levels of these contaminants.

H5. Marine mammals remain a major dietary source of POPs and mercury, so that people who eat large quantities of marine mammals have higher POPs and mercury levels than those who do not.

H6. Emerging compounds such as brominated flame retardants and fluorinated compounds are a concern for three reasons: they are present in Arctic people and biota, levels globally have increased over the last 15 years, and their toxic effects have not been studied in detail. There is little information on the routes of exposure or trends of these contaminants in Arctic populations.

H7. Reliable interpretation of information on trends and inter-regional differences is critically dependent on an ability to compare data from different studies and different laboratories. Laboratory performance testing procedures initiated by AMAP and others, including the AMAP inter-laboratory comparison programme for analysis of contaminants in human tissue have markedly improved analytical co-operation, data comparability, data reliability and data accuracy in studies using the participating laboratories, and have led to more reliable data on contaminant levels in human tissues. Further improvements can be achieved through continued efforts in this respect.

H8. Increased industrial activity in parts of the Arctic is likely to lead to an increase in local sources of contaminants. Anticipated changes in global and Arctic climate may also result in changes in contaminant transport to the Arctic. Such changes may affect exposure patterns to some contaminants.

## Communication

H9. Communicating the results of studies concerning contaminants and people is important in helping Arctic residents make informed food choices. Health advisories issued in response to findings reported in past AMAP assessments have succeeded in reducing exposure to contaminants in some Arctic population groups.

H10. Risk communication must be carried out with great care and respect for culture at a community-level. The involvement of community members and organizations, regional health officials, and indigenous organizations is the key to developing and disseminating messages that are appropriate and relevant.

# Radioactivity

R1. Radioactivity in the Arctic is a concern because contamination can persist for long periods in soils and some plants and because pathways in the terrestrial environment can lead to high exposures of people.

## Potential sources

R2. In parts of the Arctic, there is a very high density of sources of radionuclides. The risk of accidents combined with the vulnerability of the Arctic environment to radioactive contamination raises a need for continued actions to reduce risks.

R3. Partly as a result of national and international actions addressing concerns highlighted by AMAP, significant progress has been made with respect to actions to reduce

risks of radioactive contamination from several of these potential sources. Previous AMAP assessments recommended actions to address potential sources of radioactive contamination of the Arctic including nuclear powered vessels that were poorly maintained or being decommissioned; dumped and stored radioactive wastes, including wastes stored under inadequate conditions; radioisotope thermoelectric generators (RTGS) used as energy sources in northern regions; and nuclear power plants and reprocessing facilities located close to the Arctic. Many of these potential sources are located in northwest Russia. Other issues remain a source of concern:

- As of 2008, 164 of the 198 obsolete nuclear submarines of the Russian northern fleet had been defueled and dismantled; work to safely decommission these vessels continues. Similar plans exist for dealing with nuclear icebreakers and their associated facilities, including the Lepse storage vessel.
- The facilities at Andreeva Bay and Gremikha are used as temporary storage sites for radioactive wastes, spent fuel, and reactors from decommissioned submarines. Progress has been made in improving the physical infrastructure and the legal arrangements to manage these sites. However, much remains to be done, including transport of spent fuel and waste to safer storage sites.
- About half of the radioisotope thermoelectric generators (RTGS) in northern Russia have been removed or will be in the near future.

R4. Some risk reduction has been achieved through significant joint Russian-international action. This includes a regulatory framework for handling the clean-up actions. Moreover, a long-term strategic master plan has been developed, which could become an important tool for further management of radiation risks.

### New potential sources

R5. Russian plans for building floating nuclear power plants raise issues about how waste will be handled and about increased marine transport of spent fuel in the Arctic. These power plants would represent new potential sources and may increase risks of radioactive contamination.

R6. Technologically enhanced naturally occurring radioactive material (TENORM) can become a radiation risk in context of mining of uranium and other minerals, phosphate production, oil- and gas extraction, coal mining and the use of geothermal energy. Several of these activities are likely to increase in the Arctic and more knowledge about waste streams and releases are needed in order to assess human and environmental risks.
Historical contamination

R7. Previous AMAP assessments documented fallout from past nuclear weapons tests, the 1986 Chernobyl accident, and releases from reprocessing plants close to the Arctic as the three major sources of anthropogenic radioactive contamination in the Arctic. Evidence from long-term monitoring in the European Arctic shows that levels of radioactivity in the environment are declining. However, monitoring and mapping activities have decreased in recent years and documen-

tation is therefore lacking for much of the Arctic. Unless environmental pools are re-mobilized, this historical contamination will continue to decrease as sediments are buried and radionuclides decay.

R8. Application of new technology has reduced routine releases of radionuclides to the marine environment from European reprocessing plants, including releases of technetium-99 from Sellafield that were highlighted in the 2002 AMAP assessment.

### Climate change and radioactivity

R9. The current assessment identifies the potential of climate change to mobilize radionuclides in the Arctic terrestrial environment and in glaciers. This may also affect radon emission from the ground, which is a major contributor to human exposure to radiation.

R10. Changes in permafrost, erosion, precipitation and extreme weather events may also affect infrastructure related to nuclear activities.

### Protecting the environment

R11. Following recommendations of previous AMAP assessments, a framework for protecting Arctic ecosystems from radiation effects has been developed as a complement to the previous focus on protecting human health. It also opens for assessing combined effects with other environmental stressors. There is a need for more data that are relevant for Arctic conditions and organisms to provide the basis for a comprehensive application of this framework.

## Recommendations for actions to reduce contaminant levels and effects through international agreements:

- *Encourage countries that have not yet done so to sign and ratify the Stockholm Convention and LRTAP POPS Protocol (P2, H2, H4, H5).*
- *Support the addition of polybrominated compounds and fluorinated compounds to the Stockholm Convention and the regulation of these compounds under other international and national mechanisms because they undergo long-range transport and bioaccumulation in human tissues similar to other POPS. (P2, P3, H6)*
- *Support the development of a global agreement to limit mercury emissions to complement regional and national efforts that reduce environmental levels and lower human exposure to mercury in the Arctic. (H1)*

## Recommendations for actions to promote healthy diets and reduce human exposure to contaminants:

- *Continue to encourage public health officials to recommend breast feeding among Arctic populations as a health practice that optimizes infant growth and development. (H2, H9)*
- *Recommend to health authorities to promote healthy diets through improved access to and consumption of local tradi-*

tional foods that are high in nutrients but relatively low in contaminants along with improved availability and consumption of store-bought foods with high nutritional value. (H2)

- Evaluate past communication efforts in order to improve and refine communication strategies. (H9)

## Recommendations to address potential sources of radioactivity:

- Continue work to decommission remaining obsolete nuclear vessels, remove remaining RTGs, and to manage spent nuclear fuel and waste at sites in or close to the Arctic. (R3)
- Implement additional actions to address continued concerns, especially the storage facilities at Andreeva Bay and Gremikha, and the Lepse storage vessel (R3)
- Strengthen plans to ensure safe and secure transport of spent fuel and waste to storage facilities. (R3)
- Consider the need to further develop regulatory systems, especially for addressing clean-up operations and improved safety of nuclear facilities. (R4)
- Increase attention to technologically enhanced naturally occurring radioactive materials (TENORM) in future assessments, including information from all countries engaged in or planning Arctic oil and gas extraction and uranium and other mining. (R6)

## Recommendations for actions to address gaps in knowledge concerning combined effects:

### Monitoring

- Continue and enhance the geographical coverage of monitoring programs to:
- Document the effectiveness of controls on the use and emissions of POPs (P2, P3, P4)
- Investigate the possible effects of climate change on Arctic contaminants levels, including changes in transport and remobilization (P4, H8, R9, R10)
- Detect health threats related to climate change and contaminants (H8)
- Identify new sources of contaminants and new contaminants that may pose a threat to Arctic residents and the environment (P6, H6, R7)

### Research

- Investigate the respective and combined roles of changing contaminant emissions, changing pathways due to climate change, local sources of contamination, and dietary change to determine the causes of changing environmental levels and human exposures. (P4, H8, R9, R10)
- Improve predictive models of contaminant transport and behaviour in the Arctic to better understand the likely impacts of climate change with respect to contaminant levels and human exposures. (P4, H8)
- Conduct further studies to better understand the combined effects of contaminants and other stressors on Arctic wildlife and humans. (P5, P9, H8, R11)

- Include in future assessments the combined effects of POPs, radioactivity, and other stressors on human health and the environment in the Arctic (P5, P9, H8, R11)

## Recommendations to address gaps in knowledge concerning POPS:

### Monitoring

- Continue monitoring of occurrence and trends of brominated flame retardants (including alternatives being introduced to replace phased-out BFRS) and fluorinated compounds. (P8)
- Increase monitoring of current- use pesticides and their breakdown products in the Arctic environment. (P7)

### Research

- Examine the many other chemicals in commerce, such as the cyclic siloxanes for potential Arctic accumulation potential and design programs to search for these chemicals and their breakdown products (to avoid past surprises such as detection of PFOS). (P8)

## Recommendations to address gaps in knowledge concerning human health:

### Monitoring

- Continue and extend the laboratory intercomparison and testing schemes introduced and promoted by AMAP for laboratories engaged in analysis of Arctic human media to cover emerging POPs. The quality assurance group for the human health program should be provided with adequate resources to ensure quality assurance/quality control on an ongoing basis. Only data that have been approved by this group should be used in AMAP human health assessments. (H7)
- Continue to monitor for trends in legacy POPs, mercury, and lead in human tissues and traditional food items. Dietary assessments should combine contaminant and nutrient analyses in traditional foods as consumed. (H2, H4)
- Conduct further studies combining dietary assessments with contaminant and nutrient analyses in the traditional foods as consumed. (H2)
- Continue and expand monitoring for emerging POPs in human tissues and traditional food items, including development of analytical methods (H6)(H7)
- Continue gathering basic health statistics on a regular basis by all circumpolar jurisdictions at appropriate regional levels, including ones not currently gathered in all areas (e.g., neonatal vs. post-neonatal death rates in Russia).(H2)

### Research

- Maintain and expand current human population cohorts in the Arctic in order to provide the information needed to track adverse health outcomes associated with contaminants and changing conditions related to climate change, socio-cultural conditions, and diet. (H1, H2, H3)

- *Conduct further research on contaminant effects in humans, including interaction between POPs and mercury and other factors such as genetic susceptibility, diet, and lifestyle, and the resulting health impacts on the cardiovascular, reproductive, neurological or metabolic systems. (H1)(H2)(H3)*
- *Conduct further studies to determine causes of regional variations and discrepancies in exposure to contaminants (e.g., low mercury levels in Chukotka in contrast with high POPs levels). (H2)(H8)*
- *Conduct further toxicological studies of POPs mixtures, and emerging compounds where a lack of information is limiting human health risk assessment. (H1)*
- *Conduct further studies on risk perception, dietary patterns, and determinants of food choice to improve risk communication. (H9)*

## Recommendations to address gaps in knowledge concerning radioactivity:

### Monitoring

- *Improve coverage and implementation of monitoring of radioactivity in the Arctic to meet AMAP objectives and/or to highlight specific regional needs. (R7)*
- *Improve collection and reporting of data relevant to Arctic species and conditions to allow improved radiation protection of Arctic ecosystems. (R11)*

*Chapter 1*

# Introduction

This assessment contains information relating to the levels and distribution of radioactive contamination and radiological consequences of radioactivity in the Arctic. In addition, consideration is given to the consequences of accidents and other possible future sources of contamination in the Arctic. New data concerning actual and potential sources of radioactive contamination in the Arctic are also presented. This is the third AMAP assessment of radioactivity in the Arctic and, in line with the approach taken for the second assessment, 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.

The previous AMAP assessment (AMAP, 2004a) was based on information available up to the beginning of 2002 and constituted a comprehensive follow up to the first AMAP assessment (AMAP, 1997). In the first AMAP assessment, the focus of radiological protection was limited to the consideration of human health. With regard to new material in the second assessment, particular attention was given to the development of a radiological protection system for the environment that presents the methodology to determine doserates and assess potential effects for Arctic biota. By the time the second AMAP assessment was being drafted, a consensus had emerged for the rapid development of an internationally recognized system and framework for the protection of the environment. The International Union of Radioecology, with support from AMAP, was among the first international organizations to promote this. Updated information was also reported on the increase in seawater concentration of the long-lived water-soluble fission products technetium-99 ($^{99}$Tc) and iodine-129 ($^{129}$I), originating from nuclear fuel reprocessing in Western Europe. Other new topics in the previous assessment concerned the foundering and sinking of the nuclear powered submarine *Kursk* off Murmansk in August 2000, and the potential for increased transport by sea of spent nuclear fuel.

The general recommendations from the second AMAP assessment were that:

- Detailed studies of the remobilization of radionuclides from sediment and its potential long-term effects on the Arctic should be conducted.
- More openness for restricted information should be promoted.
- An active part in continued efforts to address environmental protection, taking special responsibility for the Arctic should be taken.
- The vulnerability and impact of radioactivity on the Arctic environment and the consequences for emergency preparedness planning should be clarified.
- Risk and impact assessment programs, including uncertainty estimates, should be performed before action is taken to reduce risk.
- Risk and impact assessments, including accident scenarios, should be undertaken for the transport of radioactive waste and spent nuclear fuel within the Arctic and nearby areas, and with regard to storage and reprocessing within the Arctic and nearby areas.
- Cooperation with Russia to improve the safety and safeguards of nuclear installations and waste sites should be continued.

Chapter 2 updates information available on sources of radioactivity to the Arctic. The information is divided according to sources of past contamination and potential future sources. Chapter 3 (TENORM - technologically enhanced naturally-occurring radioactive materials) addresses concerns associated with the natural radioactivity present in a number of industrial activities. The oil and gas industry is one major source of TENORM, and will be of great concern for the Arctic environment if this industry expands its activities into new areas of the Arctic. Monitoring of different radionuclides is discussed in Chapter 4. Radioisotopes of interest include $^{99}$Tc and cesium-137 ($^{137}$Cs). Measurements are made in various environmental compartments, such as seawater and fish. Measurements of reindeer/caribou from different countries are included and an update of radioactivity in soil samples is presented. Chapter 5 addresses the protection of the Arctic Environment. Arctic climate change is a hot topic, the tempo of which is accelerating faster than earlier thought. The predicted changes will have long-lasting, extensive and fundamental impacts on Arctic ecosystems, their biotic and abiotic constituents, and the populations and societal structures that are fundamentally linked to them. How climate change impacts the radioecology in the Arctic is discussed in Chapter 6. Chapter 7 gives an update of the threats and risk to the Arctic and describes the mitigating actions taken since the last assessment. It also looks at how the recommendations from that last assessment have been taken forward.

# Sources of Artificial Radionuclides

The two previous AMAP assessments of radioactivity in the Arctic have categorized knowledge about existing and potential anthropogenic sources of radionuclides and radiological effects relevant to the Arctic environment. Some sources give rise to planned radioactive discharges, such as nuclear fuel reprocessing plants, nuclear power plants in the vicinity of the Arctic, and fallout of radionuclides from atmospheric nuclear weapons testing. Others are potential sources; these are contained sources of radionuclides that are managed in a manner that has the goal of preventing substantial releases to the environment. These include civilian and military nuclear reactors, nuclear waste storage, and authorized storage of radionuclides in controlled areas such as the Mayak storage ponds. Such containments can fail, leading to additional releases of radionuclides to the open environment (i.e., to areas outside those of normal regulatory control) and associated threats of increased exposures of humans and other organisms.

This chapter covers newly identified sources of radionuclides as well as sources covered by the previous assessment for which new information has become available. Previous assessments may have been incomplete and in other cases the circumstances of the source have changed due to, for example, active intervention to reduce the size of the source, to reduce planned releases, and to reduce the chance of unplanned releases. It should be noted that not all military sources of artificial radionuclides in the Arctic region are covered in this assessment. Some facilities are in transition from military to civilian supervision.

## 2.1. Existing sources

### 2.1.1. Northwest Russia

For major remediation projects which have implications for national strategies, the role of strategic planning is fundamental, as illustrated in the completion of the Phase 2 Strategic Master Plan (SMP) to integrate all Rosatom programs and plans with those of other Russian agencies involved in decommissioning activities, including bilateral and multilateral international agreements. The development of the SMP was initiated by the Russian Ministry for Atomic Energy (Minatom) jointly with the Northern Dimension Environmental Partnership (NDEP) Support Fund in 2003. The SMP development is a step towards the implementation of the Global Partnership Programme approved by the G8 Leaders in Canada in 2002. The first phase of the SMP was completed in 2004, approved by the Assembly of the donor countries, and put into action by a special order of the Minister for Atomic Energy of the Russian Federation. Phase 2 is a key document that exists alongside other programs dealing with the problems of decommissioning the Russian Nuclear Fleet. The SMP covers the majority of facilities in Northwest Russia related to retired military and civil nuclear naval fleet together with their supporting infrastructure. Unlike the majority

of previously accepted programs, the time interval for planning under the SMP is not limited by a short-term framework (five to ten years), but is determined by the period needed to achieve strategic goals for decommissioning and remediation of all facilities.

The SMP is endorsed by Rosatom and was developed for the implementation of projects mainly under its jurisdiction. At the same time, the SMP is a document of international status, supported by the NDEP Nuclear Operating Committee and subject to approval by the Assembly of donor countries.

The Centre for Nuclear and Radiation Safety of Rosatom was established by Rosatom in 2007 to manage spent nuclear fuel (including the import of spent nuclear fuel from abroad) and radioactive waste, and also to decommission nuclear facilities. In March 2008 it was charged with implementation of international nuclear legacy projects in the Northwest Region of Russia – the role previously played by Sev RAO (the Federal State Unitary Enterprise). This change was made in line with the new Rosatom policy of separating customers from operators.

Addressing the legacy site issues in Northwest Russia creates a need for improved and additional facilities for storage of radioactive waste. A new facility for long-term interim storage of reactor compartments (LTSF RC) has been developed at Saida Bay. The first phase for storing 120 reactor compartments, including relevant infrastructure, has been handed over to the operating company, Sev RAO. The repair workshop for cleaning and painting reactor compartments is under construction. The planned date for completion of the workshop is 30 April 2009. Currently, there are 20 reactor compartments being stored at the storage area of the LTSF. The next transport of 7 reactor compartments intended for storage is planned for August 2008. Sixty percent of the second phase of construction of the new facility, for storing 58 reactor compartments and other nuclear objects, has been completed. Handover of this part to the operating company is planned for the third quarter of 2009. The design of the third phase of construction for establishing of the Regional Centre for Conditioning and Long-Term Storage of Radioactive Waste in the North-West Region of the Russian Federation has started.

As progress is made towards achieving the planned objectives, the overall SMP will serve as a reference point for ongoing operational planning and will: 1) incorporate additional technical information as it becomes available; 2) identify the necessary legal and regulatory framework; 3) identify accessible financial resources; and 4) provide the benchmark for new key decision-making.

Environmental monitoring clearly provides important input for the iterative process indicated above, and should thus be planned so as to take account of the time frame of the decommissioning projects; now recognized to be at least ten years.

The Action Plan for Nuclear Safety is the Norwegian authorities' main instrument for cooperation on nuclear safety and prevention of radioactive contamination from nuclear

activities in Northwest Russia. The overarching goal of the action plan is to protect health, the environment, and business activity against radioactive contamination in Northwest Russia. The action plan involves technical support projects and regulatory cooperation. The goals of these technical support projects are as follows:

- industrial support to remove radioisotope thermoelectric generators (RTGs) and decommission submarines;
- development of updated regulatory norms and standards, and corresponding regulatory guidance, relevant to the situation of nuclear legacy management, covering
  - worker protection in especially hazardous operations for spent fuel and radioactive waste recovery;
  - standards for site monitoring and public protection during site remediation activities; and
  - criteria for clearance of sites and for management of radioactive waste;
- development and practice of improved inspection procedures, to check that work is proceeding in accordance with requirements;
- improved emergency preparedness and response; and hence
- greater confidence that projects are being planned and implemented in accordance with safety and human and environmental health protection objectives; and
- an enhanced safety culture.

The Action Plan for Nuclear Safety has recently been updated (http://handlingsplan.nrpa.no/English/left/actionplan/actionplan.htm). This confirms that Norway's future focus will be on the handling and storage of radioactive waste and spent nuclear fuel in Northwest Russia. Priority areas are the decommissioning and dismantling of submarines, rehabilitation of facilities in Andreyev Bay, and removal of radioactive sources from RTGs.

### 2.1.1.1. Radioisotope thermoelectric generators

Radioisotope thermoelectric generators have been used as local sources of electricity in the Arctic as well as other remote sites that are not coupled to an electricity net. RTGs have a radioisotope heat source which generates thermal energy caused by the radioactive decay. The thermal energy is converted to electricity using thermoelectric converters (see AMAP 2004 for further details about RTGs).

More than 1000 RTGs were manufactured in the former Soviet Union, mainly for the purposes of sea navigation and meteorology. Following the collapse of the Soviet Union, some are now in neighboring countries. As of 1 February 2008, 519 RTGs were in operation in Russia; 223 belonging to the Ministry of Defence, 293 belonging to Rosmorrechflot, and 3 belonging to Roshydromet. Figure 2·1 shows the geographical distribution of overall activity of the operating RTGs and compares this with the activity at storage facilities in Andreev's Bay containing spent nuclear fuel and the activity of a nuclear powered submarine containing spent fuel.

The radioisotope heat sources in the RTGs are usually strontium-90 ($^{90}$Sr) and have an activity of 0.7 PBq to 15 PBq, depending on the RTG type. Hence, according to the International Atomic Energy Agency (IAEA), they should be classi-



Activity concentration, PBq
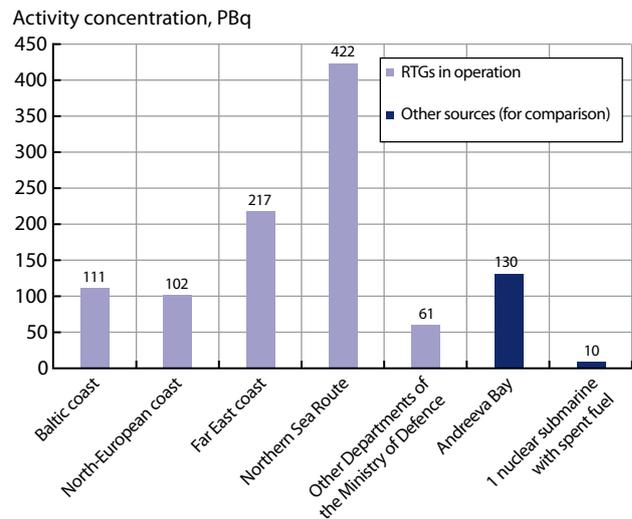
**Figure 2·1.** Overall activity of RTGs operating at coastal sites in different regions of the Russian Federation by the start of 2008 in comparison to activity at storage facilities in Andreev's Bay with spent nuclear fuel and the activity of a nuclear powered submarine with spent fuel (total activity: 916 x 10$^{15}$ Bq = 247,5 x 10$^5$ Ci). Source: Kurchov Institute (2008).

fied as 'Category 1' radioactive sources, i.e., sources that could give exposures at levels that might lead to death after a relatively short period of time (Reka et al., 2006). The half-life of $^{90}$Sr is 29 years and the strontium decay produces the daughter isotope $^{90}$Y, which also is a β-emitter, but with a half-life of 64 hours. The β-emissions from these two isotopes are in themselves a potential radiation threat. In addition, γ- and x-ray radiation caused by the β-radiation exposure of the source cladding, posses a potential radiation hazard. Because of the very high initial activity, radioisotope heat sources remain hazardous for hundreds of years.

Most RTGs have no form of normal security, such as fences or warning signs, and there have been thefts and break-ins. Should RTGs go astray, they present significant radiation hazards. They could also, in the wrong hands, be used in 'dirty-bombs', in which conventional explosives are used to spread radioactive material.

Reported incidents (Standring et al., 2005; Reka et al., 2006) include the following:

- In summer 2001, four people were taken to hospital after being exposed to radiation when trying to disassemble the beacon light near Kandalaksha in the Murmansk area. They were also in pursuit of non-ferrous metal.
- In September 2003, supervision personnel from the Northern Fleet stopped a theft attempt at an RTG beacon light at Golets Island in the White Sea. The beacon light had a particularly strong RTG, with six strontium cores.
- In November 2003, two beacon lights from Kola Bay were found disassembled and everything except the strontium cores had been taken. A third theft was discovered south of the mouth of the Nerpa river.
- Two accidental RTG drops from a helicopter happened in September 2004, when the RTGs dropped onto rocks from an altitude of 100 m. The initial radioactivity was 4.3 PBq for the two RTGs. No release of radioactive $^{90}$Sr from the RTGs was registered.

So far, thieves have only been interested in the metal shielding, leaving the radioactive source behind. They might not even have been aware of their presence. However, these thefts have proven how easy access to highly radioactive sources is, and have demonstrated the need for safe decommissioning of RTGs.

In 2008, Norway will fund the decommissioning of 46 RTGs from the Arkhangelsk and Nenets regions. Thirty belong to the Northern Fleet and the remaining 16 belong to the Ministry of Transport.

In 2007, the last RTG in the Murmansk region was removed. By the end of 2008, 169 out of 180 RTGs in the bilateral deal between Norway and Russia around the Barents Sea will have been removed. The remaining 11 RTGs, located in the Arkhangelsk and Nenets regions, will be removed in 2009. The replacement of the latter RTGs with solar panels will be performed in the same time period.

Decommissioning work funded by Norway has been carried out based on environmental consequence assessments to achieve the following aims: to avoid radioactive contamination of the marine and terrestrial environments; to replace RTGs with less environmentally hazardous solar-energy panels; and to hinder access to radioactive sources.

In spring 2008, the Foreign Ministry of Finland agreed in principle to fund the dismantlement of up to 15 Russian RTGs in the period 2009 to 2011 via the Norwegian program in Russia. The legal arrangements were completed in spring 2009 and the work is planned to begin within 2009.

Two tasks are identified for bilateral French–Russian action: to remove 24 RTGs from the Baltic Sea and to create a new hot cell at Mayak Production Association (PA) for the RTG dismantling. At present, negotiations are conducted on two steps: 1) removing four RTGs from the Baltic Sea, and 2) designing the hot cell at Mayak PA.

Within the US support program, by the end of 2008:

- 24 RTGs will be recovered from Bilibino to the DalRAO temporary storage facility at Vladivostok. The US project team is currently investigating possible sources of funding to move these RTGs for disassembly and disposition in FY2009.
- The United States has recovered 87 RTGs from along the coast of the Northern Sea Route (NSR) and another 10 RTGs were recovered in 2008 using Canadian funding. In 2009, the United States, in cooperation with Canada, will recover all remaining RTGs in the Far East. Once the Far East RTGs have been recovered, the United States will focus its efforts on the remaining NSR RTGs.

As well as the removal of RTGs from the Arctic, the programs include provision for their safe management at new locations. In addition, an enhanced regulatory process has been developed to provide for better safety supervision during decommissioning and transport operations (Sneve and Reka, 2007).

The Contact Expert Group (CEG) for International Radwaste Projects in the Russian Federation was established under the auspices of the IAEA in 1996 to promote international cooperation and assistance in the field of resolving problems caused by radioactive waste and spent nuclear fuel left as a Cold War legacy. To facilitate continuing cooperation in RTG decommissioning, the CEG agreed to set up an

RTG Working Group with its first meeting in autumn 2008, where Russian, Norwegian, French, United States, Canadian, and Finnish experts discussed joint plans and activities.

### 2.1.1.2. Decommissioning of nuclear submarines

A total of 198 Russian nuclear submarines have been taken out of service from the Russian navy. As of March 2008, 164 are decommissioned. This is the result of comprehensive efforts by the Russian Federation and contributions by donor countries, especially G8 countries following the G8 Global Partnership commitment from Kananaskis in 2002. Of the remaining 34 Russian nuclear submarines, 11 are under decommissioning and 20 are waiting to be decommissioned (9 in the Northwest and 11 in the Far East). Three submarines are associated with accidents (CEG, 2008). According to Russian plans, the submarines will be dismantled by 2010.

The decommissioning of nuclear powered submarines has a high priority in the Russian SMP (see section 2.1.1). This is a complex and demanding process, to which Russia and several other countries contribute.

Significant progress has been made in this area since the previous AMAP assessment. Apart from the decommissioning of the submarines themselves, spent fuel has been removed and transferred for safe storage, notably from the liquid-metal cooled reactors in submarines at Gremikha (Figure 2·2). The first operation was carried out in 2005. This required the reactor to be heated and so a powerful boiler house was installed. The personnel engaged in the operation had to undertake training arranged by the Russian Federal Nuclear Agency and Defence Ministry officials (Barents Observer, 2005).

According to the CEG (2008), the main Russian activities planned for 2008 were: dismantling eight nuclear powered submarines (twelve were dismantled in 2007); reprocessing 215 tonnes of liquid radioactive waste (218 tonnes were reprocessed in 2007); treating 1080 $m^3$ of solid radioactive waste to ecologically safe conditions (1184 $m^3$ in 2007); and preparing 42 casks with spent nuclear fuel for transportation from Gremikha.

As part of Canada's Global Partnership program, a third Implementation Arrangement has been agreed with the Zvezdochka shipyard for defueling and dismantling two Yankee Class nuclear powered submarines. Project approval and implementation were to be obtained in July.

The design of a new incinerator at the Zvezdochka shipyard was finalized under the French support program. This also covered the design of the civil construction refitting, which was completed by Zvezdochka and Onega under AREVA TA supervision. Construction of the incinerator began in February 2008, and the post-manufacture tests are scheduled for October 2008. Without such ancillary facilities, the decommissioning work on submarines cannot proceed effectively.

Norway financed the decommissioning of two Viktor II-class nuclear submarines in 2003-04. One in Nerpa on the Kola Peninsula and the other in the Zvezdochka shipyard in Severodvinsk outside Arkhangelsk; both pilot projects. In 2005, a Viktor III-class submarine was decommissioned with Norwegian funding at Nerpa. This involved close cooperation between Norwegian and UK authorities because the

**Figure 2·2.** Location of northern fleet servicing facilities in NW Russia.

possible environmental consequences in the submarine 609 decommissioning project. The project is now well underway and there have been no incidents or accidents connected to this or to earlier decommissioning projects with Norwegian funding.

In February 2008, a contract was signed to dismantle submarine 291 at the Nerpa shipyard. Norway and the UK are sharing the costs and have put together a joint project management team. This will be the fifth Russian submarine dismantled for Norway and the fourth for the UK. In June a contract was signed for converting three three-compartment units of earlier dismantled submarines (also funded by Norway) to one-reactor compartment units at Nerpa for storage at the new facilities at Saida Bay (see section 2.1.1).

The US Department of Defence recently completed a contract with the Sevmash shipyard on the dismantlement of a Typhoon-class strategic nuclear submarine, and is currently funding ongoing work on another Typhoon located at Zvezdochka (elimination of the ballistic missile launcher section and its six-compartment reactor unit). Also participating in this project are Canada and Rosatom. To date, the United States has via the Cooperative Threat Reduction provided assistance to Russia for the dismantling of 31 nuclear powered submarines, including ballistic missile launchers. This assistance includes the procurement and maintenance of dismantling equipment at the Russian shipyards, and the design and construction of facilities, railcars, and spent nuclear fuel casks.

In June 2008, the European Bank for Reconstruction and Development (EBRD) signed implementation agreements with Rosatom's Centre for Nuclear and Radiation Safety for the work in the Northwest to the sum of €70 million. These include:

UK financed the decommissioning of a similar submarine in Nerpa. The Norwegian and UK parties cooperated on contract negotiations and decisions on the necessary documentation and impact assessments. This cooperation reduced costs and increased quality assurance on both sites.

The decommissioning of nuclear powered submarines, including the handling of radioactive waste or spent nuclear fuel, is not a risk-free task. To reduce the probability of accidents and negative effects on health, environment, and safety, Norway emphasizes the importance of undertaking impact assessments of projects financed through the Norwegian Government's Action Plan (e.g., Smith et al., 2004). Such assessments consider not only radiological impacts but also conventional safety and other potential pollution issues from non-radioactive contamination.

After decommissioning of the three submarines was completed, Norway financed the decommissioning of submarine 609, which was a Viktor I-class submarine. The reactors in the three types of submarine are identical and there are several similarities in the general construction. Hence, the decommissioning of submarine 609, and the additional environmental impact assessments have important similarities with the existing documentation on the earlier projects. The Norwegian Radiation Protection Authority (NRPA) has also collected important information, including updated information on fire regulations after a fire at the Zvezdochka shipyard (July, 2007) and additional information on radiation protection and measurement at the Nerpa shipyard.

The reactor section on submarine 609 is to be cut directly into a separate section and stored onshore in the new facilities at Saida Bay (see section 2.1.1). This differs from earlier operations and is seen as an improvement relative to former procedures where larger units were constructed and temporarily stored offshore and floating in Saida Bay. This solution requires the reactor cover units to be towed back to Nerpa for re-building into separate sections which will be stored onshore in the existing storage built through German funding and again, this project requires additional safety justification.

The NRPA considers its earlier experiences with decommissioning projects at the Nerpa shipyard to have been positive, and to have generated sufficient information on

- Defuelling and decommissioning of the Lepse Floating Maintenance Base (spent nuclear fuel from nuclear powered ice-breakers).
- Defuelling of a Papa class nuclear powered submarine. This includes manufacture of defuelling equipment, upgrading of the onshore facility where the reactor cores will be unloaded, and unloading the spent nuclear fuel.
- Further management of the spent nuclear fuel and the resultant radioactive waste. The submarine will be dismantled by Rosatom with the formation of a three-compartment unit so that the actual defuelling will be performed from the unit located in a floating dock.
- Upgrading of the radiation monitoring and emergency response system in the Archangelsk region. This €5.1 million agreement was signed with the Archangelsk region administration.

### 2.1.1.3. Andreeva Bay and Gremikha

In the 1960s, the Russian navy developed Shore Technical Bases at Andreeva Bay and Gremikha on the Kola Peninsula (see Figure 2·2) to service nuclear powered submarines of the Northern Fleet, for example, by providing storage facilities for spent nuclear fuel and radioactive waste. During the relatively short period from the late 1980s to the early 1990s, a large number of nuclear powered submarines were maintained, refuelled, and decommissioned there. After the cessation of support operations, the status of the sites was changed to Sites of Temporary Storage. The maintenance of the Andreeva Bay and Gremikha facilities was not kept up according to initial plans, and the degradation of the storage facilities over the years has given rise to concern.

The spent nuclear fuel at Andreeva Bay was originally stored in building number 5, which was constructed for this purpose. After the spent nuclear fuel was removed from the marine vessels, it was stored for five years before being transported to the Mayak reprocessing plant. In 1982 however, containment failure occurred at building 5, resulting in the leakage of radioactive water and a necessary relocation of the spent nuclear fuel.

Three blocks of dry storage (BDS2A, 2B and 2C) were reconstructed in order to store the spent nuclear fuel moved from building 5. There are about 3000 containers with spent nuclear fuel, holding about $1.3 \times 10^{17}$ Bq (Shandala and Sneve, 2007). The radioactive waste at the site is stored in seven buildings and three open storage sites, containing a total of about 4500 m$^3$ of solid radioactive waste, holding about $6.0 \times 10^{14}$ Bq. Liquid radioactive waste is stored in four tanks located in building number 6. These tanks contain about 1400 m$^3$ to 1600 m$^3$ of liquid radioactive waste, holding about $2.2 \times 10^{12}$ Bq (NRPA, 2003).

The temporary storage facility at Gremikha was mainly used for storing liquid-metal cooled reactors used on the submarines and consists of about 30 buildings. Nineteen have technological functions: spent nuclear fuel, radioactive waste, liquid radioactive waste, containers with control rods, and eight spent extracted parts from nuclear submarines with liquid-metal cooled reactors are stored here. In addition to the buildings and storage systems, there is a dry dock, obsolete service ships, and floating tanks. The facility at Gremikha holds about $1.3 \times 10^{16}$ Bq of spent nuclear fuel and about $3.3 \times 10^{13}$ Bq of radioactive waste (Shandala and Sneve, 2007).

Both sites have areas containing highly radioactive materials and parts of the territory are severely contaminated, with dose rates exceeding 1 mSv/h at some parts (NRPA, 2003). The highest level of radioactive contamination of soil on the site of temporary storage in Andreeva Bay is around the area of the old technological pier and around some spent nuclear fuel storage facilities. Here the $^{137}$Cs activity reaches $5.7 \times 10^7$ Bq/kg, and $5.7 \times 10^6$ Bq/kg for $^{90}$Sr. Within the controlled access area at Gremikha, the gamma dose rate varies from 0.2 µSv/h to 500 µSv/h. The maximum values being four times those of Andreeva Bay. Monitoring of the coastal strip has shown considerable exceedance of typical background values for $^{137}$Cs and $^{90}$Sr activity concentrations in seaweeds, bottom sediments, and vegetation. Preliminary studies on local soils and groundwater indicate that radionuclide migration

from highly contaminated areas on site, via groundwater flow pathways, is possible (Shandala and Sneve, 2007).

### 2.1.1.3.1. Spontaneous chain reaction scenario

A spontaneous chain reaction has been proposed as a possible accident scenario at the Andreeva Bay site. The storage sites for spent nuclear fuel contain large amounts of fissionable material in poor condition and with the presence of salt water. These circumstances make it possible for a spontaneous chain reaction to occur, although the chances of this happening are very low. Even if the storage conditions are much worse than described, the chances of a spontaneous chain reaction are still low. There is however great uncertainty about what type of incidents might occur, the probability of such an incident, and the type of local and regional consequences that could result (Sneve et al., 2007a).

### 2.1.1.3.2. Work in progress

In 2002, the Federal State Unitary Enterprise (Sev RAO) was established to manage the spent nuclear fuel and radioactive waste accumulated in the decommissioning of nuclear powered submarines, and to carry out environmental rehabilitation of hazardous radiation facilities in Northwest Russia.

Much has been achieved in the last few years regarding conditions at the facilities and in securing the spent nuclear fuel at the Andreeva Bay site as part of an OBIN justified program (VNIPIET, 2005). Priorities identified in 2007 (Sneve and Kiselev, 2008) included preparing the right conditions for the removal of spent nuclear fuel and the management of radioactive waste. The international community is helping Sev RAO to overcome these challenges, with support from Norway, the UK, Sweden, Italy and the EBRD for the Andreeva Bay site. France is providing support at the Gremikha site.

Before transportation of the spent nuclear fuel and radioactive waste could start, specially designed equipment and infrastructure needed to be put in place. This included construction of roofs on the tanks to prevent the penetration of rainwater, installations for the removal and decontamination of buildings and material, security of the area, the building of laboratories and the purchase of measuring instruments. The present plan suggests that the transport of spent nuclear fuel and radioactive waste out of Andreeva Bay can begin in 2013 to 2014. There are some uncertainties about the spent nuclear fuel which is not suitable for reprocessing, but it is noted that a proposal for radioactive waste management in Northwest Russia has been developed (CEG, 2008).

Part of the June 2008 EBRD agreement with Rosatom's Centre for Nuclear and Radiation Safety for the works in the Northwest was to provide a system of spent nuclear fuel transportation at Andreeva Bay, including procurement of cranes, construction of accumulation pads for spent nuclear fuel casks, and the development and supply of vehicles for transporting spent nuclear fuel casks to the pier and loading them on to the ship.

A review of the justification document for the treatment and conditioning facilities for solid and liquid radioactive waste at the Andreeva Bay site (VNIPIET, 2005) has resulted in significant modifications of the plans for radioactive waste management facilities (buildings 1 and 203). The current

activities are aimed at designing and integrating all waste management activities and facilities, and the temporary storage facility (building 205). Construction work is expected to begin at the end of 2009 and to be completed in 2013 (CEG, 2008).

Reconstruction of a pier funded by Norway at Andreeva Bay (a key piece of infrastructure for spent nuclear fuel removal) is to be completed by December 2008. Work is currently underway on a detailed design of electricity and water distribution systems at Andreeva Bay (CEG, 2008).

A suite of UK-funded projects is being undertaken at Andreeva Bay. A lightweight cover for the manufacture of boxes will be completed in 2008. A tender for a Principal Contractor was launched in May for the construction of Building 154 (decontamination, maintenance, and storage of support equipment). Of three bids, the Kurchatov Institute was declared the winner, subject to satisfactory contract negotiations (CEG, 2008).

Feasibility studies for the rehabilitation of the Gremikha site are currently on schedule. Urgent work has started, mainly related to refitting the infrastructure and preparing for the VVR reactors spent nuclear fuel removal from Gremikha to Mayak. The removal of the first 294 assemblies is scheduled for the end of 2008 (CEG, 2008). Longer-term remediation factors are also being considered, including the strategy for achieving alternative endstates for the site (Bylkin et al., 2008).

Cooperation has been established between the Federal Medical-Biological Agency (FMBA) of Russia and the Norwegian Radiation Protection Authority to ensure radiation protection of workers, the public, and the environment, as well as to mitigate the risk of accidents in Northwest Russia (Sneve et al., 2008). Cooperation with the civilian nuclear and radiation safety authority, Rostechnadzor, has been ongoing for many years in relation to the regulatory aspects of waste management. Cooperation began in 1998, focusing on Lepse (Sneve et al., 2001), and later on RTGs and other issues (Sneve and Reka, 2007).

Similar cooperation was established in 2007 between the Russian Ministry of Defence, Department for Radiation and Nuclear Supervision and Control, and the NRPA. The combination of these cooperation agreements is intended to support comprehensive and coordinated regulatory activities, taking account of the changing status of Sev RAO-operated facilities.

Under the FMBA cooperation agreement, significant regulatory development has taken place relevant to the Andreeva Bay and Gremikha sites (Sneve et al., 2007a, 2008). This includes development of regulatory norms and guidance on the optimization of worker protection during especially hazardous operations, monitoring requirements on and off the site of the temporary storage facility, criteria for implementing emergency procedures, and the regulatory radiological basis for design and operation of the VLLW facility at Andreeva Bay for the disposal of industrial waste containing very low levels of radioactive contamination, due to be constructed with support from Sweden. Implementation of these regulatory documents and monitoring of compliance forms part of the continuing work program as does a further emergency response exercise at Ostrovnoy (Gremikha), following the successful exercise carried out at Andreeva Bay.

## 2.1.2. Mayak

The Mayak Production Association (Mayak PA) is situated outside the town of Ozyorsk, just to the east of the Ural Mountains at the head of the Techa River (Figure 2·3). The Mayak PA facility began producing weapon-grade plutonium in 1948 and ceased production in 1987 (Malyshev et al., 1997). Of the seven nuclear reactors operating in 1987, two are still operational. The facility also reprocesses spent nuclear fuel, converts weapon-grade plutonium into mixed oxide (MOX) fuel, produces plutonium dioxide ($PuO_2$), uranium dioxide ($UO_2$), and radioisotopes, and manufactures electrical devices and monitoring and control equipment for pipelines (Standring et al., 2008). Very large quantities of radioactive waste have been generated at this site (AMAP, 1998).

In a future scenario, the European Commission (2000) estimated that the inventory of radioactive waste at the Mayak PA facility would be almost 59 000 PBq (Table 2·1) by 2010. This inventory is expected to be dominated by the radioisotopes $^{137}Cs$, $^{90}Sr$, and plutonium-239+240 ($^{239+240}Pu$).
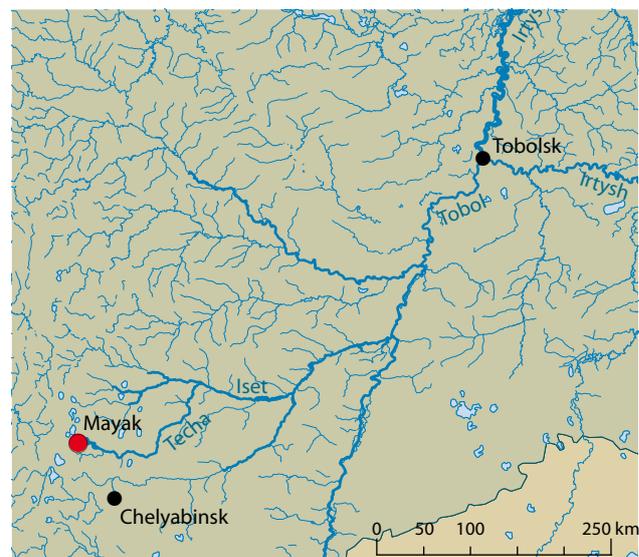




**Figure 2·3.** The Mayak PA is located at the head of the Techa river, part of the Techa-Iset-Tobol-Irtysh-Ob river system that flows into the Kara Sea.

**Table 2·1.** Inventories and categories of radioactive waste associated with the Mayak PA for 2010. Source: EC (2000).

| Radioactive waste/location | Activity, PBq | Volume, m³ |
|---|---|---|
| Liquid MLW | | |
|    Slurry in tanks | $2.81 \times 10^{-1}$ | 330 |
|    Spent extractant in tanks | $2.44 \times 10^{-2}$ | 660 |
|    Lake Karachay | $4.75 \times 10^{3}$ | 930 000 |
|    Staroye Boloto | $7.41 \times 10^{1}$ | 1 291 000 |
| Liquid LLW | | |
|    R2, R3, R4, R6, R10 & R11 | 18.7 | 428 658 000 |
| Solid HLW | | |
|    Vitrified waste, engineered storage[a] | $5.33 \times 10^{4}$ | 4 917 |
|    Engineered repositories (vaults) | $5.52 \times 10^{2}$ | 51 184 |
|    Waste from decommissioning 5 military reactors | $5.03 \times 10^{1}$ | 1 260 |
|    Waste from decommissioning 5 isotope production reactors | $2.41 \times 10^{-1}$ | 26 |
| Solid MLW | | |
|    Engineered repositories (vaults) and trenches | 9.15 | 85 350 |
|    Waste from decommissioning 5 military reactors | $3.64 \times 10^{-1}$ | 4 260 |
|    Waste from decommissioning 5 isotope production reactors | $1.71 \times 10^{-2}$ | 200 |
| Solid LLW | | |
|    Trenches | $5.63 \times 10^{-2}$ | 282 990 |
|    Waste from decommissioning 5 military reactors | $7.08 \times 10^{-4}$ | 5 128 |
|    Waste from decommissioning 5 isotope production reactors | $1.25 \times 10^{-4}$ | 922 |

MLW: medium-level waste; LLW: low-level waste; HLW: vitrified high-level waste; [a] assuming all liquid HLW is vitrified and RT-1 decommissioned in 2010.

**Table 2·2.** Atmospheric emissions and discharges to water from the Mayak PA facility (Bq/y). Source: Hydrometeoizdat (2001, 2002, 2003, 2004); Meteoagency of Roshydromet (2005, 2006); Nizhny Novgorod, Vektor-TiS (2007).

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| Emissions | | | | | | | |
|   $^{137}$Cs | $6.3 \times 10^{8}$ | $9.5 \times 10^{8}$ | $7.4 \times 10^{8}$ | $7.2 \times 10^{8}$ | $5.8 \times 10^{8}$ | $7.41 \times 10^{8}$ | $9.30 \times 10^{8}$ |
|   $^{90}$Sr | $5.9 \times 10^{8}$ | $8.7 \times 10^{8}$ | $5.8 \times 10^{9}$ | $8.0 \times 10^{9}$ | $7.2 \times 10^{8}$ | $7.32 \times 10^{8}$ | $7.23 \times 10^{8}$ |
|   Total α-activity | $2.1 \times 10^{8}$ | $4.0 \times 10^{8}$ | $3.4 \times 10^{8}$ | $4.4 \times 10v^{8}$ | $3.4 \times 10^{8}$ | $6.71 \times 10^{8}$ | $4.60 \times 10^{8}$ |
|   Noble gases | $1.8 \times 10^{13}$ | $1.6 \times 10^{14}$ | $1.2 \times 10^{14}$ | $1.2 \times 10^{14}$ | $7.3 \times 10^{13}$ | $2.62 \times 10^{13}$ | $1.79 \times 10^{13}$ |
| Discharges | | | | | | | |
|   $^{90}$Sr | $1.2 \times 10^{12}$ | $2.2 \times 10^{12}$ | $2.3 \times 10^{12}$ | $1.8 \times 10^{12}$ | $1.9 \times 10^{12}$ | $1.51 \times 10^{12}$ | $1.06 \times 10^{12}$ |
|   $^{3}$H | $8.6 \times 10^{12}$ | – | $6.2 \times 10^{13}$ | $2.7 \times 10^{13}$ | – | – | – |

**Table 2·3.** Activity concentrations of artificial radionuclides in water samples from the Iset, Tobol, Irtysh and Ob rivers collected on 11 to 28 September, 2004.

| Section line | Suspended matter | $^{137}$Cs, Bq/m³ | | $^{90}$Sr, Bq/m³ | $^{239,240}$Pu, mBq/m³ | $^{3}$H, kBq/m³ |
|---|---|---|---|---|---|---|
| | | filtrate | total | filtrate | filtrate | filtrate |
| 1 LB | 0.24 ± 0.09 | 0.40 ± 0.06 | 0.64 ± 0.15 | 15.0 ± 1.8 | – | 2.3 ± 0.2 |
| 1 RB | – | – | – | 6.2 ± 0.9 | – | 2.2 ± 0.1 |
| 2 M | 0.14 ± 0.05 | 0.18 ± 0.05 | 0.32 ± 0.10 | 5.5 ± 0.8 | 13.0 ± 2.1 | 2.6 ± 0.2 |
| 3 M | 0.17 ± 0.02 | 0.20 ± 0.02 | 0.37 ± 0.04 | 26 ± 3 | 6.6 ± 2.0 | 4.4 ± 0.2 |
| 4 LB | < 0.02 | 0.25 ± 0.02 | 0.25 ± 0.02 | 31 ± 4 | – | 3.6 ± 0.2 |
| 5 M | < 0.02 | 0.26 ± 0.02 | 0.26 ± 0.02 | 32 ± 4 | 3.1 ± 1.0 | 4.0 ± 0.2 |
| 6 LB | 0.08 ± 0.01 | 0.22 ± 0.02 | 0.30 ± 0.03 | 57 ± 6 | 5.4 ± 1.2 | 4.0 ± 0.2 |
| 6 M | – | – | – | 18.0 ± 2.2 | – | 2.7 ± 0.1 |
| 6 RB | – | – | – | 12.4 ± 1.5 | – | 2.8 ± 0.2 |
| 7 LB | < 0.1 | 0.18 ± 0.04 | 0.18 ± 0.04 | 69 ± 8 | – | 4.9 ± 0.2 |
| 7 M | – | – | – | 23 ± 3 | – | 3.1 ± 0.2 |
| 7 RB | – | – | – | 8.5 ± 1.1 | – | 3.9 ± 0.2 |
| 8 LB | 0.11 ± 0.04 | 0.15 ± 0.03 | 0.26 ± 0.07 | 72 ± 8 | 7.5 ± 2.0 | 4.3 ± 0.2 |
| 9 LB | < 0.02 | < 0.02 | < 0.04 | 5.9 ± 0.8 | < 1.6 | 2.6 ± 0.2 |
| 10 M | – | – | – | 84 ± 10 | – | – |
| 11 M | – | – | – | 180 ± 22 | – | – |
| 12 LB | 0.13 ± 0.05 | 0.18 ± 0.02 | 0.31 ± 0.07 | 185 ± 22 | – | 6.5 ± 0.3 |
| 13 M | – | – | – | 220 ± 25 | – | – |
| 14 M | – | – | – | 600 ± 55 | – | – |
| 15 LB | < 0.02 | 0.17 ± 0.02 | 0.17 ± 0.02 | 740 ± 60 | 3.3 ± 1.6 | 11.3 ± 0.4 |
| 15 RB | – | – | – | 705 ± 6 | – | 11.3 ± 0.4 |
| 16 LB | 0.09 ± 0.02 | 0.18 ± 0.02 | 0.27 ± 0.03 | 1 030 ± 80 | 4.1 ± 1.6 | 15.2 ± 0.5 |
| 17 RB | < 0.05 | 0.12 ± 0.03 | 0.12 ± 0.03 | 10.3 ± 1.4 | < 2.4 | 2.9 ± 0.2 |

LB: left bank of river; RB: right bank of river; M: middle of river.
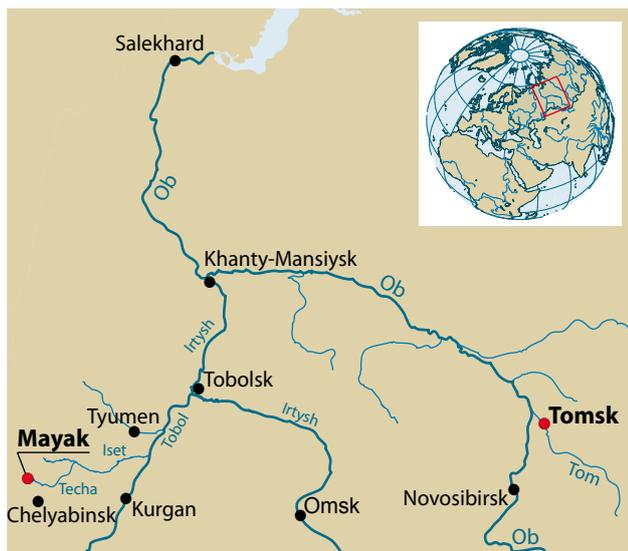
**Figure 2·4.** Schematic location of the Mayak PA and the Siberian Chemical Combine on the Ob-Irtysh river system.





**Figure 2·5.** Trends in $^{90}$Sr activity concentrations in Tobol river water (upstream of the confluence with the Irtysh river – point 1) and in Irtysh river water (upstream of the confluence with the Tobol river – point 2, and on the left bank downstream of the confluence with the Tobol river – point 3). Source: Nikitin et al. (2007).

Data on atmospheric and aquatic discharges from the Mayak PA between 2000 and 2006 are presented in Table 2·2.

Radioactive waste from the Mayak PA facility has been transported through the Techa-Iset-Tobol-Irtysh-Ob river system (Figure 2·4) for many years. Adequate methods for handling the large amounts of radioactive wastes generated during the reprocessing activities were not available when the facility first became operational and so considerable amounts of radioactivity have entered the environment here.

Many studies have been undertaken on the impacts of these radioactive wastes on the Techa-Iset-Tobol-Irtysh-Ob river system, although most studies had until recently been undertaken close to the source on the Techa and Iset rivers (e.g., Malyshev et al., 1997) with just a few studies on the Tobol-Irtysh sections (e.g., Gedeonov, 1971; Trapeznikov et al., 1995). This changed in 2004 when monthly monitoring of radionuclide activity concentrations in the Tobol and Irtysh rivers began. Levels of $^{137}$Cs, $^{90}$Sr and tritium in water are now being measured every month in the area of the Tobol and Irtysh confluence. To establish $^{137}$Cs, $^{90}$Sr, $^{239,240}$Pu and tritium levels in water throughout the system as a whole, a radioecological survey of the Tobol and Irtysh rivers from the mouth of the Iset River to the confluence with the Ob River was undertaken in 2004 (Nikitin et al., 2007).

Monthly monitoring at the Tobol and Irtysh confluence has shown that $^{90}$Sr activity concentrations in water in the Tobol River upstream of its entry into the Irtysh River and on the left bank of the Irtysh River downstream from the confluence with the Tobol River (33 – 235 Bq/m³) are an order of magnitude higher than the background level for rivers in Russia. In the Irtysh River upstream of its confluence with the Tobol River, $^{90}$Sr activity concentrations in water are effectively background (Figure 2·5). Measured concentrations of $^{137}$Cs (< 0.2 – 2.9 Bq/m³) and tritium (2900 – 7900 Bq/m³) are at background levels at all three sites, although tritium levels are slightly higher in the Tobol River and downstream of its confluence with the Irtysh River, than upstream in the Irtysh River.

The results of the radionuclide survey of river water samples within the Techa-Iset-Tobol-Irtysh-Ob river system (Figure 2·6 and Table 2·3) indicate progressive changes downstream of the Mayak PA facility. Figure 2·7 illustrates the change in activity concentration of $^{90}$Sr in filtered river water in the Tobol section of the river system as a function of distance from the sampling point at the mouth of the Iset River. It is clear that $^{90}$Sr activity concentrations increase with proximity to the mouth of the Iset River. The $^{90}$Sr activity concentration in water from the Iset River near the confluence with the Tobol River (1000 Bq/m³) is the highest observed in the study and about 200 times higher than the typical level in Russian rivers (5 – 6 Bq/m³), but is lower than the intervention level for drinking water (5000 Bq/m³; Radiation, 1999).
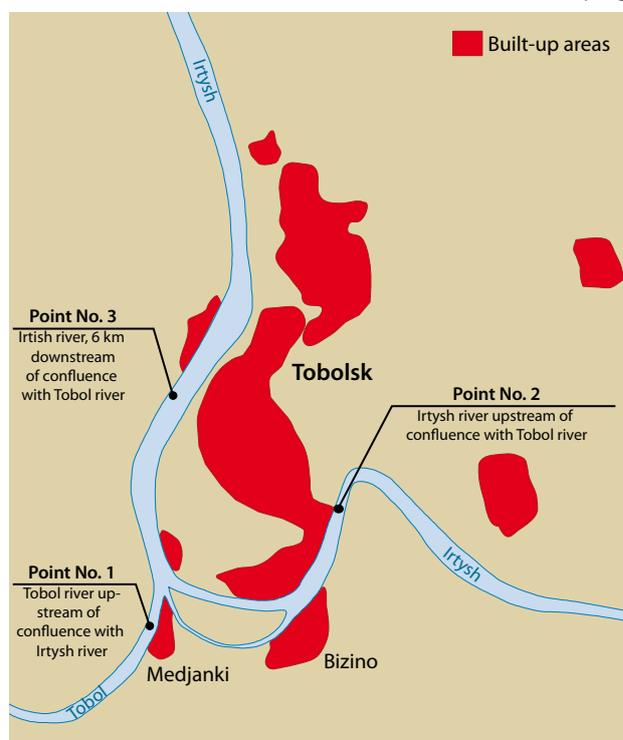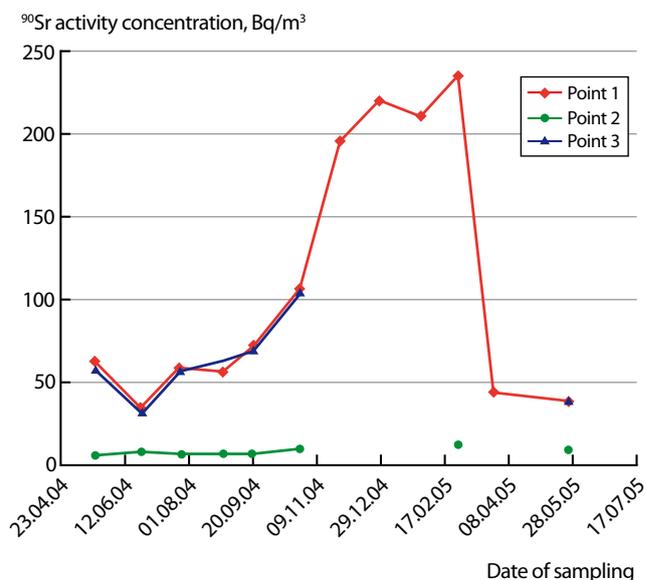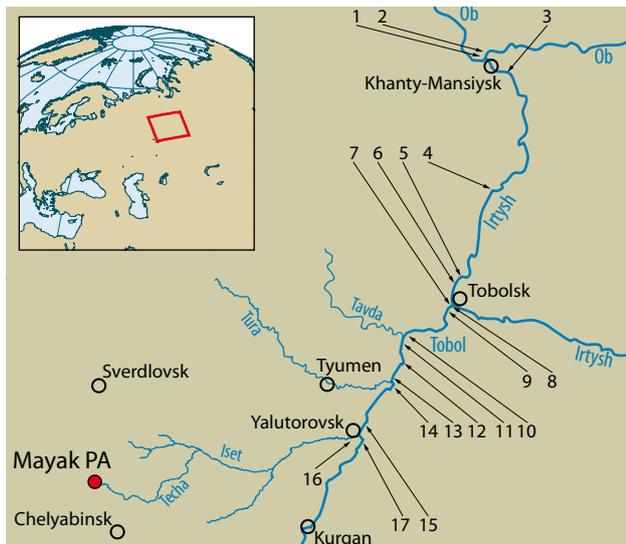
**Figure 2·6.** Location of the section lines for the water sampling in September 2004.



**Figure 2·7.** Changes in $^{90}$Sr activity concentration in filtered Tobol River water along the section between its confluence with the Iset River and its confluence with the Irtysh River, September 2004.

$^{90}$Sr activity concentrations in the Tobol River upstream from the confluence with the Iset River are effectively background at about 10 Bq/m³. The impact of the waste discharges from the Mayak PA facility on the river system can be traced as far as the confluence of the Irtysh and Ob rivers; activity concentrations in the Irtysh river near its confluence with the Ob river are about four to five times higher than background, while activity concentrations in the Ob river upstream from the confluence with the Irtysh river are around background at about 6 Bq/m³. The impact of the Mayak PA facility is also evident in the activity concentrations of tritium, although to a much lesser degree than for $^{90}$Sr.

Another technique for investigating whether radioactive contaminants follow the river pathway is to use an established method involving mass spectrometry to determine the origin of plutonium contamination by calculating the $^{240}$Pu:$^{239}$Pu ratio. Skipperud et al. (2004) found that in Ob Estuary sediments, the main source of plutonium was global fallout from atmospheric nuclear weapons testing, rather than activities performed at the Mayak PA facility. However, Lind et al. (2006) found that in water samples along the lower sections of the Ob River, the estuary, and the adjacent Kara Sea, the main source was most likely to be the Mayak PA facility. Lind and co-workers considered that their finding reflected the fact that global-fallout derived plutonium and the low $^{240}$Pu:$^{239}$Pu ratio sources of plutonium exhibit different environmental behavior, indicating that the plutonium derived from the global-fallout source is particle reactive, whereas the non-civil, non-global fallout sources of plutonium (e.g., from Mayak PA) were more mobile.

### 2.1.3. Sellafield

The only operational nuclear reprocessing facility in the UK is at Sellafield on the west coast. The site covers 262 hectares and comprises a large, complex nuclear chemical facility that has supported the nuclear power program since the 1940s and has undertaken work for a number of organizations, including the UK Atomic Energy Authority and the Ministry
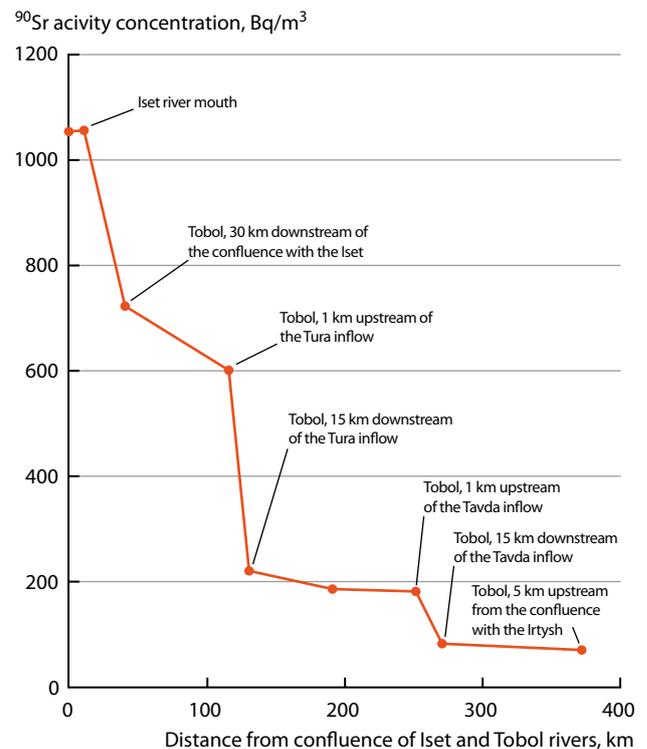
of Defence. Operations at Sellafield include the reprocessing of fuels removed from nuclear power stations, MOX fuel fabrication, and the storage of nuclear materials and radioactive wastes. Most of the waste arising from spent fuel reprocessing is temporarily stored at the facility, but some low-level liquid waste is discharged into the Irish Sea (Figure 2·8), and some gaseous effluents are released to the atmosphere. Owing to the potential for the transport of radionuclides to the Arctic, there is particular interest in planned discharges to sea. A major accident at the site could result in releases to the atmosphere (as well as to the marine environment) which could be transported to and impact upon the Arctic. Some of the radionuclides entering the marine environment from the Sellafield complex are transported over great distances. Discharges of $^{99}$Tc for instance, are transported with the Atlantic currents to the Norwegian Sea and are then transported northward along the Norwegian coastline to the Barents Sea by the Norwegian Coastal Current. The Sellafield-to-Barents Sea transport is estimated to take four to five years (Dahlgaard, 1995; Brown et.al., 2000).

#### 2.1.3.1. Discharges to the Irish Sea

Discharges of $^{99}$Tc from the Sellafield facility have declined considerably since their peak in 1995. In 1994, the Enhanced Actinide Removal Plant (EARP) became operational and began to treat the backlog of stored medium-active liquid wastes (AMAP, 2004a). EARP was designed to rinse the waste for actinides, but not for radionuclides such as $^{99}$Tc. This explains the peak in $^{99}$Tc discharges in 1995 (190 TBq/y) and the high level of discharges in subsequent years. In 2004, the $^{99}$Tc discharge dropped to 14 TBq/y. This was due to the
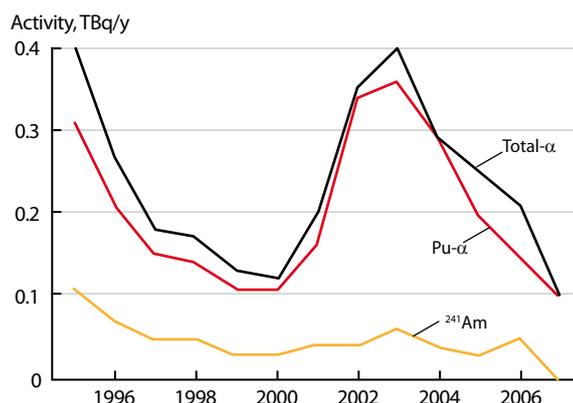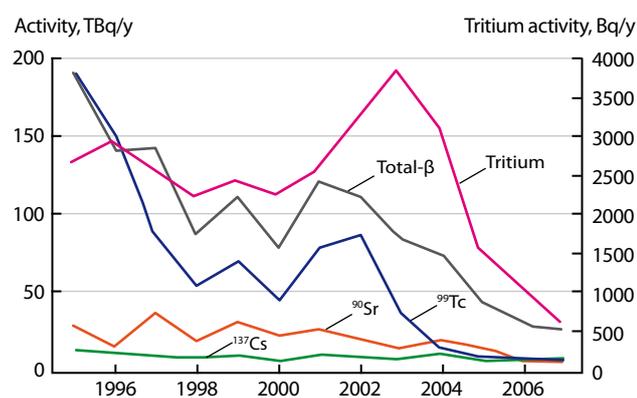
**Figure 2·8.** Trends in radionuclide discharges from the Sellafield nuclear reprocessing facility to the Irish Sea.

**Table 2·4.** Maximum permitted volumes of Highly Active Liquor to be stored at the Sellafield nuclear reprocessing facility. Source: U.K. Nuclear Installations Inspectorate (http://www.hse.gov.uk/nuclear/halstocklc32.pdf).

| Date | Volume, $m^3$ |
|---|---|
| 1 April 2007 | 1225 |
| 1 April 2008 | 1190 |
| 1 April 2009 | 1155 |
| 1 April 2010 | 1125 |
| 1 April 2011 | 1090 |
| 1 April 2012 | 1055 |
| 1 January 2013 | 1020 |
| 1 April 2013 | 960 |
| 1 April 2014 | 625 |
| 1 April 2015 | 290 |
| 1 July 2015 | 200 |
| After 2015 | 200 |
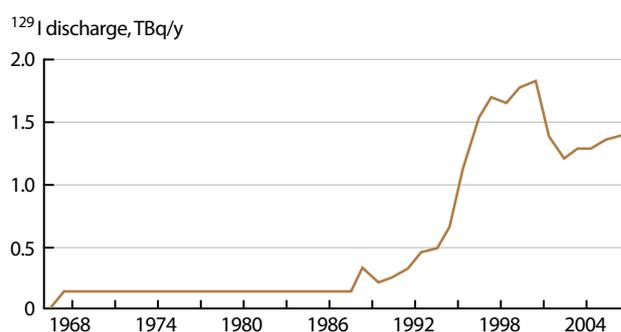


**Figure 2·9.** Trends in the discharge of $^{129}$I from the nuclear reprocessing plant at Cap de la Hague La Hague to the Channel.

implementation of a new rinse technology at the Sellafield facility, designed to extract the $^{99}$Tc.

Tritium discharges have also declined considerably, from 3900 TBq in 2003 to 630 TBq in 2007; a six-fold reduction in four years (Figure 2.8; BNFL, 1999, 2003; Sellafield Ltd, 2006).

### 2.1.3.2. Accident scenarios

In addition to planned discharges, Sellafield is a source of potential radioactive releases from accidents.

During reprocessing, Highly Active Liquor (HAL) is produced and stored in tanks. There are 21 tanks with a total volume of 2360 $m^3$ and a working volume of 1500 $m^3$ (Turvey and Hone, 2000; Brekken et al., 2004). In 2004, there were between 1000 and 1500 $m^3$ of HAL in storage at the Sellafield facility (UK POST, 2004). 95% of the activity in the tanks is due to $^{137}$Cs and $^{90}$Sr and the activities are estimated at $6.7 \times 10^{18}$ Bq and $4.8 \times 10^{18}$ Bq, respectively. The UK Nuclear Installations Inspectorate has specified limits to the volume of liquid high level waste that may be stored at Sellafield and the volumes are required to be reduced over time (Table 2·4).

Brekken et al. (2004) examined the consequences of accidental releases from four of the HAL tanks at Sellafield. Their scenario involved the discharge to the environment of 5% and 3.7% respectively of the inventory of $^{137}$Cs and $^{90}$Sr; corresponding to 105 PBq of $^{137}$Cs and 53 PBq of $^{90}$Sr. Modeling suggested that these accidental discharges would represent a considerable addition to the current inventory of radionuclides in the Barents Sea. An increase in the concentration of $^{137}$Cs in water would be apparent after two to three years.

Concentrations would then increase and peak after eight to ten years. The highest projected concentration of $^{137}$Cs would be well above the highest level from the 1980s (50 Bq/$m^3$), and even after 20 years, concentrations would be well above the current level (Turvey and Hone, 2000; Brekken et al., 2004).

### 2.1.4. La Hague

The nuclear reprocessing plant at Cap de la Hague on the north coast of France is the other major reprocessing plant in Europe. This site (as is the case at Sellafield), has planned discharges via pipelines. Annual discharges of $^{129}$I to the Channel are shown in Figure 2·9. Discharges increased substantially in the early 1990s, from less than 0.5 TBq/y to a peak of 1.83 TBq/y in 1999, but have since fallen and are now between 1 TBq/y and 1.5 TBq/y.

### 2.1.5. Operational releases from nuclear powerplants and other industrial facilities

The main sources of artificial radioactivity discharge in the Russian Arctic are the operational nuclear power plants at Kola and Bilibino. These sites result in atmospheric emissions and wastewater discharges of several radionuclides (Tables 2·5 and 2·6). Quantities of radionuclides emitted to the atmosphere from the Kola nuclear power plant have pro-

**Table 2·5.** Atmospheric emissions and discharges to water from the Kola nuclear power plant (Bq/y).

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| Emissions | | | | | | | |
| $^{60}Co$ | $4.5 \times 10^8$ | $3.4 \times 10^8$ | $1.9 \times 10^8$ | $1.2 \times 10^8$ | $1.9 \times 10^8$ | $8.82 \times 10^7$ | $8.05 \times 10^7$ |
| $^{131}I$ | $3.6 \times 10^9$ | $4.1 \times 10^9$ | $1.7 \times 10^9$ | $1.0 \times 10^9$ | $6.1 \times 10^8$ | $1.34 \times 10^8$ | $1.88 \times 10^7$ |
| $^{137}Cs$ | $1.6 \times 10^8$ | $6.2 \times 10^7$ | $4.2 \times 10^7$ | $9.7 \times 10^7$ | $4.3 \times 10^7$ | $5.33 \times 10^7$ | $8.20 \times 10^6$ |
| Nobel gases | $5.0 \times 10^{13}$ | $4.9 \times 10^{13}$ | $2.0 \times 10^{13}$ | $2.3 \times 10^{13}$ | $7.6 \times 10^{12}$ | $4.19 \times 10^{12}$ | $7.50 \times 10^{11}$ |
| Discharges | | | | | | | |
| $^{51}Cr$ | $5.7 \times 10^6$ | $2.0 \times 10^7$ | $7.9 \times 10^6$ | $5.1 \times 10^6$ | $1.6 \times 10^6$ | $4.1 \times 10^6$ | $8.5 \times 10^6$ |
| $^{54}Mn$ | $2.0 \times 10^6$ | $3.4 \times 10^6$ | $1.6 \times 10^6$ | – | $1.0 \times 10^6$ | – | – |
| $^{57}Co$ | $1.1 \times 10^6$ | $2.9 \times 10^6$ | – | – | – | – | – |
| $^{59}Fe$ | $1.5 \times 10^6$ | $5.1 \times 10^5$ | – | – | – | – | – |
| $^{60}Co$ | $4.0 \times 10^7$ | $5.7 \times 10^7$ | $5.2 \times 10^6$ | $1.0 \times 10^7$ | $2.3 \times 10^7$ | $5.5 \times 10^6$ | $8.6 \times 10^5$ |
| $^{131}I$ | $1.0 \times 10^7$ | $5.6 \times 10^7$ | $3.3 \times 10^6$ | $2.7 \times 10^6$ | $1.4 \times 10^7$ | $3.9 \times 10^5$ | – |
| $^{134}Cs$ | $5.0 \times 10^6$ | $7.0 \times 10^6$ | $8.2 \times 10^6$ | $2.0 \times 10^7$ | $1.0 \times 10^7$ | $1.0 \times 10^5$ | $6.4 \times 10^5$ |
| $^{137}Cs$ | $2.0 \times 10^7$ | $8.8 \times 10^6$ | $1.1 \times 10^7$ | $2.7 \times 10^7$ | $1.9 \times 10^8$ | $4.1 \times 10^6$ | $4.0 \times 10^6$ |

**Table 2·6.** Atmospheric emissions and discharges to water from the Bilibino nuclear power plant (Bq/y).

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| Emissions | | | | | | | |
| $^{60}Co$ | – | – | $5.4 \times 10^6$ | $1.1 \times 10^7$ | $7.9 \times 10^6$ | $7.33 \times 10^6$ | $7.91 \times 10^6$ |
| $^{90}Sr$ | – | – | $2.2 \times 10^4$ | $7.0 \times 10^2$ | $3.0 \times 10^3$ | $1.2 \times 10^4$ | $1.00 \times 10^4$ |
| Noble gases | $4.6 \times 10^{14}$ | $5.2 \times 10^{14}$ | $5.1 \times 10^{14}$ | $5.2 \times 10^{14}$ | $4.2 \times 10^{14}$ | $4.09 \times 10^{14}$ | $3.55 \times 10^{14}$ |
| Discharges | | | | | | | |
| $^{54}Mn$ | – | $1.7 \times 10^6$ | – | $2.5 \times 10^7$ | $3.2 \times 10^8$ | $7.4 \times 10^8$ | $2.4 \times 10^8$ |
| $^{60}Co$ | – | $6.0 \times 10^6$ | $8.0 \times 10^8$ | $9.0 \times 10^8$ | $7.3 \times 10^8$ | $1.2 \times 10^9$ | $7.5 \times 10^8$ |
| $^{90}Sr$ | $7.3 \times 10^7$ | – | $6.1 \times 10^6$ | – | $3.7 \times 10^5$ | $2.0 \times 10^6$ | $6.9 \times 10^5$ |
| Total β-activity | $7.3 \times 10^7$ | – | – | – | – | – | – |

**Table 2·7.** Atmospheric emissions and discharges to water from the Siberian Chemical Combine (Bq/y).

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| Emissions | | | | | | | |
| $^{90}Sr$ | $6.3 \times 10^8$ | $3.7 \times 10^8$ | $4.8 \times 10^8$ | $3.5 \times 10^8$ | $2.1 \times 10^8$ | $2.90 \times 10^8$ | $2.00 \times 10^8$ |
| $^{131}I$ | $5.3 \times 10^9$ | $4.5 \times 10^9$ | $3.9 \times 10^9$ | $2.9 \times 10^9$ | $2.4 \times 10^9$ | $2.49 \times 10^9$ | $1.77 \times 10^9$ |
| Total β-activity | $6.4 \times 10^{10}$ | $1.0 \times 10^{11}$ | $1.0 \times 10^{11}$ | $8.2 \times 10^{10}$ | $5.0 \times 10^{10}$ | $5.10 \times 10^{10}$ | $3.00 \times 10^{10}$ |
| Total α-activity | $2.4 \times 10^9$ | $2.5 \times 10^9$ | $2.2 \times 10^9$ | $2.1 \times 10^9$ | $2.2 \times 10^9$ | $2.16 \times 10^9$ | $2.37 \times 10^9$ |
| Discharges | | | | | | | |
| $^{24}Na$ | $7.8 \times 10^{14}$ | $9.6 \times 10^{13}$ | $7.0 \times 10^{13}$ | $6.2 \times 10^{12}$ | $7.6 \times 10^{13}$ | $9.91 \times 10^{13}$ | $1.24 \times 10^{14}$ |
| $^{32}P$ | $4.0 \times 10^{13}$ | $1.8 \times 10^{13}$ | $1.1 \times 10^{13}$ | $8.8 \times 10^{12}$ | $9.2 \times 10^{12}$ | $1.09 \times 10^{13}$ | $1.40 \times 10^{13}$ |
| $^{239}Np$ | $1.5 \times 10^{13}$ | $7.5 \times 10^{12}$ | $8.1 \times 10^{12}$ | $6.2 \times 10^{12}$ | $7.5 \times 10^{12}$ | $1.30 \times 10^{13}$ | $1.46 \times 10^{13}$ |

gressively decreased since 2000, and wastewater discharges of $^{51}Cr$, $^{54}Mn$ and $^{131}I$ have also decreased although there has been some variation between years. For $^{60}Co$, $^{134}Cs$ and $^{137}Cs$, there is no clear trend, but discharges were lower by one or two orders of magnitude in 2005 and 2006 compared to previous years. Trends in emissions and discharges from the Bilibino nuclear power plant are less clear.

Radionuclides are also released from the plant Mayak PA facility, the Siberian Chemical Combine, and the Mining and Chemical Complex at Zheleznogorsk. The wastewater discharges are transported to the Kara Sea by the Ob and Yenisey rivers. For the Mayak PA facility (see section 2.1.2) there has been considerable variability in emissions to the air, with $^{137}Cs$ emissions almost as high in 2006 as in 2001. In contrast, wastewater discharges appear to be declining. Emissions to the air from the Siberian Chemical Combine (Table 2·7) appear to have progressively decreased since 2000, while wastewater discharges are more variable and there is no apparent trend. Emissions to the air from the Mining and Chemical Complex of Zheleznogorsk (Table 2·8) peaked in 2001 and although they have since fallen they are still highly

variable, as are the wastewater discharges from this site.

Radionuclides are also emitted to the air from the Loviisa and Olkilouto nuclear power plants in Finland (Table 2·9). Emissions from both sites have been relatively stable since 2001, but with some variation between years.

## 2.1.6. Abrosimov Bay

From 1966 to 1981, solid radioactive waste, including spent nuclear fuel was dumped in Abrosimov Bay on the east coast of the Novaya Zemlya southern island. The first detailed survey of marine radioactive contamination in Abrosimov Bay was undertaken in 1994 during a joint Russian–Norwegian expedition (Strand et al., 1996, 1997a). The results of the survey indicated the need for periodic monitoring of radioactive contamination in this area, as well as in other areas in which solid radioactive wastes are dumped. The waste dumped ranges from containers with contaminated equipment to nuclear submarines with fuelled reactors. A second survey of radioactive contamination in Abrosimov Bay took place in 2002, eight years after the first survey (ISTC, 2003). Seawater

**Table 2·8.** Atmospheric emissions and discharges to water from the Mining and Chemical Complex at Zheleznogorsk (Bq/y).

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| Emissions | | | | | | | |
| $^{90}$Sr | $7.4 \times 10^8$ | $2.0 \times 10^{10}$ | $1.2 \times 10^9$ | $3.3 \times 10^8$ | $8.0 \times 10^8$ | $1.18 \times 10^9$ | $4.78 \times 10^8$ |
| $^{131}$I | $2.1 \times 10^8$ | $8.8 \times 10^9$ | $2.7 \times 10^8$ | $3.2 \times 10^8$ | $6.8 \times 10^8$ | $3.66 \times 10^8$ | $6.34 \times 10^8$ |
| $^{137}$Cs | $1.7 \times 10^9$ | $2.9 \times 10^9$ | $2.1 \times 10^8$ | $2.7 \times 10^8$ | $5.1 \times 10^8$ | $5.41 \times 10^8$ | $2.53 \times 10^8$ |
| Total α-activity | $7.2 \times 10^7$ | $1.8 \times 10^8$ | $3.4 \times 10^8$ | $4.5 \times 10^8$ | – | – | – |
| Discharges | | | | | | | |
| $^{32}$P | $8.0 \times 10^{12}$ | $9.1 \times 10^{12}$ | $7.9 \times 10^{12}$ | $8.4 \times 10^{12}$ | $1.3 \times 10^{13}$ | $1.11 \times 10^{13}$ | $1.10 \times 10^{13}$ |
| $^{131}$I | $3.2 \times 10^{10}$ | $4.2 \times 10^{10}$ | $3.7 \times 10^{10}$ | $3.3 \times 10^{10}$ | $3.6 \times 10^{10}$ | $3.54 \times 10^{10}$ | – |
| $^{137}$Cs | $3.1 \times 10^{10}$ | $3.1 \times 10^{10}$ | $1.7 \times 10^{10}$ | $1.1 \times 10^{10}$ | $7.2 \times 10v^9$ | $3.08 \times 10^{10}$ | $1.52 \times 10^{10}$ |
| Total α-activity | $1.9 \times 10^9$ | $1.3 \times 10^9$ | – | – | – | – | – |

**Table 2·9.** Atmospheric emissions from the Loviisa and Olkiluoto nuclear power plants in Finland (Bq/y).

| | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 | 2007 |
|---|---|---|---|---|---|---|---|
| Loviisa | | | | | | | |
| Noble gases [a] | $4.9 \times 10^{12}$ | $5.0 \times 10^{12}$ | $6.5 \times 10^{12}$ | $6.6 \times 10^{12}$ | $6.6 \times 10^{12}$ | $5.8 \times 10^{12}$ | $5.55 \times 10^{12}$ |
| $^3$H | $1.9 \times 10^{11}$ | $2.2 \times 10^{11}$ | $2.2 \times 10^{11}$ | $2.2 \times 10^{11}$ | $1.9 \times 10^{11}$ | $1.8 \times 10^{11}$ | $1.68 \times 10^{11}$ |
| $^{14}$C | $3.1 \times 10^{11}$ | $3.7 \times 10^{11}$ | $3.2 \times 10^{11}$ | $3.2 \times 10^{11}$ | $2.7 \times 10^{11}$ | $1.5 \times 10^{11}$ | $2.42 \times 10^{11}$ |
| Aerosols | $4.1 \times 10^7$ | $6.7 \times 10^7$ | $8.0 \times 10^7$ | $1.2 \times 10^8$ | $1.1 \times 10^8$ | $1.1 \times 10^8$ | $1.12 \times 10^8$ |
| Iodines [b] | <DL | $9.9 \times 10^5$ | $3.5 \times 10^6$ | $1.1 \times 10^7$ | $6.2 \times 10^4$ | $3.0 \times 10^5$ | $7.34 \times 10^5$ |
| Olkiluoto | | | | | | | |
| Noble gases [a] | $5.7 \times 10^{10}$ | $2.8 \times 10^{10}$ | $1.4 \times 10^{11}$ | < DL | $1.5 \times 10^{11}$ | $6.5 \times 10^{11}$ | $1.13 \times 10^{11}$ |
| $^3$H | $3.9 \times 10^{11}$ | $3.9 \times 10^{11}$ | $2.8 \times 10^{11}$ | $3.2 \times 10^{11}$ | $3.2 \times 10^{11}$ | $3.1 \times 10^{11}$ | $3.76 \times 10^{11}$ |
| $^{14}$C | $8.7 \times 10^{11}$ | $9.5 \times 10^{11}$ | $6.8 \times 10^{11}$ | $8.4 \times 10^{11}$ | $6.6 \times 10^{11}$ | $7.7 \times 10^{11}$ | $1.08 \times 10^{12}$ |
| Aerosols | $3.3 \times 10^7$ | $3.0 \times 10^7$ | $3.3 \times 10^7$ | $2.1 \times 10^7$ | $3.8 \times 10^7$ | $4.1 \times 10^7$ | $3.02 \times 10^7$ |
| Iodines [b] | < DL | $9.8 \times 10^6$ | $1.7 \times 10^7$ | < DL | $6.9 \times 10^7$ | $1.6 \times 10^8$ | $1.48 \times 10^7$ |

DL: detection limit; [a] noble gases expressed as $^{87}$Kr; [b] iodines expressed as $^{131}$I equivalent.

**Table 2·10.** Comparison of $^{137}$Cs, $^{90}$Sr and $^{239,240}$Pu activity concentrations in the water of Abrosimov Bay and the adjacent part of the Kara Sea in 1994 and 2002.

| Sampling horizon | $^{137}$Cs, Bq/m³ | | $^{90}$Sr, Bq/m³ | | $^{239,240}$Pu, mBq/m³ | |
|---|---|---|---|---|---|---|
| | 1994 | 2002 | 1994 | 2002 | 1994 | 2002 |
| Inner part of the bay | | | | | | |
| Surface | $4.2 - 6.8$ | $0.3 - 3.0$ | $2.2 - 3.5$ | $3.4 - 4.7$ | $4.4 - 6.8$ | $1.2 - 4.9$ |
| Near-bottom | $5.5 - 8.5$ | $2.9 - 3.2$ | $2.2 - 3.5$ | $3.5 - 6.0$ | $3.5 - 4.4$ | $6.2 - 6.4$ |
| Outer part of the bay | | | | | | |
| Surface | $4.7 - 7.0$ | $2.8$ | $1.9 - 2.7$ | $2.8$ | $3.5 - 4.2$ | $3.0$ |
| Near-bottom | $4.4 - 9.4$ | $4.3$ | $2.0 - 3.6$ | $2.5$ | $3.9 - 5.1$ | $3.5$ |

and sediment samples were obtained from the same sites in both surveys (Figure 2·10), which enabled a comparison of the two sets of data (Table 2·10).

The activity concentrations of $^{137}$Cs in seawater in 2002 had declined to around half the levels measured in 1994. This decrease was due to reduced global fallout of $^{137}$Cs, decreased $^{137}$Cs discharges from Sellafield, and the decline in the removal of Chernobyl-derived $^{137}$Cs from the Baltic area. For $^{90}$Sr, activity concentrations in seawater in 2002 were higher than in 1994 for the inner part of Abrosimov Bay, but had not changed for the outer part of the bay. The $^{239,240}$Pu activity concentrations in seawater were lower in 2002 than in 1994 except for near-bottom samples from the inner part of the bay where levels had actually increased.

The spatial distribution of $^{137}$Cs activity concentrations in surface sediments in 1994 (Strand et al., 1996, 1997a) and 2002 (ISTC, 2003) are compared in Figure 2·10. It is clear that activity concentrations are lower in 2002 than in 1994 for most of Abrosimov Bay and the adjacent Kara Sea. This indicates the absence of noticeable radioactive leakage from the solid radioactive waste containers in the eight-year period between the two surveys. In the absence of additional radioactivity input, this decrease in activity in the surface sediments is as would be expected and reflects the deposition of uncontaminated suspended sediments and radioactive decay. By 2002, specific activities of $^{60}$Co had also decreased across most of the bay and in the adjacent Kara Sea (Table 2·11).

It is impossible to make similar comparisons for $^{90}$Sr and $^{239,240}$Pu because activity concentrations for these radionuclides were mainly determined in samples collected by means of a remotely operated underwater vehicle (ROV) in close proximity to the dumped objects. However, the authors stress that low activity concentrations of these radionuclides were observed in bottom sediments across most of the bay in 2002 and that these are typical of activity concentrations characteristic of the Barents and Kara Seas in recent years.

Thus, the radiation situation for most of Abrosimov Bay improved between 1994 and 2002 and the results indicate that there had been no noticeable radioactive input to the marine environment from the solid radioactive wastes dumped in the bay. The $^{137}$Cs content of seawater in the bay in 2002 was about 3 Bq/m³ or less, which is lower than in
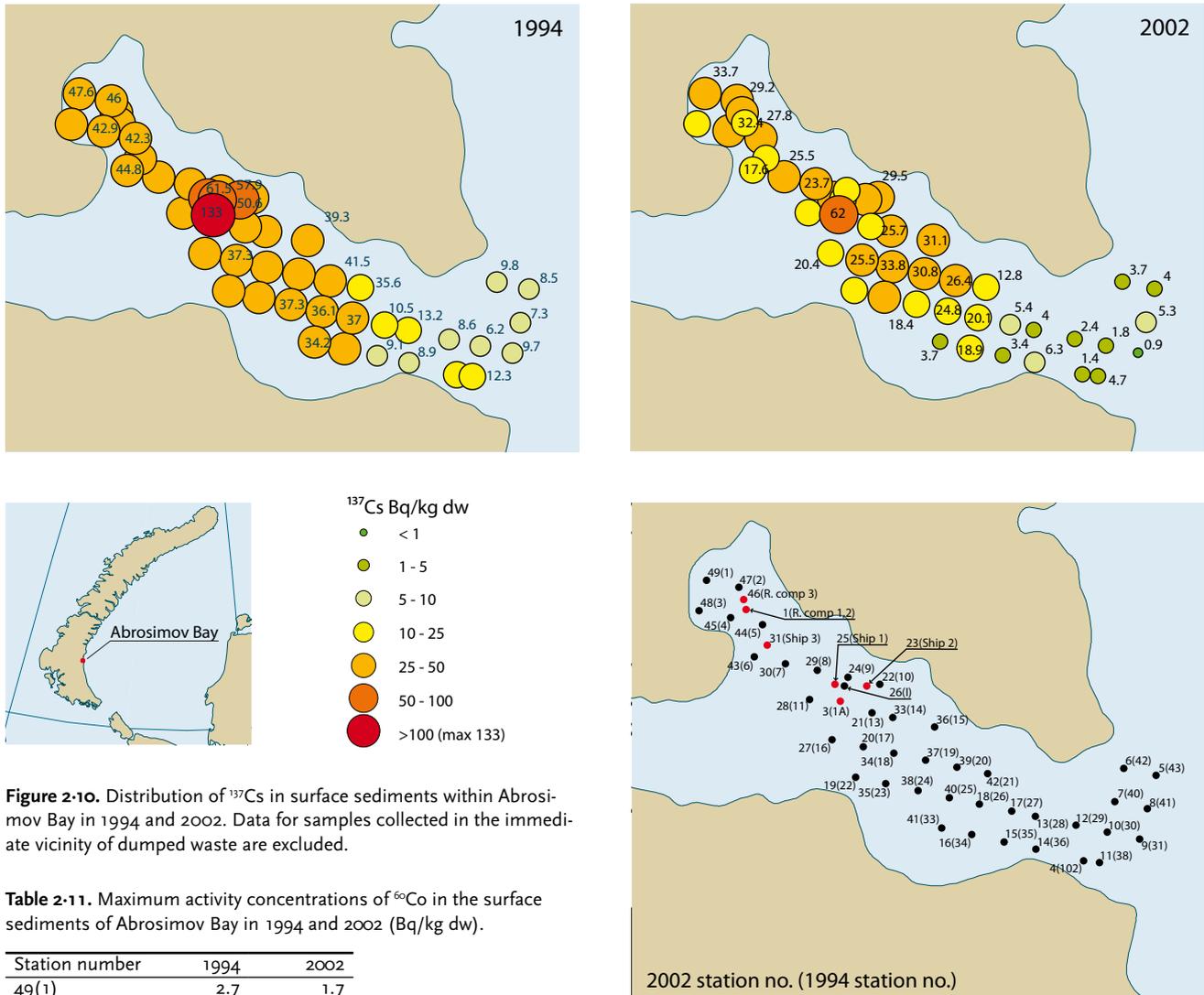
**Figure 2·10.** Distribution of ¹³⁷Cs in surface sediments within Abrosimov Bay in 1994 and 2002. Data for samples collected in the immediate vicinity of dumped waste are excluded.

**Table 2·11.** Maximum activity concentrations of ⁶⁰Co in the surface sediments of Abrosimov Bay in 1994 and 2002 (Bq/kg dw).

| Station number | 1994 | 2002 |
|---|---|---|
| 49(1) | 2.7 | 1.7 |
| 47(2) | 2.5 | 1.6 |
| 48(3) | 2.9 | < MDL |
| 45(4) | 2.9 | < MDL |
| 44(5) | 3.7 | < MDL |
| 43(6) | 2.9 | 2.3 |
| 30(7) | 2.7 | 0.7 |
| 29(8) | 1.1 | < MDL |
| 24(9) | 0.9 | < MDL |
| 22(10) | 1.6 | < MDL |
| 28(11) | 1.2 | < MDL |
| 21(13) | – | < MDL |
| 33(14) | 2.4 | < MDL |
| 36(15) | 0.7 | < MDL |
| 16(34) | – | < MDL |
| 27(16) | – | < MDL |
| 20(17) | – | < MDL |
| 34(18) | 0.5 | < MDL |
| 37(19) | – | < MDL |
| 39(20) | – | < MDL |
| 42(21) | < 0.5 | < MDL |
| 19(22) | – | < MDL |
| 35(23) | – | < MDL |
| 38(24) | 0.4 | < MDL |
| 40(25) | – | < MDL |
| 18(26) | – | < MDL |
| 17(27) | – | < MDL |
| 13(28) | – | < MDL |
| 41(33) | – | < MDL |
| 14(36) | – | < MDL |
| 15(35) | – | < MDL |

MDL: minimal detection limit (0.3 – 1.2 Bq/kg dw).

1994 (4 – 9 Bq/m³). Activity concentrations of ⁹⁰Sr in seawater (3.4 – 6.0 Bq/m³) and ²³⁹,²⁴⁰Pu in seawater (< 1.2 – 6.4 mBq/m³) in 2002 were effectively the same as in 1994 and, as for ¹³⁷Cs, are typical of those characteristic of the Barents and Kara Seas in recent years. For most of Abrosimov Bay and the adjacent Kara Sea, lower levels of ¹³⁷Cs activity were observed in surface sediments in 2002 than in 1994. In 2002, levels of ¹³⁷Cs, ⁹⁰Sr and ²³⁹,²⁴⁰Pu in surface sediments across most of Abrosimov Bay area did not exceed 40, 2.5 and 1.2 Bq/kg dw, respectively. Levels of ⁶⁰Co in surface sediments also decreased; in 2002 most surface sediment samples were below the detection limit. As in 1994, the greatest contamination by ¹³⁷Cs, ⁹⁰Sr, ⁶⁰Co and ²³⁹,²⁴⁰Pu in sediments was observed immediately adjacent to the dumped objects. Despite the data indicating low levels of contamination across most of the bay, regular monitoring in this and other areas of solid radioactive waste disposal in the Kara Sea is important, preferably at five to eight year intervals. Regular monitoring will ensure that any leakage from the containers is quickly discovered.

## 2.1.7. Thule

The effects of the plane crash at Thule air base in January 1968 have been reported in both previous AMAP assessments
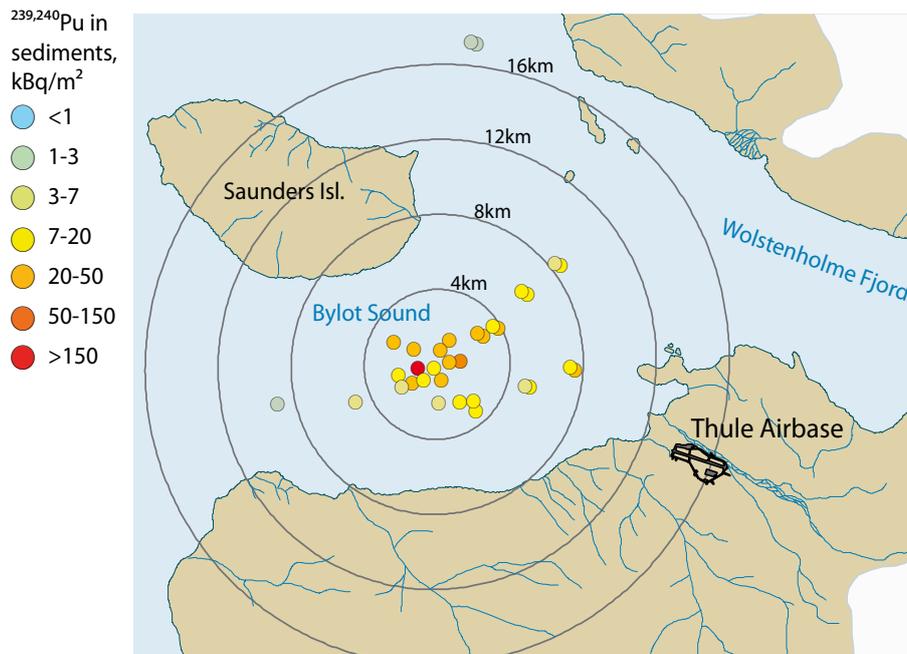
Figure 2·11. Locations for sediment sampling in Bylot Sound and observed values of deposition. The concentric circles indicate distances of 4, 8, 12 and 16 km from the point of impact of the B52 plane.

(AMAP 1997, 2004). This assessment reports on the latest investigation at the site, and is based on the work of Nielsen and Roos (2006).

Analyses of marine and terrestrial samples collected in August 2003 from Bylot Sound show that plutonium from nuclear weapons in the American B52 plane, which crashed onto the sea ice and then caught fire, persists in the environment at Thule.

Successful sampling of marine sediments was achieved by selecting sample sites using information from acoustic mapping of the seabed. The highest activity concentrations of plutonium were found in marine sediments at the site where the plane crashed (Figure 2·11). The distribution of plutonium in marine sediments is very inhomogeneous and is associated with hot particles, with activities of up to 1500 Bq $^{239, 240}$ Pu found. Sediment samples collected in Wolstenholme Fjord north of the accident site show plutonium activities of over an order of magnitude above background levels, which indicates the redistribution of plutonium after the accident.

The total plutonium inventory in the sediments was assessed based on systematic analyses that took into account hot particles. The inventory of $^{239, 240}$ Pu in the sediments within 17 km from the plane's point of impact (Figure 2·12) is estimated at 2.9 TBq (1 kg) with an approximate 95% confidence interval of 1.4 TBq to 6 TBq. This supports findings from sampling in 1997 which found that earlier investigations might have underestimated the total inventory of $^{239, 240}$ Pu by not systematically considering the contribution from hot particles. Earlier estimates of the inventory were about 1.4 TBq $^{239, 240}$ Pu.

Seawater samples show increased concentrations of particle-associated plutonium in near-bottom water in Bylot Sound and seaweed samples show increased activity concentrations in Bylot Sound compared to sites outside Bylot Sound. The increased activity concentrations are due to resuspension of plutonium-containing particles from the seabed and transport further away from the area. Continuous mixing of the sediments by benthic fauna has the
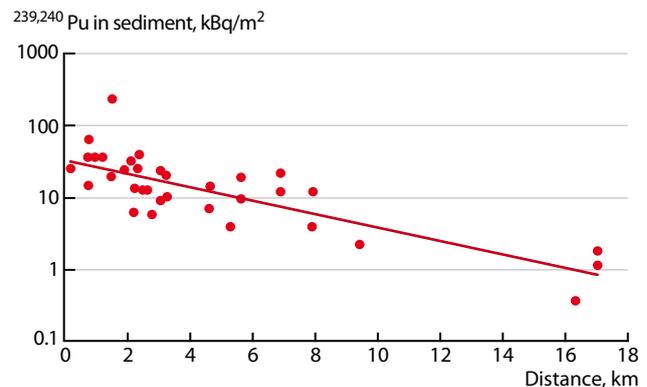


Figure 2·12. Deposition of $^{239,240}$Pu at locations in Bylot Sound by distance from the point of impact of the B52 plane. An exponential function is fitted to the data as indicated (Nielsen and Roos, 2006).

effect that plutonium concentrations in the surface sediment layers are high in general and not buried under uncontaminated sediment.

Activity concentrations of plutonium in seawater, seaweed and benthic animals in Bylot Sound are low but clearly above background levels.

All soil samples collected from Narssarssuk show accident plutonium with levels above background. Plutonium is very inhomogeneously distributed and associated with particles in the surface layers. Hot particles were found in soil with activities of up to 150 Bq $^{239, 240}$ Pu.

Plutonium in the marine environment at Thule presents an insignificant risk to man. Most plutonium remains in the seabed under Bylot Sound far from settlements and under relatively stable conditions. Activity concentrations of plutonium in seawater and animals are low and cannot present any risks to human health, even by the consumption of shellfish at the highest concentrations. However, plutonium contamination of surface soil at Narssarssuk could constitute a small risk to people visiting the area if radioactive particles are resuspended in the air such that they might be inhaled.
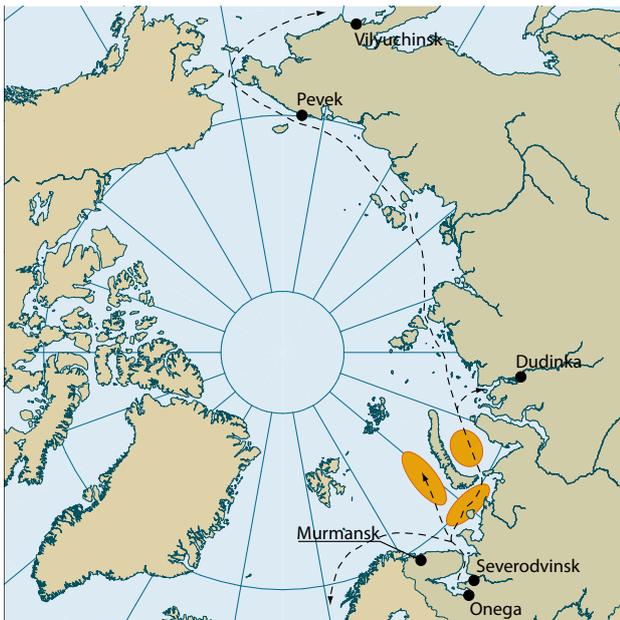
**Figure 2·13.** Potential sites of floating nuclear power plants within the Arctic region. Orange shaded regions indicate areas of oil or gas exploration where floating nuclear power plants may be employed as power sources. Dashed lines indicate potential routes from an assumed manufacturing/servicing centre in the Arkhangelsk region to international customers or to Russian sites outside the Arctic.

## 2.2. Potential sources

### 2.2.1. Floating nuclear power plants

From the mid- to late 1990s sporadic media reports have indicated Russian intentions regarding potential development of floating nuclear power plants for use in the Arctic regions and possibly for export. During the 1980s, Russia showed significant interest in low capacity power plants for the provision of power and heat in remote Arctic areas and this period began to see the description of Russian designs for potential floating nuclear power plants (see for example Golovin et al., 1981). Russia's nuclear industry has undergone and continues to undergo a major restructuring that is preceding the intended large-scale expansion of nuclear energy provision within Russia. Global environmental concerns about fossil fuel carbon emissions, supplies of those fuels, security concerns and demands for cheap energy for a range of problems such as desalination have resulted in a focus on floating nuclear power as a potential solution. This is clear in a number of initiatives and projects conducted by bodies such as the International Atomic Energy Agency (IAEA, 1997, 2000, 2002, 2004, 2007a,b) in which a range of countries have participated.

In 1991, JSC Malaya Energetika opened a competition for the best design for a low-capacity nuclear power plant; this was won by a design for a floating nuclear power plant submitted by Atomenergo. In 1996, reports announced intentions for floating power plants at a variety of locations in the Russian north. Further reports appeared periodically as plans appeared to undergo various modifications until a series of official announcements made it clear that the keel of the floating nuclear plant *Academician Lomonosov* had been

laid at the Sevmash shipyards at Severodvinsk on 15 April 2007, for completion by 2010.

A wide range of estimates have been made as to the total number of plants to be built, ranging from one per year to a total of fifteen over an unspecified period. Estimating the total number is difficult for several reasons. The floating power plant idea appears to have been developed at least partly as a commercial product. Russia has marketed these plants internationally for several years and there are indications that countries such as China are involved in the project and that a number of other countries appear to have expressed interest in the concept. An international conference entitled 'Small Power Plants: Results and Prospects' was held in Moscow (10 – 11 October 2001) at which Minatom stated that some 33 towns in the Russian far north and far east would be supplied by small nuclear power plants and that of that number, 11 would be floating at the following locations: Severodvinsk and Onega (Arkhangelsk Oblast), Vilyuchinsk (Kamchatka Oblast), Pevek (Chukotka Autonomous Okrug), Sovetskaya Gavan and Nikolayevsk-na-Amure (Khabarovsk Kray), Nakhodka, Olga and Rudnaya Pristan (Primorskiy Kray), Dudinka (Taymir Autonomous Okrug), and the site of the Trukhanskaya hydro-electric plant (Evenkiyskiy Autonomous Okrug). Russia has also indicated the possibility of employing such plants to drive oil and gas extraction activities in the rapidly opening-up and resource-rich Barents Sea.

It is likely that the industrial infrastructure for the construction, refuelling, servicing and decommissioning of such plants will be based, at least initially, around the Kola Peninsula/Arkhangelsk region as this is already the site of support centers for the Russian military and civilian nuclear fleets. Should numbers of floating nuclear power plants be built, not only will there be an increase in the numbers of reactors in the Arctic but there will also be a concomitant increase in nuclear traffic within and potentially in and out of the Arctic. Such traffic would consist of vessels loaded with fresh fuel and possibly more significantly, spent nuclear fuel and nuclear waste on the return journey. In this context, the potential for floating nuclear power plants as an internationally viable product is of some interest. Figure 2·13 shows the potential sites of floating nuclear power plants within the Arctic.

### 2.2.2. Transport of spent nuclear fuel along the Norwegian coastline

In Norway, public concern has been raised about the import of spent nuclear fuel for reprocessing and storage because this is likely to result in sea transport of spent nuclear fuel along the Norwegian coastline. Concern has also been raised about the possibility of a new transport route along the northern coast of Russia for spent nuclear fuel, due to less sea-ice cover in the Arctic. In Norway, major economic and cultural interests are connected to the production and export of marine food products, and past experience has shown that even rumors of radioactive contamination in seafood can lead to economic consequences for producers.

In 2007, the Norwegian Radiation Protection Authority published a report in which the radioecological consequences of potential accidents during transport of spent nuclear fuel along Norway's coastline were modeled. The model
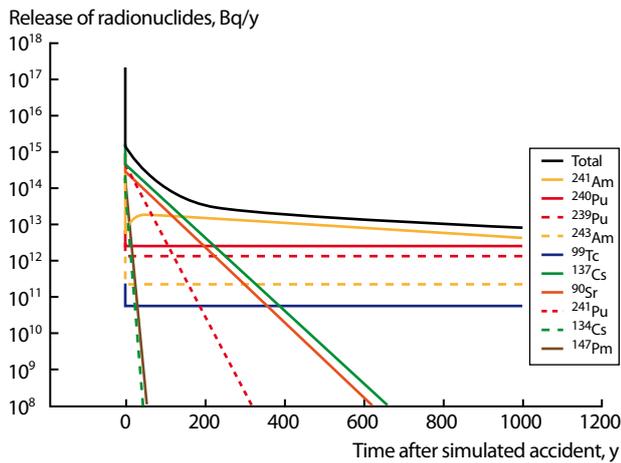
**Figure 2·14.** The worst case release scenario modelled by the Norwegian Radiation Protection Authority for the transport of spent nuclear fuel along Norway's coastline.
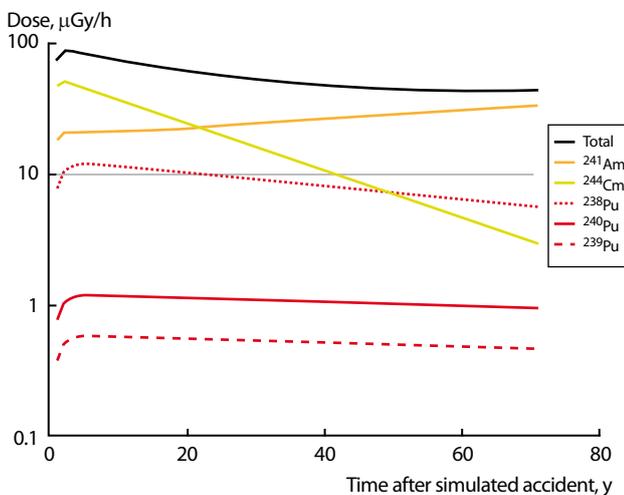


**Figure 2·15**. Doses to a species of polychaete worm in comparison with screening dose 10 μGy/h.

considered doses to critical groups, concentrations of radionuclides in seafood species, and doses to marine organisms based on a 'worst-case scenario'. The scenario was that a ship loaded with 100 fuel packages of standard fuel assembly of 30.1 PBq radioactivity sank at an accident location with the highest radioactivity sensitivity index *RS* (Iosjpe et al., 2003); i.e., an area considered to be most vulnerable to radioactive contamination and where the consequences of the accident would be highest (Figure 2·14). The source term comprised an immediate 10% release of the total inventory, followed by annual releases of 0.09% of the inventory.

### 2.2.2.1. Concentrations of radionuclides in biota/seafood

The calculated activity concentrations of radionuclides in biota (fish, shellfish and crustaceans) were evaluated with regard to international guideline levels (CAC, 2006), where the radionuclides were divided into four groups based on the type and potential damage following intake. The results indicated that activity concentrations of radionuclides in biota remain below acceptable guideline levels for the radionuclides in groups 3 and 4, while those for some group 1 and 2

radionuclides became higher than guideline values. Results for radionuclides in groups 1 and 2 are shown in Figure 2·16.

Calculated concentrations of group 2 radionuclides in fish and crustaceans were lower than guideline levels. However concentrations in shellfish were calculated to be higher than guideline levels during the first three years after the accident. Concentrations of radionuclides for group 1 are higher than guideline levels for all reference biota. The most significant concentrations were found in shellfish; concentrations of radionuclides in shellfish were higher than guideline levels during the first four years after the accident and for more than ten years when considering guidelines for infant foods.

### 2.2.2.2. Doses to the critical group

Doses were calculated for the critical group using an investigation of consumption patterns for populations living on the coast of Norway (Bergsten, 2003). Maximal consumption for seafood was reported as 200 g/d for fish, 40 g/d for crustaceans and 4 g/d for shellfish. Model calculations showed that maximal impacts to total dose from fish, crustacean and shellfish were 0.3 mSv/y, 0.2 mSv/y and 0.1 mSv/y, respectively, and that the maximum total dose-rate for the critical group is 0.6 mSv/y. This demonstrates that the doses to the critical group are most likely to be below the international personal guideline level of 1 mSv/y.

### 2.2.2.3. Doses to marine organisms

Doses were also calculated for marine organisms. There is international consensus that dose rates of 10 μGy/h are non-dangerous for biota (Brown et al., 2006a). Comparing results of dose calculations for the species considered with a screening dose of 10 μGy/h indicates that maximal doses to biota are generally below the recommended level. An example of doses exceeding the recommended level is provided by a species of polychaete worm; in this case doses exceed the screening dose by up to one order of magnitude over a long period of time (Figure 2·15). This is explained by the habitat of the polychaete worm. This species lives in sediments that generally have high distribution coefficients for radionuclides held in the sediment/seawater. It is important to note, however, that doses to the polychaete worm exceeding the screening dose of 10 μGy/h do not automatically mean damage at the population level. Nevertheless, the result may merit that the situation would have to be taken under special consideration.

### 2.2.2.4. Concluding comments

In spite of the very conservative scenario, the collective dose rates to people and to the critical group are not higher than recommended guidelines of 1 mSv/y. Results did indicate, however, that activity concentrations of radionuclides for some marine organisms exceeded guideline levels after the radioactive releases. Elevated levels of radionuclides in marine food products may lead to economic consequences in a market that is very sensitive to reports of contamination. However, health consequences due to the elevated radiation doses in humans were shown to be insignificant. Comparing dose calculations for biota with screening dose limits agreed

Comparison of activity levels in fish (left) and crustanceans (right) with international recommendations for group 1 radionuclides
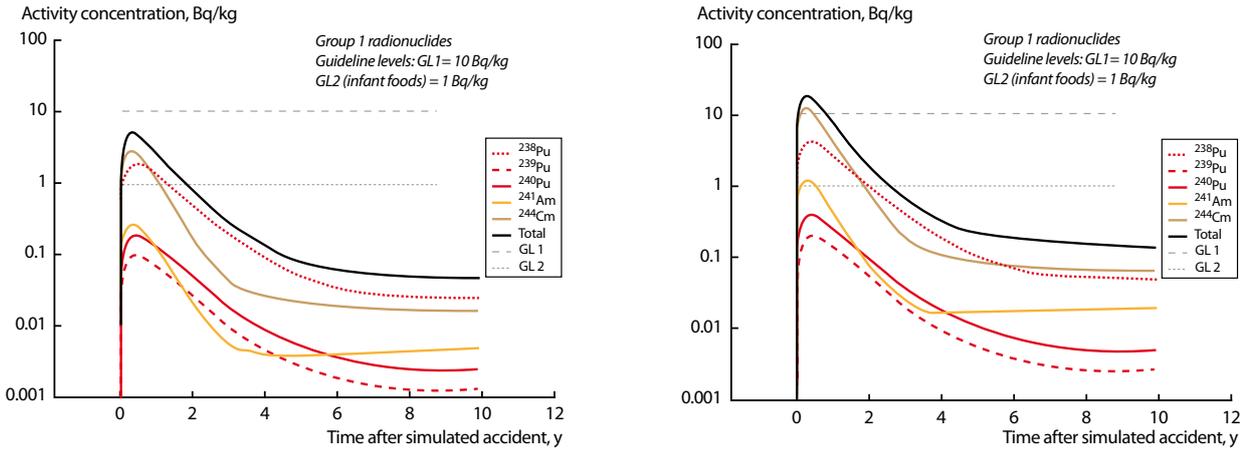
Comparison of activity levels in fish (left) and crustanceans (right) with international recommendations for group 2 radionuclides
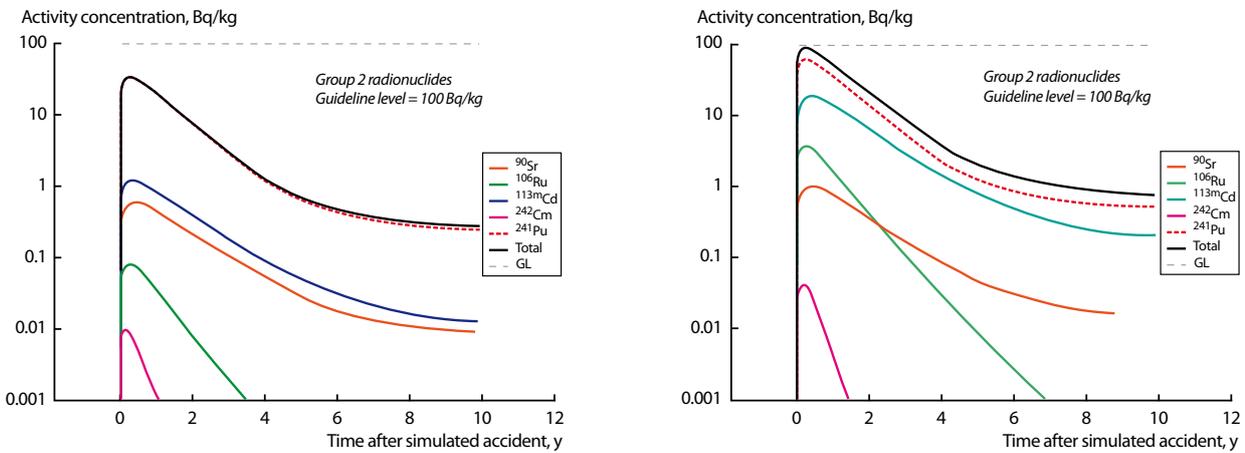
Comparison of activity levels in molluscs with international recommendations for group 1 (left) and 2 (right) radionuclides
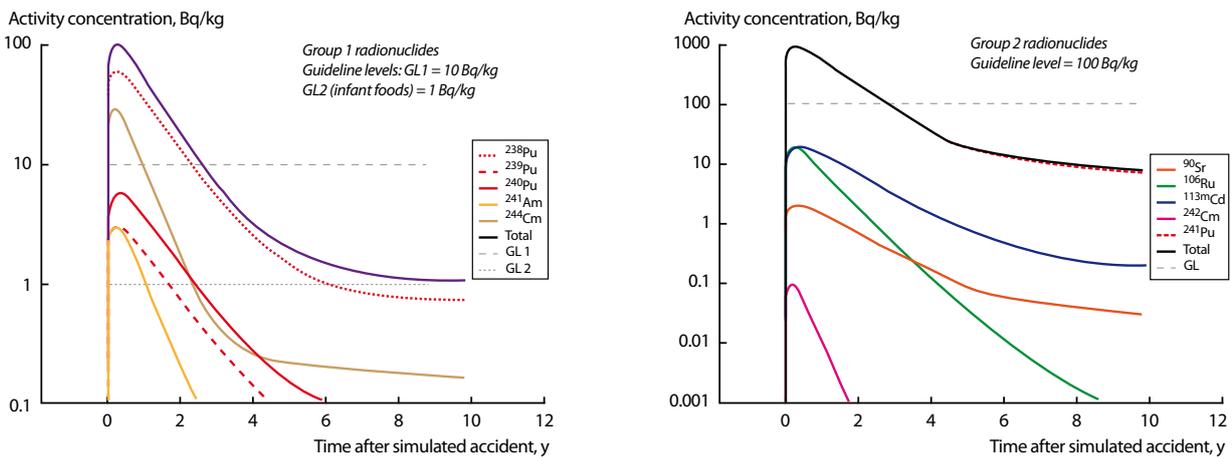
**Figure 2·16.** Comparison of activity concentrations of CAC group 1 and group 2 radionuclides in fish, crustaceans and shellfish against international guidelines.

within the ERICA project (Brown et al., 2006a) indicates that doses to the majority of marine organisms are far below the level where adverse effects are expected (i.e., the screening dose of 10 µGy/h). However, doses to some marine organisms can be much higher (up to one order of magnitude) than the screening dose of 10 µGy/h over long periods, which

means that statistically significant effects could be expected for these organisms (Real et al., 2004).

*Chapter 3*

# TENORM

## 3.1. Introduction

This chapter deals with naturally-occurring radioactive material that arises through a number of industrial activities, including the front end of the nuclear fuel cycle. Certain raw materials used in industrial processes, often in bulk quantities, contain elevated levels of naturally-occurring radioactive elements. After processing, the activity concentrations may be altered and enhanced, and products, by-products or wastes may become a potential hazard for workers, the public and the environment. Examples of such technologically enhanced naturally-occurring radioactive materials (TENORM) from non-nuclear industries include radium-containing scale from the oil and gas industry and radium-containing gypsum from phosphoric acid production. In the former, radium has co-precipitated with chemically similar elements (calcium, barium, strontium) and can be found in the scale at activity concentrations of up to several hundred Bq/g. The amount of scale formed, however, is relatively low compared with the millions of tonnes of phosphogypsum generated through phosphoric acid production, where the alpha activ-

ity concentration is of a similar order to that found in the phosphate ore; from 0.2 Bq/g to 1.5 Bq/g. Other major types of industrial activity where TENORM may be encountered are metal and mineral mining (including uranium mining) and processing, and energy production from coal, peat and natural gas. Many of these potential sources of TENORM can be found or have been present in the AMAP area, or are planned to be established in the AMAP area in the future. For example, oil and gas exploitation in the Barents Sea. There are several problems that may be encountered in TENORM industries that are related to the presence of natural radioactivity. Areas of concern include the handling of raw materials and exposure of workers, and sometimes the public, to radiation during processing, and exposure of workers, the public and the environment during waste handling and disposal.

### 3.1.1. Natural radioactivity

Radioactive elements of natural origin are found throughout the environment at varying concentrations. They are grouped according to their origin: cosmogenic radionuclides (e.g., $^{14}$C,
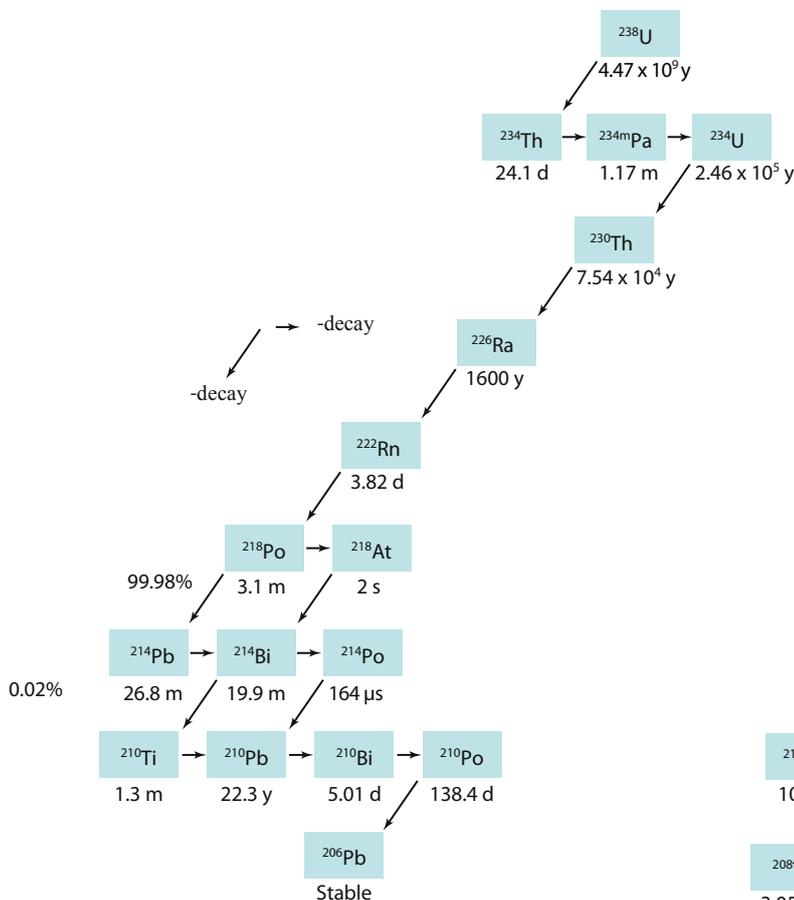


**Figure 3.1.** The uranium decay series.
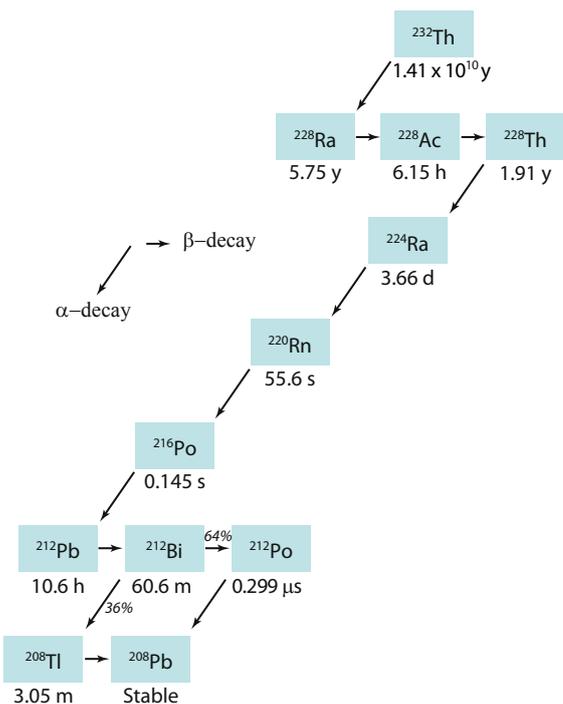
**Figure 3.2.** The thorium decay series.

**Table 3·1.** Typical levels of $^{238}$U and $^{232}$Th in some common rock types.

| Rock type | Name | $^{238}$U, Bq/kg | $^{232}$Th, Bq/kg |
|---|---|---|---|
| Igneous | Granite [a] | 30 − 83 | 32 − 134 |
| | Granite [b,c] (rich in uranium and thorium) | 100 − 500 | 40 − 350 |
| | Basalts [a] | 1.4 − 14 | 0.8 − 20 |
| | Phosphates [d] | 10 − 250 | < 10 − 500 |
| Metamorphic | Gneiss [a] | 35 | 20 − 110 |
| | Gneiss [b,c] | 25 − 130 | 20 − 80 |
| Sedimentary | Carbonatite [b,c] | 10 − 650 | 40 − 10 000 |
| | Phosphates [a,d,e] | 320 − 4 800 | < 10 − 150 |
| | Shale [c,f] | 10 − 120 | 8 − 60 |
| | Middle Cambrian alum shale [d,f] | 120 − 600 | 8 − 40 |
| | Upper Cambrian or Lower Ordovician alum shale [d,f] | 600 − 4 500 | 8 − 40 |

[a] Gascoyne (1992); [b] the Radiation Protection Institutes in Denmark, Finland, Iceland, Norway and Sweden (2000); [c] typical for the Nordic countries; [d] Karhunen and Vermeulen (2000); [e] UNSCEAR (1982); [f] UNSCEAR (1993).

**Table 3·2.** Typical levels of naturally-occurring radionuclides in seawater. Source: IAEA (1988), Brown et al. (2004b).

| | Activity concentration, Bq/m³ | |
|---|---|---|
| | Typical | Normal range |
| $^3$H | 50 | 25 − 100 |
| $^{14}$C | 6 | 5 − 7 |
| $^{40}$K | 18 000 | 9 000 − 36 000 |
| $^{210}$Po | 2 | 1 − 4 |
| $^{226}$Ra | 2 | 1.5 − 3 |
| $^{228}$Ra | 1 | 0.25 − 4 |
| $^{228}$Th | 0.05 | 0.025 − 0.1 |
| $^{230}$Th | 0.02 | 0.1 − 0.04 |
| $^{232}$Th | 0.001 | 0.0005 − 0.002 |
| $^{238}$U | 40 | 20 − 80 |

$^3$H, and $^7$Be) and serial and non-serial primordial radionuclides (uranium decay series, thorium decay series, and $^{40}$K). In the context of TENORM industries, the relevant radionuclides are serial primordial radionuclides in the uranium and thorium decay series and to a certain extent the single primordial $^{40}$K.

### 3.1.2. Primordial radionuclides

Primordial radionuclides are radionuclides that have been present on earth since it was created. They are characterised by their long half-lives, which are at least of the order of the age of the earth. Primordial radionuclides with shorter half-lives exist only as decay products that can be traced to a parent nuclide with a long half-life.

### 3.1.3. Serial radionuclides

Three decay series exist in nature: the uranium series (Figure 3·1), the thorium series (Figure 3·2) and the actinium series; with the uranium series and the thorium series the most abundant and important in the context of NORM (naturally-occurring radioactive materials) industries.

There are some important differences between the uranium and thorium decay series. The thorium decay series includes decay products with shorter half-lives than the uranium decay series, which means that after separation from the parent nuclide, secular equilibrium will be established in a shorter period of time. Another important difference concerns the inert gas radon isotopes $^{222}$Rn (radon) and $^{220}$Rn (thoron). Rn-222 and its short-lived daughters are the main contributors to the dose to man from natural sources. Inhalation of airborne radon isotopes and their short-lived daughters, especially indoors, results in irradiation of the respiratory tract, predominantly with high-LET (linear energy transformer) alpha radiation that may cause lung cancer. Because of its inert properties radon has the ability to escape from the matrix in which it was formed and reach open air and because of the shorter half-life of thoron (55 s) compared to radon (3.8 d), thoron has less time to reach the open air before it decays.

## 3.2. Typical levels of natural radioactivity

Activity concentrations of uranium and thorium decay series radionuclides may vary considerably from place to place depending on the geological characteristics at the location. Typical ranges of activity concentrations in some common rock types are given in Table 3·1.

Naturally-occurring radionuclides also show large differences in solubility in seawater. While uranium(VI) forms soluble carbonate complexes, which are found in relatively high concentrations in seawater, other radionuclides such as $^{232}$Th(IV) are found in low activity concentrations. Typical activity concentrations of naturally-occurring radionuclides in seawater are listed in Table 3·2.

## 3.3. TENORM industries

Materials containing natural radioactivity, such as uranium ore and produced water, can be moved from their original locations and modified by chemical or physical processes during human activities, resulting in changes in exposure conditions. A large number of industries process material with elevated levels of natural radioactivity. Important examples include the phosphate industry, the oil and gas industry, mineral extraction industries, metal ore processing industries and energy production from coal, peat and natural gas combustion. In contrast to the situation for industries involved in the nuclear fuel cycle, where the material is used because of its fissile or radioactive properties, the radioactivity of the materials processed is not of primary interest and is, almost without exception, undesired, and the awareness of potential problems concerning natural radioactivity may be low. It is

**Table 3.3.** Activity concentration of radium in produced water.

| | $^{226}$Ra, Bq/L | | $^{228}$Ra, Bq/L | | Source |
|---|---|---|---|---|---|
| | Range | Average | Range | Average | |
| US Gulf Coast | < 0.002 − 58 | 11.7 | 0.02 − 59 | 15.5 | Kraemer and Reid, 1984 |
| The Netherlands | < 2 − 302 | | < 1 − 20 | | Van Hattum et al., 1992 |
| Brazil | < 0.01 − 6.0 | | < 0.05 − 12.0 | | Jerez Vegueira et al., 2002 |
| USA, Poland and Austria | 0.05 − 191 | | | | Fisher, 1998 |
| Louisiana, USA | < DL − 34.4 | 5.9 | < DL − 34.3 | 6.1 | Hamilton et al., 1991 |
| USA | 0.1 − 21.6 | 9.7 | 0.7 − 21.7 | 10.2 | Hamilton et al., 1991 |
| US Gulf Coast | 2 − 55 | | 2.6 − 22 | | Lagera et al., 1999 |
| Denmark | < DL − 11.1 | | | | Varskog, 2003 |

**Table 3.4.** Activity concentrations of $^{226}$Ra, $^{228}$Ra and $^{210}$Pb in solid waste and sludge from the Norwegian oil and gas industry.

| Waste type | $^{226}$Ra, Bq/g | | $^{228}$Ra, Bq/g | | $^{210}$Pb, Bq/g |
|---|---|---|---|---|---|
| | Average | Range | Average | Range | |
| Hard scale | 21 | 4.0 − 39 | 12 | 2.6 − 33 | < 0.3 |
| Porous scale | 12 | 0.3 − 24 | 8.2 | 0.3 − 19 | < 0.2 |
| Sand | 4.0 | < 0.1 − 22 | 2.5 | < 0.1 − 13 | < 0.5 |
| Sludge | 2.5 | < 0.1 − 4.7 | 2.1 | < 0.1 − 4.6 | < 0.7 |

typical for many of these industries that very large amounts of material are used or processed, which often leads to large amounts of waste material, which in turn results in certain challenges concerning, for example, in waste management.

Because some of these industries have the potential to give rise to doses of the same order of magnitude as industries involved in the nuclear fuel cycle (which are under strict regulatory control, and where the potential hazard is of course much greater), increasing attention has been paid to this subject in recent years.

### 3.3.1. Oil and gas industry

During oil and, to a lesser extent, gas production, large volumes of water are co-extracted with the hydrocarbons from the reservoirs. This water may be formation water that has been trapped within the reservoir for millions of years, or a mixture of seawater and formation water if seawater has been injected into the reservoir. Produced water has been shown to contain elevated concentrations of radioactive isotopes, especially radium isotopes. Reducing conditions within the reservoir mean that the produced water generally has low concentrations of uranium isotopes. A correlation between salinity and concentrations of radium isotopes has been observed by several authors (Kraemer and Reid, 1984; Fisher, 1998; Wiegand and Feige, 2002), which is explained by increased competition with other cations for adsorption sites when the salinity of the water is high. At high sulfate concentrations, the low solubility product of $BaSO_4$ (barium sulfate), results in low radium concentrations. An explanation of the high radium concentrations in oilfield brines is that the low redox potential ensures low sulfate concentrations, which in turn permit higher radium concentrations in the water (Bloch and Key, 1981; Zukin et al, 1987).

Levels of radioactivity in produced water, especially the most abundant, long-lived radium isotopes, $^{226}$Ra and $^{228}$Ra, have been reported by several authors (Jonkers et al., 1997; Fisher, 1998; Strålberg et al., 2002; Neff, 2002). The activity concentrations range from below detection limits to several hundred Bq/L. Most concentrations reported are in the lower range, up to a few tens of Bq/L (see Table 3.3).

A survey of radium concentrations in produced water from all installations on the Norwegian continental shelf in 2003 (NRPA, 2005) showed an average activity concentration of 3.3 Bq/L (range < DL – 16 Bq/L) and 2.8 Bq/L (range < DL – 21 Bq/L) of $^{226}$Ra and $^{228}$Ra, respectively.

Produced water discharged to the sea will be rapidly diluted in well-mixed waters (Neff, 2002). The dilution upon discharge can be divided into two phases: a near-field phase, describing the discharge plume during the first few minutes after discharge, and a far-field phase. The dilution in the near field is due to, for example, turbulence caused by discharge momentum and plume buoyancy (Jirka et al., 1983). The far-field mixing occurs at a slower rate than in the near-field phase. In the North Sea, thermal stratification occurs during summer (the thermocline is at about 50 m), while the water column is well-mixed during winter (Ducrotoy et al., 2000). This can lead to a lower dilution of the discharge plume during summer. Chemical reactions between ions in the seawater and in the discharged produced water may also occur, leading to the formation of insoluble compounds. The formation of $BaSO_4$ from $Ba^{2+}$ ions in the produced water and $SO_4^{2-}$ ions in the seawater is of importance for radium. If $BaSO_4$ is formed near the discharge point, some of the $Ra^{2+}$ ions may co-precipitate and be removed from solution and transferred to the sediments (Hamilton et al., 1991). For produced water with a high $Ba^{2+}$ concentration, the fraction of the radium that co-precipitates with $BaSO_4$, may be significant. However, no studies investigating this could be found in the literature. Jerez Vegueria et al. (2002) investigated sediments and seawater close to two offshore platforms (discharging about 30 MBq/d and 41 MBq/d) at the *Bacia de Campos* oilfield (Brazil) and concluded that both sediments and seawater showed normal background levels, even at the closest sampling distance of 250 m from the platforms.

The radium that follows the produced water in the production stream may co-precipitate with other alkaline earth metals, such as barium, calcium and strontium as sulfates and carbonates, and form scale inside pipes and valves. Activity concentrations of $^{226}$Ra and $^{228}$Ra in scale can vary from very low up to several hundred Bq/g. Activity concentrations in

different solid wastes and sludge from the Norwegian oil and gas industry were reported by Strand (2004; see Table 3·4).

### 3.3.1.1. Northern Canada and Alaska

Commercial exploitation of oil reserves in northern Canada began in the 1920s at Norman Wells in the Northwest Territories (AMAP, 2008). Much of this oil was shipped southward to satisfy energy needs during the Second World War. The completion of the Trans-Alaska pipeline in 1977 allowed the transport of large quantities of offshore oil from Prudhoe Bay, Alaska. At this time major oil and gas reserves were being discovered in the Mackenzie Delta and Beaufort Sea areas. By the end of 2004, a total of 15 billion ($15 \times 10^9$) barrels of oil had been produced in Alaska with a further 250 million barrels produced in northern Canada. The expected completion of the Mackenzie Valley gas pipeline in 2014 will greatly increase this production activity.

### 3.3.1.2. Norwegian Sea

Nine oil and gas fields were in production in the Norwegian Sea as of May 2008. The discharges of radionuclides to the sea are mainly in the form of produced water during the operational phase and cuttings from drilling operations. In 2007, the total discharges of $^{226}$Ra and $^{228}$Ra through produced water from these fields were 46 GBq and 38 GBq, respectively (OLF, 2008).

## 3.3.2. Uranium mining

During uranium mining huge amounts of waste are produced in the form of waste rock and uranium mill tailings. Waste rock is generated during open pit mining or underground mining when rock or low grade ore are removed in order to reach the uranium-rich deposits. The content of uranium in ore normally ranges from 0.1% up to 40% (IAEA, 1992). Compared to normal rock, waste rock from uranium mining areas has elevated uranium levels and can constitute a radiological problem due to leaching of radionuclides, dust generation and radon emanation.

Uranium mill tailings are the waste, often in the form of sludge, that is generated after the uranium ore has been milled and treated with sulfuric acid (sometimes alkaline leaching is used) in order to extract uranium from the ore. Alternatives to leaching in uranium mills are *in-situ* leaching and heap leaching. After most of the uranium has been extracted from the milled ore about 85% of the original activity in the ore (non-uranium radionuclides and a few percent of uranium that has not been extracted) remains in the tailings. The tailings are often disposed of in ponds or piles where they constitute a potential radiological hazard due to external gamma radiation, dust, radon emanation and groundwater contamination. In addition to the radiological hazard there are often environmental problems associated with the heavy metals in the ores.

### 3.3.2.1. Northern Canada

Northern Canada is rich in mineral resources, including uranium. The uranium industry, which includes mining, mill-

ing and transport, has been the greatest single contributor to local levels of TENORM in the Canadian Arctic environment. There are several abandoned and decommissioned uranium mines in the Northwest Territories. In addition, a number of shipping points along the Northern Transportation Route were contaminated with low levels of natural radionuclides from the loading and unloading of uranium ores. In Canada today, active uranium mining continues only in the northern part of Saskatchewan. However, with the worldwide resurgence of nuclear power and the increasing demand for uranium, it is likely that uranium mining activities will be revived in the far north of Canada.

### 3.3.2.1.1. Port Radium

The history of uranium mining in northern Canada began in 1930, with the discovery of pitchblende ore at the Port Radium site (66°05' N; 118°02' W) on the east shore of Great Bear Lake, 440 km north of Yellowknife, Northwest Territories, and 265 km from the nearest community, Déline (population 550; predominantly Sahtú Dene people). From 1932 to 1940, the mine was operated for the extraction of radium, used in the production of luminous dials and for cancer treatment. In 1942, operations were switched to uranium production in support of the Manhattan Project. Uranium mining continued until 1960, when declining prices and reduced ore quality led to the closure of the mine. The mine was subsequently re-opened between 1964 and 1982 for the production of silver.

A total of 6200 tonnes of uranium were extracted over the lifetime of the mine. The operations left about 910 000 tonnes of tailings, of which about a fifth was deposited in depressions around the mine site with the remainder deposited nearby on the bottom of Great Bear Lake. After final closure in 1982, most of the site was covered with waste rock.

A comprehensive environmental survey of the Port Radium site was conducted between 2001 and 2004 as part of the Canada/Déline Action Plan (CDUT, 2005). This Action Plan was undertaken as a partnership between the federal government and the Déline First Nation for the purpose of addressing health concerns surrounding the mining operation. Results from the survey showed concentrations of uranium series radionuclides up to 37 000 Bq/kg in parts of the exposed tailings. Gamma radiation levels on the site varied from background (100 – 150 nGy/h) to a maximum of 740 nGy/h. Outdoor radon levels at the mine site were slightly elevated, with activity concentrations of up to 44 Bq/m³, compared to a background value of about 4 Bq/m³ in that area. Water samples from Great Bear Lake taken just offshore from the mine site showed some elevation of several trace metals, with an excess of $^{228}$Ra at one location. Fish from the lake showed no detectable radionuclides. The CDUT report recommended that the Port Radium site be fully remediated as soon as possible and detailed a plan to achieve this goal.

### 3.3.2.1.2. Northern Transportation Route

The Port Radium mine site is not accessible by land transportation routes. In the early years of mine operation, much of the ore was flown out by aircraft. However, as production increased, a water transportation route was developed, leading from Port Radium along the length of Great Bear

Cumulative dose, mSv
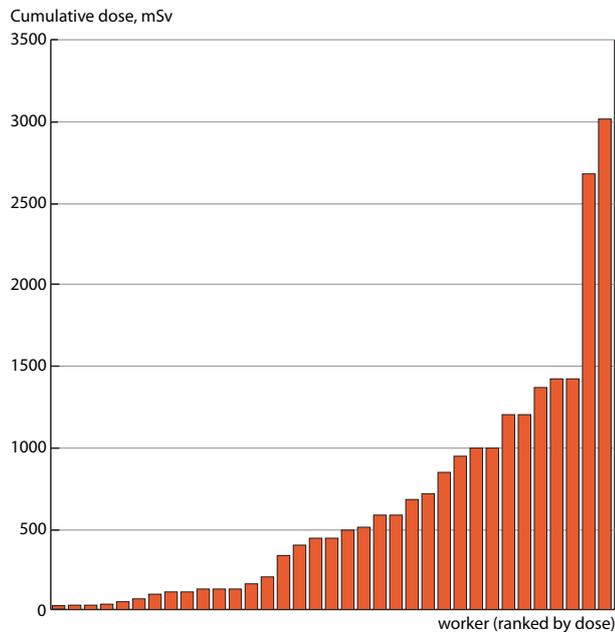


**Figure 3·3**. Reconstructed lifetime radiation doses to 35 ore handling workers.

Lake, through a system of rivers, to eventually reach a railway access point at Fort McMurray, Alberta. This was termed 'The Northern Transportation Route'. The total length of the route was 2100 km and involved a number of trans-shipment points, where the material was off-loaded from barges, carried over portages and around rapids, and then re-loaded onto barges further along the route. Inevitably, there was some spillage of material at the trans-shipment points and some local contamination by radionuclides of the uranium decay series. A full remediation of these sites is recommended in the CDUT report.

Many indigenous Canadians were employed to load and unload the barges and carry bags of uranium ore. In 1998 concerns arose within the Déline community that many of the ore carriers may have suffered cancers as a result of radiation exposure. As part of the CDUT Action Plan, a dose reconstruction project was undertaken for the ore carriers (SENES 2005). This reconstruction was based on historical records of the ore grade transported, oral recollections of ore handling practices, and number of years worked as ore carriers. The results for 35 individuals, for whom relatively complete work histories were available, are shown in Figure 3·3. The cumulative radiation doses during the period of employment varied from 27 to 3015 mSv.

### 3.3.2.1.3. Rayrock Mine

The Rayrock Mine site is located 145 km northwest of Yellowknife, Northwest Territories. Uranium was discovered there in 1948, although mining operations were not carried out until 1957 to 1959. During operations at Rayrock Mine, approximately 70 000 tonnes of ore were processed, yielding 207 tonnes of uranium concentrate. Radioactive mine tailings were deposited on land and partly flowed into three small lakes. Measurements taken in 1985 showed mildly elevated levels of uranium, $^{226}$Ra, and $^{210}$Pb in the small lakes, although none of the activity concentrations exceeded Canadian Drinking Water Guidelines (Hatfield Consultants, 1985).

The mine was also a potential source of radon gas emissions from mine openings and ventilation shafts.

The site was remediated in 1996 and 1997 following several site assessments. This work included sealing all mine openings and ventilation shafts, relocating radioactive material from the dump to the tailings piles and capping the tailings with a thick layer of silt-clay, followed by revegetation. The site is undergoing long-term monitoring and is being monitored every year until 2009; it will then be monitored once every 10 years for a further 100 years. Monitoring assesses the integrity of sealed mine openings, water quality and potential risks to humans (INAC, 2008). To date, long-term monitoring results have shown that: fish in the area are safe to eat; caribou sampled in the area are within the normal range of radionuclide concentrations for the Northwest Territories, very little risk remains to humans from radionuclide exposure; water quality in the lakes meets drinking water standards and is gradually improving over time; downstream water quality is not affected by the former mine, and seals blocking the former mine openings are in good condition.

### 3.3.2.2. Finland

Uranium exploration took place in Finland until the 1990s. Small-scale mining operations were also carried out in Eno, eastern Finland, and Askola, southern Finland. Following the resurgence in global demand for uranium, exploration started again in Finland in 2004. International uranium-prospecting companies have begun to show interest in Finland, especially in the Uusimaa, Northern Karelia and Lapland provinces. Exploration is now underway in many parts of Finland. Mining will only begin if the uranium deposits and concentrations prove adequate in size and grade. Uranium mining is subject to a concession or licence granted by the Finnish government. It is a requirement of Finnish nuclear energy law that environmental impact assessments must be undertaken before mining activities can begin.

### 3.3.3. Mining for metals other than uranium

Some minerals and ores found in nature are enriched in certain elements due to natural processes. This makes such deposits commercially attractive and they are exploited. A number of these minerals are also enriched in $^{238}$U and $^{232}$Th, which will thus enter the subsequent processing industries. Mining, extraction and processing of these ores may therefore lead to elevated radiation exposures to workers and the generation of waste containing natural radioactivity. Depending on the processing method, the $^{238}$U and $^{232}$Th series radionuclides are often separated and end up in waste or by-products.

Waste generated during mining and processing includes overburden, tailings, mine waters, scale and slag. Solid mining waste exposed to air and rainwater may, if sulfide minerals are present, generate sulfuric acid that can form soluble complexes with uranyl ions which increase leaching from the waste and could lead to groundwater contamination.

Metals are often found in compounds with oxygen or sulfur and must be extracted from the ore. Current methods involve grinding, pre-concentration and subsequent heat treatment with a reducing agent such as coal or coke.

**Table 3·5.** Typical activity concentrations of naturally-occurring radionuclides in different ore types. Source: Martin et al. (1997).

| Ore type | $^{238}$U, Bq/kg | $^{232}$Th, Bq/kg |
|---|---|---|
| Iron ore | < 5 | < 5 |
| Coal/coke | 20 | 20 |
| Tin ore | 1 000 | 300 |
| Pyrochlore (niobium) | 10 000 | 80 000 |
| Ilmentie (titanium) | 1 000 | 1 000 |
| Rutile (titanium) | 350 | 200 |
| Bauxite (aluminum) | 350 | 300 |

**Table 3·6.** Typical activity concentrations of $^{226}$Ra and $^{232}$Th in phosphate ore from different deposits. Source: Karhunen and Vermeulen (2000), UNSCEAR (1982).

| Location/deposit | $^{226}$Ra, Bq/kg | $^{232}$Th, Bq/kg | Origin |
|---|---|---|---|
| Finland/Siilinjärvi | 10 | 25 | Igneous |
| Canada/Sept-Iles | 30 | 30 | Igneous |
| Russia/Kovdor | 30 | 65 | Igneous |
| Russia/Khibiny | 40 | 100 | Igneous |
| Finland/Sokli | 250 | 150 | Igneous |
| Syria/Eastern | 600 | < 10 | Sedimentary |
| Morocco/K-11 | 1 300 | < 20 | Sedimentary |
| Morocco/K-22 | 1 440 | 10 | Sedimentary |
| USA/Central Florida | 1 500 | 30 | Sedimentary |
| USA/Idaho | 1 800 | 30 | Sedimentary |

Heat treatment of the ore may lead to airborne discharges of volatile radionuclides such as $^{210}$Po and $^{210}$Pb. Typical levels of $^{238}$U and $^{232}$Th in different ore types are listed in Table 3·5. Local and regional differences in geology may result in large variations in activity concentrations of $^{238}$U and $^{232}$Th in ores (Egidi, 1997).

### 3.3.4. Phosphate mining and processing

Phosphates have several industrial applications, the most important being as fertilizers and food supplements in agriculture. Phosphate ore is recovered from both open pits and underground mines. Sedimentary phosphate ore (for example from Florida, Morocco and Syria) is known to have elevated levels of $^{238}$U (generally in secular equilibrium with decay products), normally in the range 1000 Bq/kg to 1500 Bq/kg. Phosphate ore of igneous origin, for example Kovdor apatite from the Kola Peninsula, usually has much lower activity concentrations of $^{238}$U (< 100 Bq/kg). Typical activity concentrations of $^{226}$Ra and $^{232}$Th in phosphate from some deposits are listed in Table 3.6.

After the ore has been recovered from the mine it is normally sent for beneficiation and grinding, and is then processed either by the thermal process or more commonly, by the wet process. Waste material from mining and beneficiation includes waste rock, clay slime and sand tailings. Major airborne emissions normally occur from dust generation during grinding and open-pit mining in dry and windy climates.

The thermal process is used for producing elemental phosphorus, which is later converted to phosphoric acid of high purity. The more common wet process is used for producing phosphoric acid, which is later used for fertilizer production or animal food supplements. To produce phosphoric acid, the phosphate ore is dissolved with sulfuric acid. In this process large amounts of phosphogypsum are produced, which constitutes the major waste problem in the phosphate

industry. During phosphoric acid production, 90% to 100% of the $^{226}$Ra is normally incorporated in the phosphogypsum, together with a large fraction of $^{210}$Pb and $^{210}$Po, while most of the $^{238}$U and $^{234}$U is found in the phosphoric acid. The phosphogypsum is either stored in piles or has in some cases been discharged to sea. Concerns about land disposal of phosphogypsum include radon emanation and leaching of radionuclides and the subsequent contamination of groundwater.

#### 3.3.4.1. Phosphate mining in Finland

Exploration activities and an environmental impact assessment related to the possible opening of a phosphate mine have been carried out in the Sokli region of the Savukoski municipality.

### 3.3.5. Coal mining and energy production from coal

The main radiological impact on the public and the environment arises when coal is burnt and converted to ash, but enhanced levels of natural radioactivity can also be encountered in connection with coal mining. Chalupnik et al. (2001) found elevated levels of radium in mine waters (which in part end up in surface waters) and in the formation of radium-bearing scale in coal mines in Poland. Dowdall et al. (2004) found small but significantly enhanced radiation levels in former coal mining areas of Svalbard.

Combustion of coal for energy production results in two types of ash: bottom ash, comprising coarse particles that settle at the bottom of the furnace, and fly ash, finer particles that are carried with the hot flue gases. Fly ash normally constitutes about 80% of the residues from coal combustion. The amount of fly ash released to the atmosphere depends on the efficiency of the flue gas filtering systems. Older coal-fired plants release about 10% of the fly ash to the atmosphere, while more modern plants equipped with electrostatic precipitators or bag filters, discharge only about 0.5% of the fly ash (UNSCEAR, 1988). Activity concentrations in coal average about 20 Bq/kg for $^{238}$U and $^{232}$Th and 50 Bq/kg for $^{40}$K. After combustion (up to 1700 °C), the non-combustible elements such as $^{238}$U, $^{226}$Ra, $^{232}$Th, $^{210}$Pb and $^{210}$Po are enriched in the fly ash. Average activity concentrations in fly ash reported by UNSCEAR (1988) were 200 Bq/kg for $^{238}$U, 240 Bq/kg for $^{226}$Ra, 930 Bq/kg for $^{210}$Pb, 1700 Bq/kg for $^{210}$Po and 70 Bq/kg for $^{40}$K.

The radiological hazard from these airborne emissions has been estimated based on a number of studies that took place during the 1970s and 1980s. The general conclusion is that the dose contribution from airborne emissions to a person living 1 km from a coal combustion plant is low (1% to 5% above the normal background dose from natural radiation). For the average citizen the contribution is less. Fly ash can also be added to concrete and so constitutes a minor additional source for radon (USGS, 1997). Coal ash disposed of in piles and landfill can also be a minor source of natural radionuclides in groundwater due to leaching.

#### 3.3.5.1. Coal mining at Svalbard

Coal mining at Svalbard began in the early 1900s with mines in and around Longyearbyen and at Sveagruva at the head

of Van Mijengjord. Russian mining operations are centered at Barentsburg with some activity at Pyramiden in Billeford. Ny-Ålesund is the world's most northern settlement and is an area of special environmental interest. Coal mining at Ny-Ålesund started in 1916 and activities continued until a big accident in 1962. The remains of the operations at Ny-Ålesund take the form of old machinery, equipment and piles of waste materials from the mines. Samples obtained in areas of Ny-Ålesund contaminated with coal waste, indicate enhanced activity concentrations of $^{40}$K and the $^{238}$U and $^{232}$Th series (Dowdall et al., 2004).

## 3.3.6. Geothermal energy production

Geothermal energy is only a minor source of radiological impact in most countries. An exception is Iceland, where almost 80% of households are heated by geothermal energy. In the production of geothermal energy, hot water or steam from deep boreholes is pumped to the surface. Due to minerals and naturally-occurring radionuclides present in the water, radium-containing scale may be formed inside pipes, production equipment and ponds. Radon may be transported from the ground by water and steam and then released to the atmosphere, leading to locally elevated levels of radon.

*Chapter 4*

# Monitoring

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 may be used to detect altered discharges from known sources of contamination or to identify new sources. Monitoring is also important in helping to understand the pathways for radionuclide transport to, within and from the Arctic. Monitoring data are used to calculate the effective ecological half-lives of particular radionuclides and so provide an understanding of the long-term effects of radionuclide contamination in different environments and food webs. Radionuclides in food webs can be taken up directly from the air or sea, or indirectly through root systems and in which case are dependent on factors such as soil type, root depth, and competition with stable elements. This leads to different levels of radionuclides in different species. The geographical distribution of contamination and differences in animal diets and metabolism also result in a range of activity concentrations at different tropic levels. This chapter presents monitoring data for the atmosphere, the marine environment and marine species, the terrestrial and freshwater environment and species, and humans.

## 4.1. Radionuclides in the atmospheric environment

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 (Figure 4·1). The stations are of two types: air filter stations and/or doserate monitoring stations.
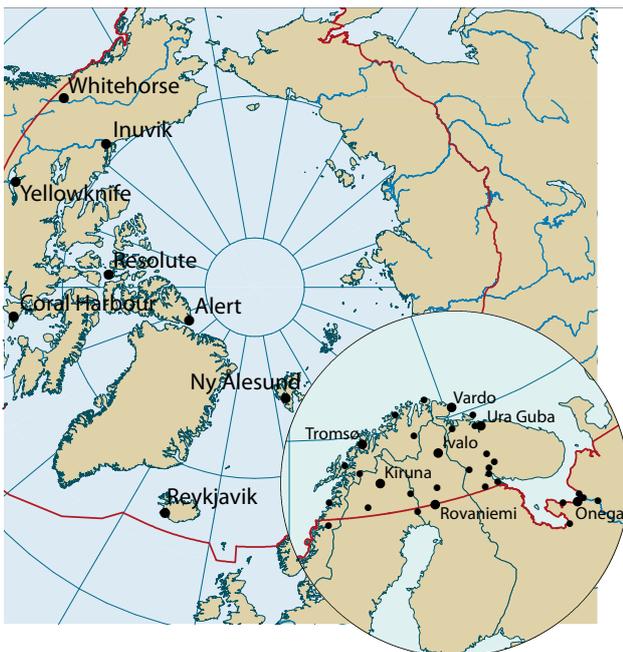


**Figure 4·1.** Atmospheric monitoring stations within the AMAP region.

### 4.1.1. Alaska, USA

There are seven fixed-site radiation monitoring stations in Alaska with one located within the Barrow area. These stations are currently under review for replacement or removal. The US government also conducts regular radiation monitoring at several locations in the Aleutian Island chain, including Amchitka.

### 4.1.2. Canada

The Radiation Protection Bureau of Health Canada operates a network of radiological air monitoring stations across Canada, including nine sites in the AMAP region. Each site is equipped with a high-volume air sampler, a precipitation collector, and a thermo-luminescent dosimeter for gamma radiation measurements. In addition, the Yellowknife site has equipment for the measurement of radioxenon.

The network is operated primarily for detecting routine emissions from nuclear facilities and for providing early warnings of major nuclear events that may impact on Canada. The stations provide daily and weekly measurements of artificial and natural radionuclides in air at all locations. These measurements contribute to a wide range of scientific studies on atmospheric transport processes and climate change effects.

#### 4.1.2.1. Artificial radionuclides

Traces of $^{137}$Cs, resulting from the atmospheric testing of nuclear weapons in the 1950s and 1960s, are still occasionally detected on air filters from northern Canada. Figure 4·2
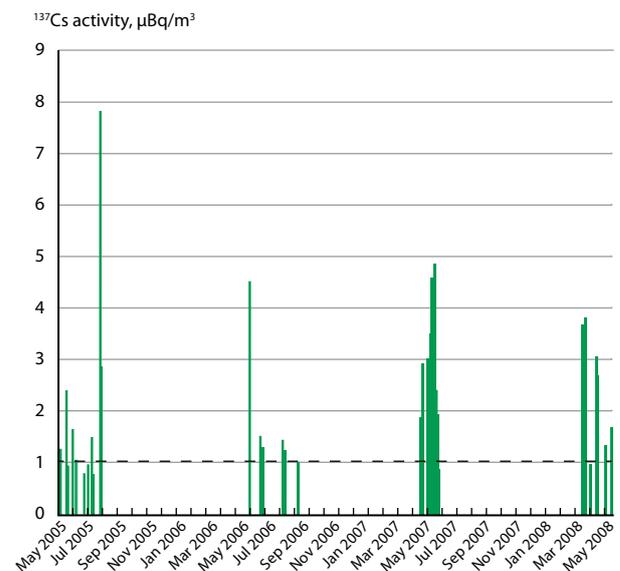


**Figure 4·2.** Events of $^{137}$Cs detection between 2005 and 2008 in 24-hour air filter samples from the monitoring site at Yellowknife, NWT. An event is defined as a concentration of $^{137}$Cs in air exceeding 1 µBq/m³.

shows the number of detections over a three-year period in daily air samples from Yellowknife, Northwest Territories (Tracy, B. and S. Swenson, Radiation Protection Bureau, Health Canada, pers. comm., 2008). Events were recorded whenever the $^{137}$Cs activity concentration exceeded the detection threshold of 1 μBq/m³. It is notable that all the detections occurred during the summer period, between May and August, when forest fires are most prevalent in boreal forests. This indicates that the $^{137}$Cs probably becomes airborne either from the re-suspension of litter from the forest floor or from the combustion of older trees that have retained significant amounts of radiocesium from the period of atmospheric testing of nuclear weapons. Wotwa et al. (2006) established a correlation between $^{137}$Cs observations at the Yellowknife station and the number of boreal forest fires determined from satellite imagery.

Between 22 and 29 October 2006, traces of $^{133}$Xe were detected at the noble gas monitoring station in Yellowknife (Saey et al., 2007). Back trajectory calculations showed that the probable source of this radioxenon was venting from an underground nuclear weapons test conducted by the Democratic Republic of North Korea on 9 October 2006. Although the amount of detected $^{133}$Xe was trivial (< 1 μBq/m³), its detection a few weeks later at a point thousands of kilometers from the test site demonstrates the vulnerability of the Arctic environment to events occurring in remote parts of the world.

### 4.1.2.2. Natural radionuclides

Airborne $^{210}$Pb results primarily from the decay of radon gas emanating from soils in continental land masses. 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 of time. Figure 4.3a summarizes the measurements of $^{210}$Pb activity concentrations in air at two locations in northern Canada (Whitehorse, Resolute) over a seven-year period. There is remarkable consistency in the annual pattern of $^{210}$Pb activity concentrations from site-to-site and from year-to-year. Concentrations rise to a maximum during mid-winter. This is when the long polar night leads to stable atmospheric conditions which can trap the $^{210}$Pb in an air layer close to the ground. Any disruption to this long-term pattern could be an indicator of input from anthropogenic sources, such as uranium mines. Alterations to the pattern could also be a barometer for changing atmospheric conditions due to global warming.

Beryllium-7 is produced by the bombardment of air molecules in the upper atmosphere by cosmic rays of galactic origin. It is brought to the surface by atmospheric mixing processes and precipitation. With a half-life of 53 days, $^{7}$Be is routinely measured on all air filters from the Canadian radiological monitoring network. Figure 4.3b summarizes the measurements of $^{7}$Be activity in air at the same two northern stations (Whitehorse, Resolute) from 1996 to 2008. As for
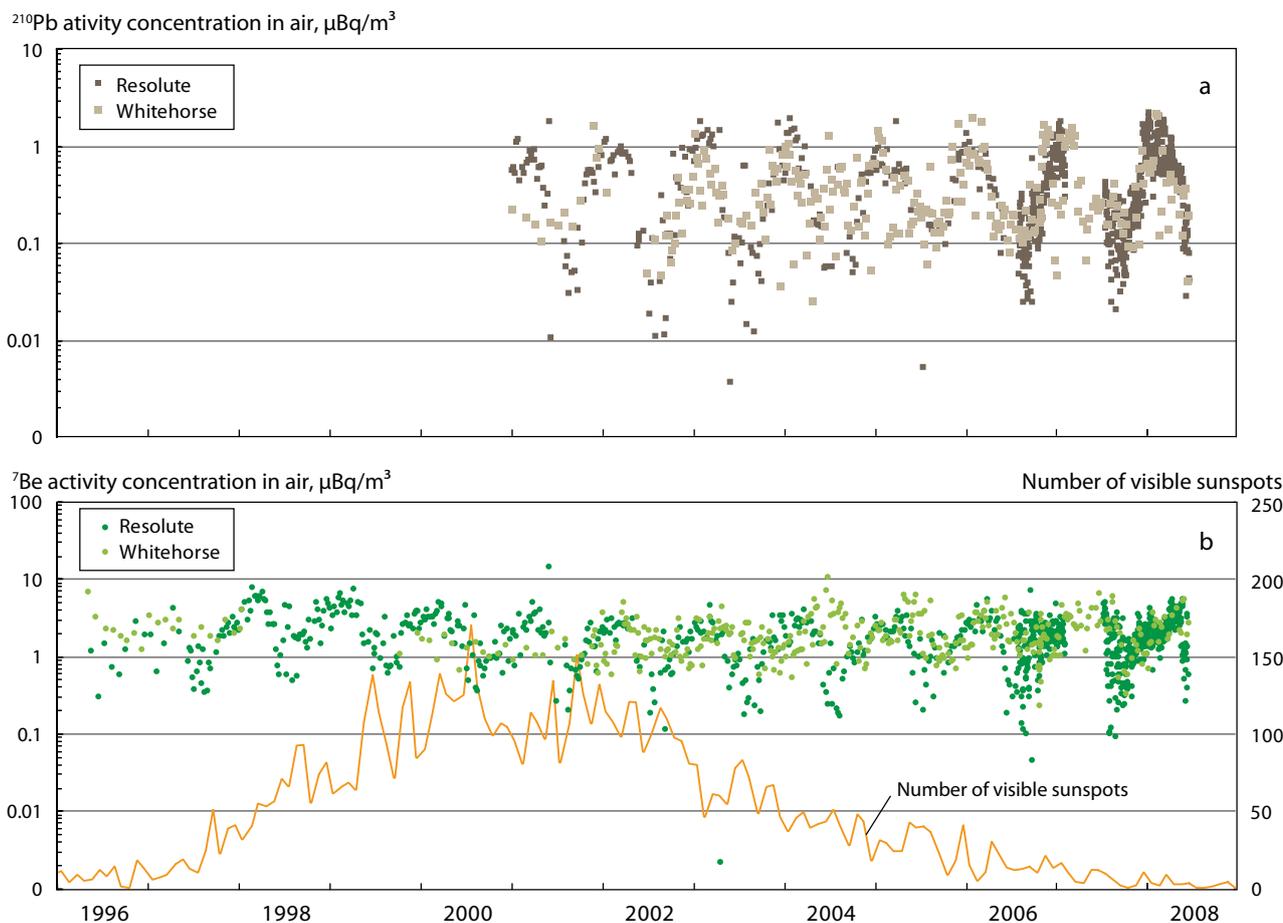


**Figure 4.3.** Activity concentrations of $^{210}$Pb and $^{7}$Be in air at Whitehorse and Resolute, northern Canada.

**Figure 4·4.** Location of the three aerosol samplers in northern Norway.
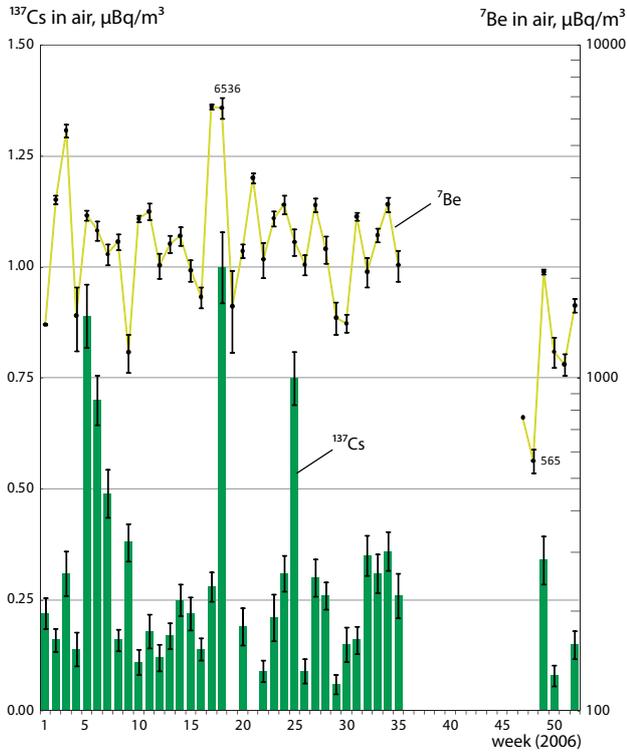


**Figure 4·5.** Weekly concentrations of $^{137}$Cs and $^7$Be in air at Svanhovd, northern Norway, in 2006. Source: Møller and Drefvelin (2008).

$^{210}$Pb, there is remarkable consistency in $^7$Be activity concentrations from site-to-site and year-to-year. However, $^7$Be concentrations tend to peak in the late spring, when air exchanges between the stratosphere and troposphere are at a maximum. Superimposed on the annual $^7$Be variations, are the effects of the 11-year sun spot cycle. The figure shows that the $^7$Be activity concentrations were lower during the sun spot maximum in 2002 and higher during the sun spot minima in 1997 and 2008. During high solar activity, ionized particles from the sun become trapped in the earth's magnetosphere and essentially act as a shield against the high energy cosmic rays that produce $^7$Be.

### 4.1.3. Norway

The Norwegian Radiation Protection Authority (NRPA) has ten atmospheric monitoring stations in the AMAP region, and data are automatically transferred to its head office near Oslo once every 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 atmospheric monitoring stations, the NRPA has three aerosol samplers in the AMAP region (see Figure 4·4).

Figure 4·5 shows the weekly activity concentrations of $^{137}$Cs and $^7$Be at Svanhovd in 2006 (Møller and Drefvelin, 2008). The values for $^{137}$Cs peak each year in late spring. The origin of this $^{137}$Cs is atmospheric fallout from nuclear weapons testing program and the Chernobyl accident, and the peaks in activity are mostly related to the resuspension of Chernobyl contamination from soil and dust or forest fires. Levels of $^7$Be activity also peak in late spring and occur when air exchange between the stratosphere and troposphere is at a maximum.
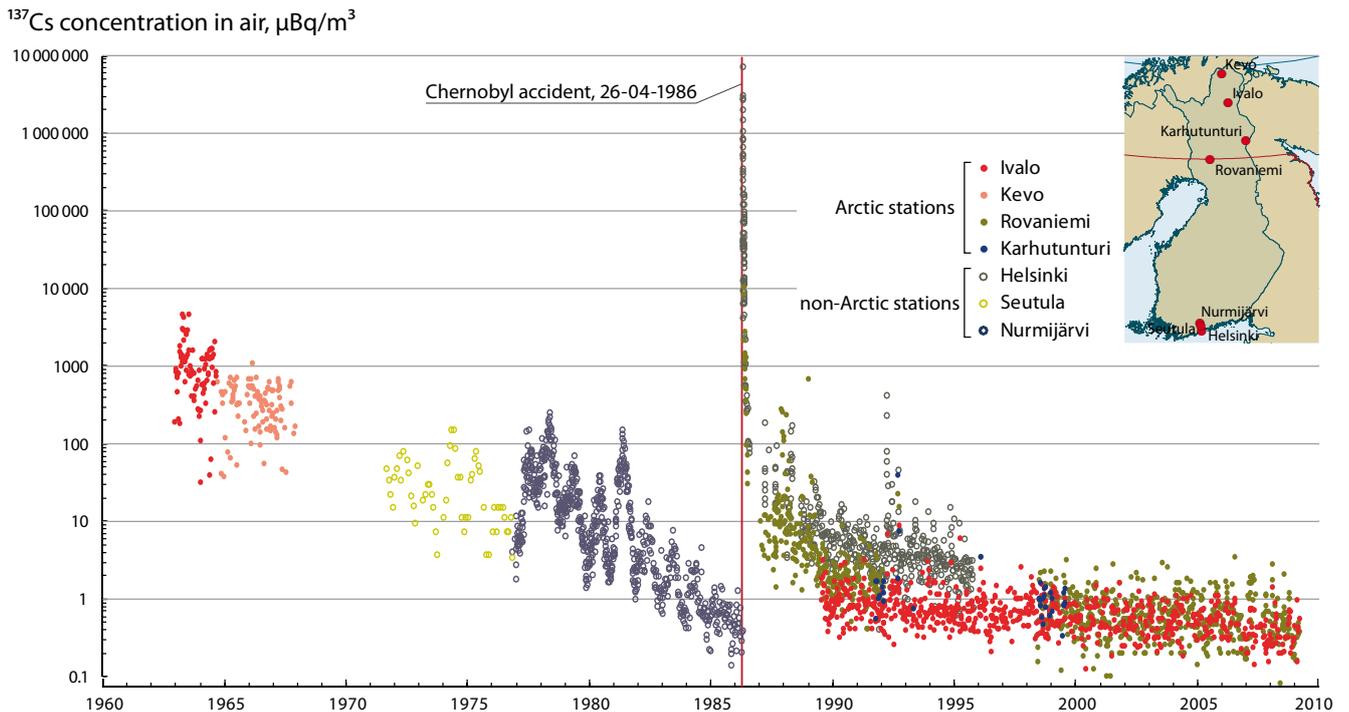


**Figure 4·6.** Air concentration of $^{137}$Cs at Finnish stations. Data: STUK and FMI.

**Table 4.1.** Mean volume activity concentrations of radionuclides in the surface atmospheric layer and total atmospheric deposition, averaged across the Russian Arctic.

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| Volume activity | | | | | | | |
| Total β-activity ($10^{-5}$Bq/m³) | 12.3 | 11.9 | 10.2 | 8.4 | 8.4 | 12.7 | 7.7 |
| $^{90}$Sr ($10^{-7}$Bq/m³) | 0.4 | 0.7 | 0.4 | 0.6 | 0.4 | 0.36 | 0.27 |
| $^{137}$Cs ($10^{-7}$Bq/m³) | 1.7 | 1.9 | 2.1 | 0.9 | 1.0 | 1.6 | 1.0 |
| Deposition | | | | | | | |
| $^{137}$Cs (Bq/m²/y) | 0.4 | 0.7 | 0.6 | 0.5 | < 0.4 | < 0.4 | < 0.4 |

$^{137}$Cs in surface air, µBq/m³
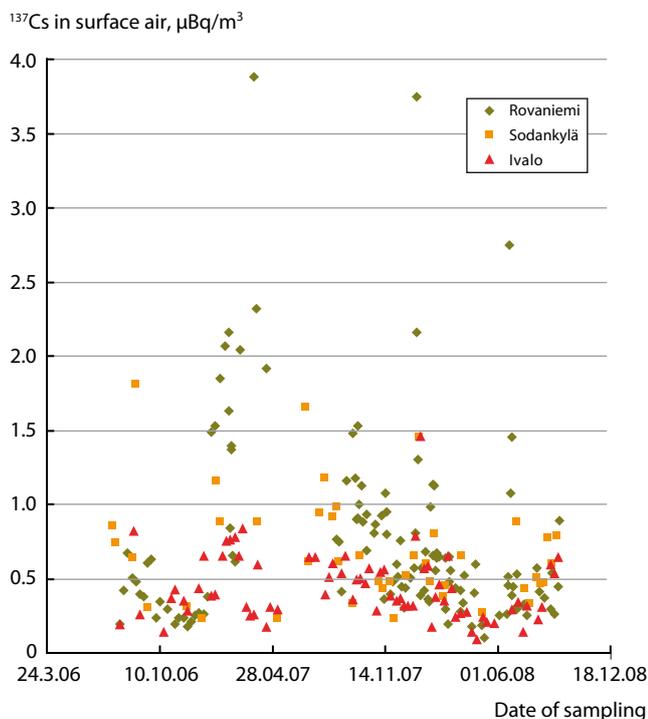


**Figure 4.7.** Seasonal variations of $^{137}$Cs in surface air at different sites in Finland over a two-year period.

### 4.1.4. Finland

The three aerosol samplers in northern Finland are located at Rovaniemi, Sodankylä and Ivalo. These are part of the national radioactivity monitoring network. 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 (Figure 4.6), which resulted in a sudden and substantial increase in $^{137}$Cs levels in the atmosphere. These high levels declined over the following decade and have since been fairly stable. The fallout (wet and dry deposition) of $^{137}$Cs shows seasonal variations with levels in surface air mostly within the range 0.1 µBq/m³ to 1.5 µBq/m³ with occasional peaks of up to 4 µBq/m³ (Figure 4.7). Observations of $^7$Be activity at the Ivalo aerosol sampler from 1989 to 2007 show occasional peaks of up to 8500 µBq/m³. The baseline behaviour of $^7$Be activity follows the 11-year sun spot cycle (Figure 4.8).

### 4.1.5. Russia

In Russia, the 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 radiation monitoring involves daily monitoring of gamma-radiation dose rate, volumetric activities in the surface atmospheric layer, and deposition of radioactive substances from the atmosphere onto the underlying surface. Table 4.1 presents

$^7$Be in air, µBq/m³      number of sunspots
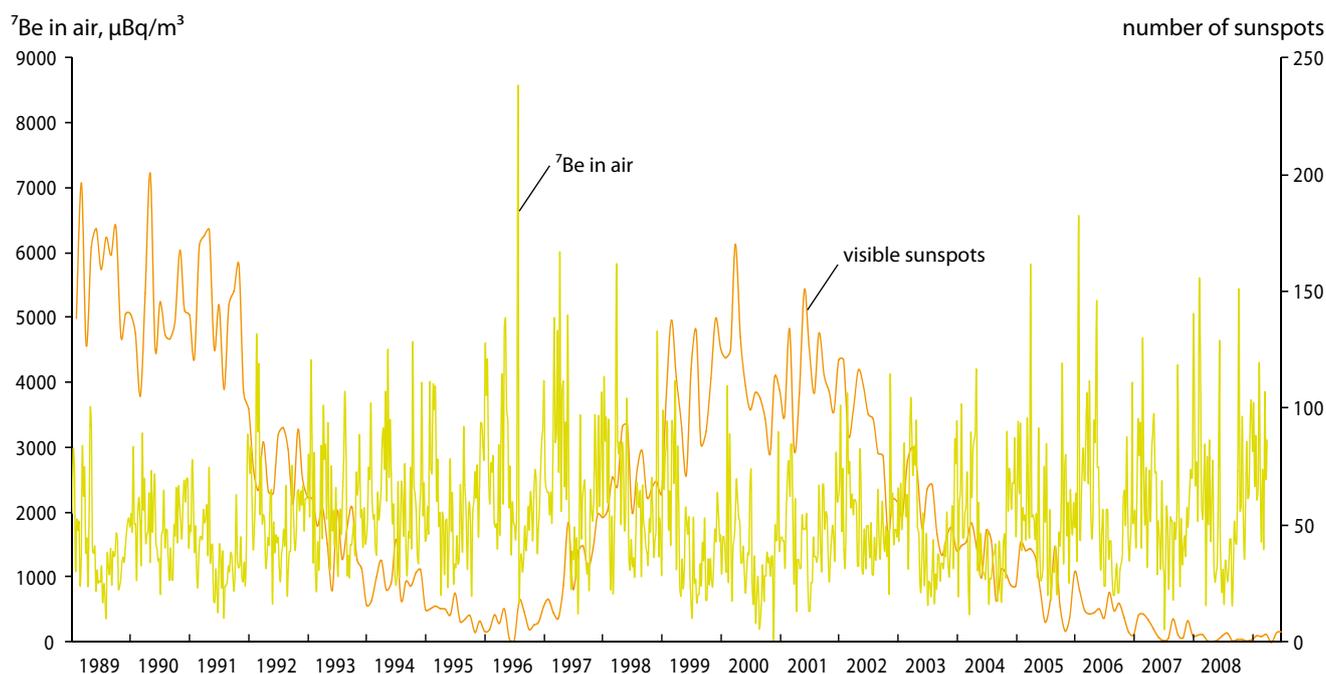


**Figure 4.8.** Observed $^7$Be activity at Ivalo, Finland, since 1989. The baseline behaviour reflects the 11-year sun spot cycle.

data on mean annual volume activities in the surface atmospheric layer and total atmospheric deposition averaged over the Russian Arctic regions. The total β-activity in the surface atmospheric layer shows a slow, decreasing trend from 2000 to 2006. For $^{137}$Cs and $^{90}$Sr there is no apparent trend, but the values are low.

## 4.2. Radionuclides in the marine environment

Present and potential radioactive contamination in the marine environment has received much attention in recent years. In the late 1980s, several incidents involving nuclear powered submarines demonstrated that the risk of release of radionuclides into the Barents Sea should be considered more carefully. In particular, it became clear that better documentation concerning radioactivity levels in fish and other seafood was important for the seafood export industries. Also, in the early 1990s, information about the dumping of nuclear waste in Arctic waters emerged and in the years that followed, concern grew regarding the safety of military and civil nuclear installations in northwest Russia. This concern was associated not only with possible reactor accidents, but also with the prolonged or sudden release of radionuclides from radioactive waste facilities and past dumping of wastes on the seabed.

In addition to threats arising at the local level, radionuclides originating from nuclear weapons fallout, the Chernobyl accident, and waste discharged from European reprocessing facilities have been detected in the marine environment. Discharges of $^{99}$Tc from the reprocessing facility at Sellafield in the UK peaked in 1995. In 1994, new technology became operational and began to treat a backlog of wastes stored at the site. This was not designed to extract $^{99}$Tc and this explains the peak in $^{99}$Tc discharges in 1995 and the high level of discharges in subsequent years. In 2004, the $^{99}$Tc dis-

charged dropped due to the implementation of new technology that was designed to extract the $^{99}$Tc (see section 2.1.3.1).

Industrial activities, such as mining and oil production, may change the distribution of naturally-occurring radionuclides in the marine environment. The discharge of radium from water produced by oil installations is one issue that has received special attention (see section 3.3.1).

### 4.2.1. $^{129}$I transport from Western Europe to North American coastal waters

Iodine-129 is a long-lived (half life = 16 million years) radionuclide that has been released to the ocean in large quantities from nuclear fuel reprocessing facilities. As a result of its long half life and conservative behavior in seawater it serves as an excellent tracer of long-term oceanic circulation patterns. Advances in the technique of accelerator mass spectrometry mean that $^{129}$I can now be measured in one liter water samples virtually anywhere in the global ocean.

Between 1990 and 2000, annual discharges of $^{129}$I into the North Atlantic from Sellafield (UK) and La Hague (France) increased by 600%. A study by Smith et al. (2005) showed that the leading edge of this increase in the $^{129}$I signal has now entered the Labrador Sea (bounded by Labrador, Baffin Island, and Greenland), with transit times of about seven and nine years from La Hague and Sellafield, respectively. Activity concentrations of $^{129}$I in the Labrador Sea have increased by 300% relative to the previous background values. Tracer studies have shown that the $^{129}$I circulates in the North Sea and flows northward along the Norwegian coast to Svalbard. From there it flows into the Arctic Ocean through Fram Strait and the Barents Sea and also circulates cyclonically (counter clockwise) around the Greenland Sea, eventually passing southward along the east coast of Greenland and entering the deep Labrador Sea at depths of 3000 m with Denmark Strait Overflow Water. Figure 4.9 provides an overview of the main surface current systems in the North
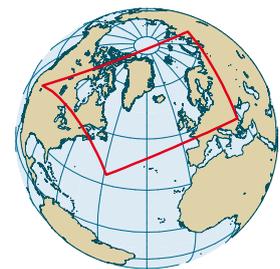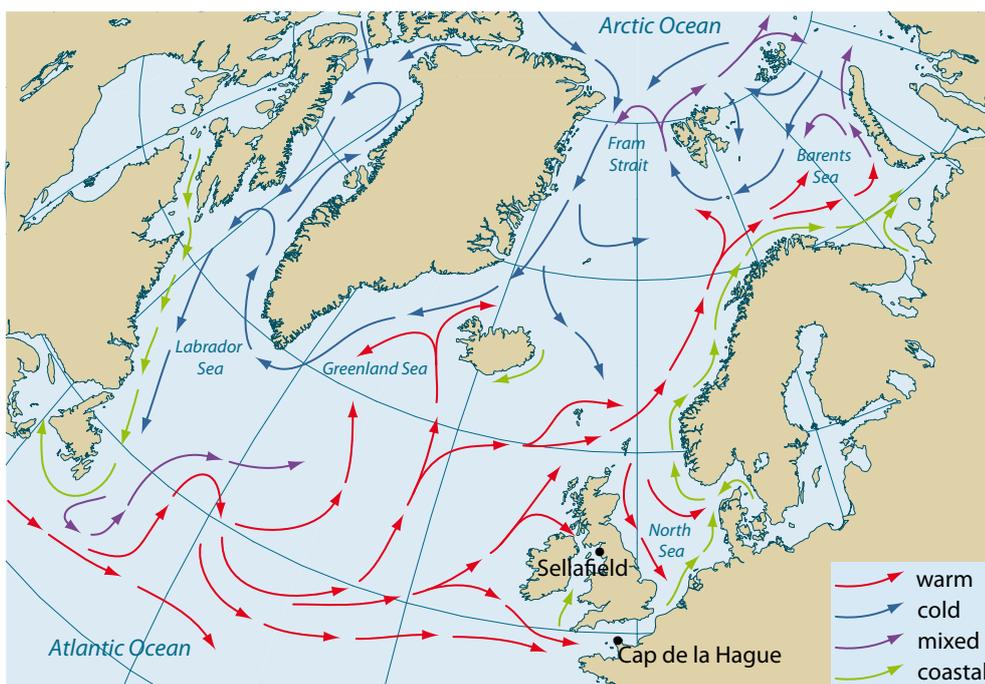


**Figure 4.9.** Overview of the main current system in the North Sea, Norwegian Sea, Greenland Sea and Barents Sea. Source: Aure et al. (1998).

Sea, the Norwegian Sea, the Greenland Sea and the Barents Sea.

Although the activity concentration of $^{129}$I in seawater is too low to be of any health or environmental concern, it can serve as an indicator of any changes in long-term oceanic circulation patterns. Furthermore, the detection of $^{129}$I of European origin in the deep North Atlantic shows that radionuclide and other contaminant releases into the oceans can quickly become globally distributed.

### 4.2.2. Seawater

Activity concentrations for anthropogenic radionuclides in seawater are available for a variety of locations covering the west coast of Greenland, Denmark Strait and the Faroe Islands. Levels of $^{90}$Sr and $^{137}$Cs show little variation across the region with activity concentrations in the range 0.9 Bq/m³ to 1.3 Bq/m³ and 2 Bq/m³ to 3 Bq/m³, respectively. Concentrations of $^{90}$Sr near Disko (Greenland) are as low as 0.2 Bq/m³ due to dilution from the melting of uncontaminated ice, whereas low concentrations of $^{137}$Cs of 1.5 Bq/m³ to 2.0 Bq/m³ are found off the Faroe Islands. Concentrations of transuranics in seawater are low and show little variability along the west coast of Greenland. Activity concentrations of Pu isotopes occur in the range 2 mBq/m³ to 6 mBq/m³ for $^{239,240}$Pu, < 0.3 mBq/m³ for $^{238}$Pu and < 1 mBq/m³ for $^{241}$Am.



**Figure 4.10.** Activity concentrations in seawater from East and West Greenland and the Faroe Islands since 1960, for (a) $^{90}$Sr and (b) $^{137}$Cs.
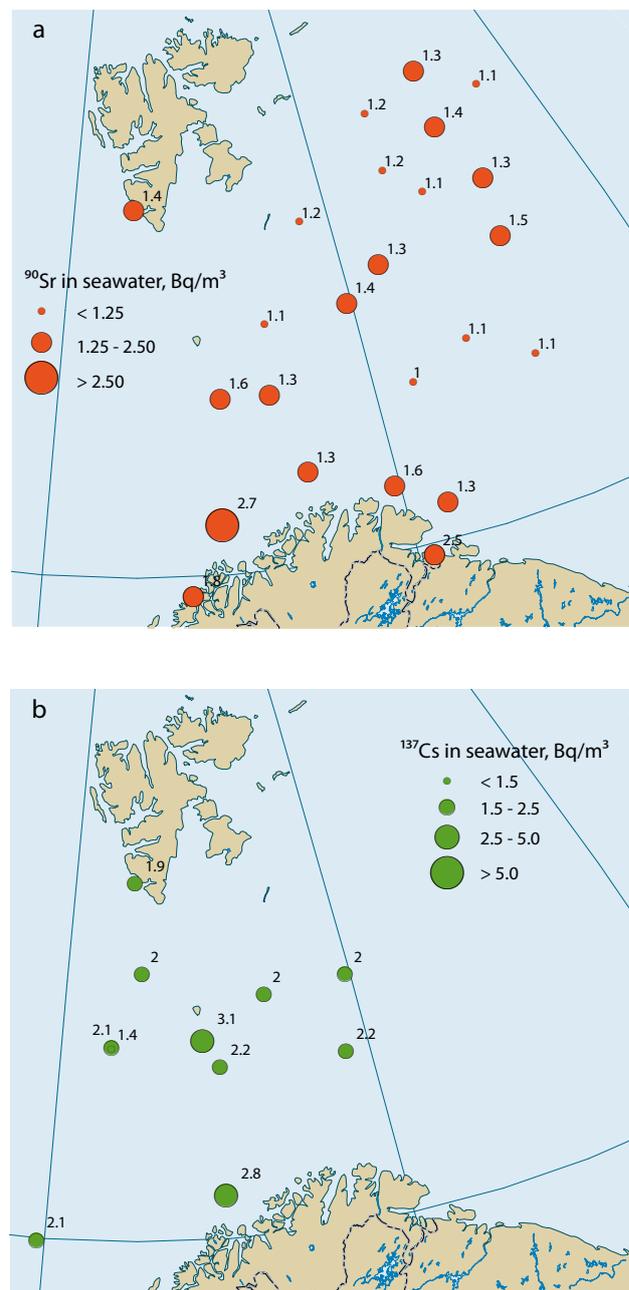
**Figure 4.11.** Activity concentrations in surface seawater from the Barents Sea in 2005, for (a) $^{90}$Sr and (b) $^{137}$Cs.

**Table 4.2.** Volume activity concentrations of $^{90}$Sr in the water (mBq/L) of the White Sea and Barents Sea. Source: Hydrometeoizdat (2001, 2002, 2003, 2004); Meteoagency of Roshydromet (2005, 2006); Nizhny Novgorod, Vektor-TiS (2007).

|             | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|-------------|------|------|------|------|------|------|------|
| White Sea   | 4.0  | 4.1  | 3.5  | 3.4  | 3.8  | 3.4  | 3.6  |
| Barents Sea | 3.4  | 3.4  | 3.1  | 3.6  | 2.8  | 2.0  | 2.3  |

**Table 4·3.** Range in activity concentrations for artificial radionuclides in seawater samples from a Russian coastal region and the open Barents Sea in September 2006.

| | $^{137}Cs$, Bq/m³ | | $^{90}Sr$, Bq/m³ | $^{239,240}Pu$, mBq/m³ | | $^3H$, Bq/m³ |
|---|---|---|---|---|---|---|
| | Suspended matter | Filtrate | Filtrate | Suspended matter | Filtrate | Filtrate |
| Coastal area near the Teriberka settlement (12/13 Sept.) | < 0.01 – < 0.06 | 2.3 – 2.7 | 1.2 – 1.6 | 6.1 | 7.2 | 490 |
| Open Barents Sea (29/30 Sept.) | < 0.04 – < 0.06 | 2.0 – 2.8 | 1.2 – 3.0 | 7.6 | 7.7 – 9.1 | 310 – 440 |
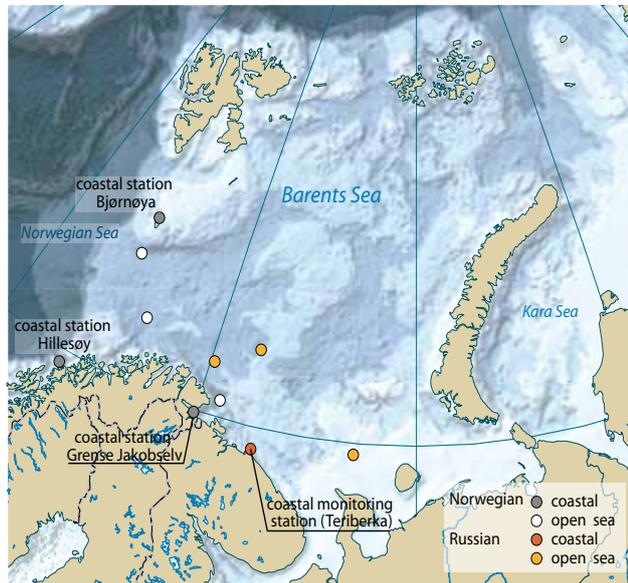


**Figure 4·12.** Location of some coastal and offshore monitoring stations in the Arctic.



**Figure 4·13.** Activity concentrations of $^{99}Tc$ in seawater samples from the Barents Sea in 2005.

Activity concentrations of $^{90}Sr$ and $^{137}Cs$ in the North Atlantic between 1960 and 2006 are shown in Figure 4·10. The data illustrate that activity concentrations in seawater are higher off the east coast of Greenland than the west coast of Greenland and that concentrations are generally lower off the Faroe Islands.

Data on $^{90}Sr$ activity concentrations in the water of the White and Barents seas are shown in Table 4.2. Activities are lower in the Barents Sea than the White Sea, and lower in the coastal area near Teriberka than the open Barents Sea (see Table 4.3). $^{90}Sr$ and $^{137}Cs$ concentrations in the Barents Sea for 2005 are shown in Figure 4·11 (NRPA, 2007a). In contrast, Figure 4·11 shows activity concentrations of $^{90}Sr$ and $^{137}Cs$ in seawater from the Barents Sea in 2005 to be highest near the coast, although levels are all low. This indicates that that contamination is transported with the Norwegian coastal current.

Some Russian monitoring data (Table 4.3 and Figure 4·12) show similar, low values for the coastal area and the open Barents Sea.

### 4.2.2.1. $^{99}Tc$ in seawater

In oxygenated seawater, $^{99}Tc$ is present as the highly soluble pertechnetate ion (TcO$_4^-$). Due to its conservative behavior in seawater, TcO$_4^-$ is able to be transported by ocean currents without being significantly affected by sedimentation processes. From the Irish Sea, into which the $^{99}Tc$ is discharged from the reprocessing facility at Sellafield, $^{99}Tc$ is transported by ocean currents to the North Sea and via the Norwegian



**Figure 4·14.** Annual average activity concentrations of $^{99}Tc$ in seawater at Hillesøy, northern Norway, and annual discharges of $^{99}Tc$ from the Sellafield nuclear fuel reprocessing plant, UK, since the early 1990s.

Coastal Current to the Barents Sea. The transit time (the time between a specific discharge and the maximum activity concentration from that discharge reaching the sampling location) for $^{99}Tc$ to reach the Norwegian coastal station at Hillesøy (see Figure 4·12) from the Irish Sea has been estimated at three to four years (Brown et al., 2002; Dahlgaard, 1995). Activity concentrations for $^{99}Tc$ in seawater in the Barents Sea ranged from 0.1 to 0.7 Bq/m³ in 2005 (Figure 4·13).

Monthly samples have also been collected at Hillesøy on the northern Norwegian coast, and annual average activity concentrations of $^{99}Tc$ in seawater at Hillesøy together with

**Table 4.4.** Activity concentrations for radionuclides in seaweed and fish sampled in the vicinity of the Teriberka settlement in Russia in 2006.

| | [137]Cs, Bq/kg | [90]Sr, Bq/kg | [239,240]Pu, mBq/kg | [241]Am, mBq/kg |
|---|---|---|---|---|
| Seaweeds [a]: *Fucus vesiculosus, F. distichus, Laminaria hyperboreja, L. sacharina, Cladophora* | < 0.09 − 1.05 | 0.2 − 0.3 | 32 − 69 | 10.4 |
| Fish [b]: plaice (*Pleuronectes platessa*), wolf-fish (*Anarhinchas lupus*), cod (*Gadus morhua*), herring (*Clupea harengus*) | 0.12 − 0.24 | 0.034 − 0.23 | 0.72 − 1.6 | − |

[a] data in dry weight; [b] data in fresh weight.

annual discharges of ^{99}Tc from Sellafield can be seen in Figure 4.14. There appears to be a similar trend in both plots but with a four-year lag, which corresponds to the estimated transit time of three to four years.

## 4.2.3. Seaweed

Seaweed is an excellent bioindicator for ^{99}Tc in the marine environment because it has a very high ability to concentrate ^{99}Tc from seawater and is easily accessible in most coastal areas.

At Hillesøy on the north Norwegian coast, seawater and *Fucus vesiculosus* (a seaweed, commonly known as bladder wrack) have been analyzed monthly with respect to ^{99}Tc since 1997. It is clear from a plot of annual average activity concentrations of ^{99}Tc in seaweed at Hillesøy and data showing the annual discharge of ^{99}Tc from Sellafield (Figure 4.15), that the peak in ^{99}Tc activity concentrations in seaweed at Hillesøy occurred about four to five years after the corresponding discharge from Sellafield. The increasing ^{99}Tc levels in seaweed at Hillesøy since 2004 can be explained by the higher discharge of ^{99}Tc from Sellafield in 2001 and 2002 compared to the period 1998 to 2000.

It is apparent that activity concentrations of ^{99}Tc in *F. vesiculosus* responded rapidly to the increased ^{99}Tc levels in the seawater up to mid-2001 (Figure 4.16). From mid-2001 onwards, a decreasing trend is observed for ^{99}Tc activity concentrations in seawater, while the activity concentrations in *F. vesiculosus* decreased at a slower rate. Future studies examining the trends in ^{99}Tc activity concentrations in seawater and seaweed at this site will show whether there will be a comparable decrease in the seaweed with time. If so, this will make it possible to estimate the ecological half-life or restitution time for seaweed.

In 2004, activity concentrations of radionuclides in seaweed were obtained for the naturally-occurring radionuclide [40]K and the two anthropogenic radionuclides ^{99}Tc and ^{137}Cs for Greenland, Denmark Strait and the Faroe Islands. Activity concentrations for ^{137}Cs and ^{99}Tc in seaweed were within the range 0.2 Bq/kg dw to 2.7 Bq/kg dw and 4 Bq/kg dw to 33 Bq/kg dw, respectively. The highest activity concentrations of ^{99}Tc and ^{137}Cs in seaweed were found on the east coast of Greenland, reflecting the influence of Chernobyl fallout and industrial pollution from Europe on the East Greenland Current. Activity concentrations of [40]K in *F. vesiculosus* from the Faroe Islands were 1320 Bq/kg dw to 1350 Bq/kg dw (Nielsen and Joensen, 2009).

Russian data for artificial radionuclides in seaweed samples obtained near the Teriberka settlement on the Barents Sea coast (see Figure 4.12) in 2006 are shown in Table 4.4. The values are low for ^{137}Cs, ^{90}Sr, [239,240]Pu and [241]Am in all seaweed species analyzed.
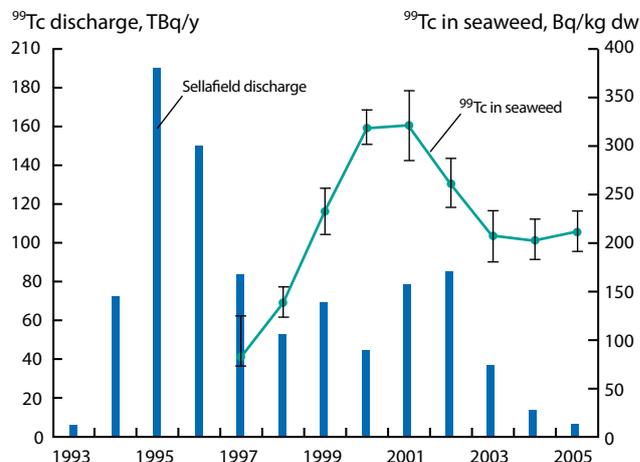


**Figure 4.15.** Annual average activity concentrations of ^{99}Tc in seaweed *(Fucus vesiculosus)* at Hillesøy, northern Norway, and annual discharges of ^{99}Tc from the Sellafield nuclear fuel reprocessing plant, UK, since the early 1990s.
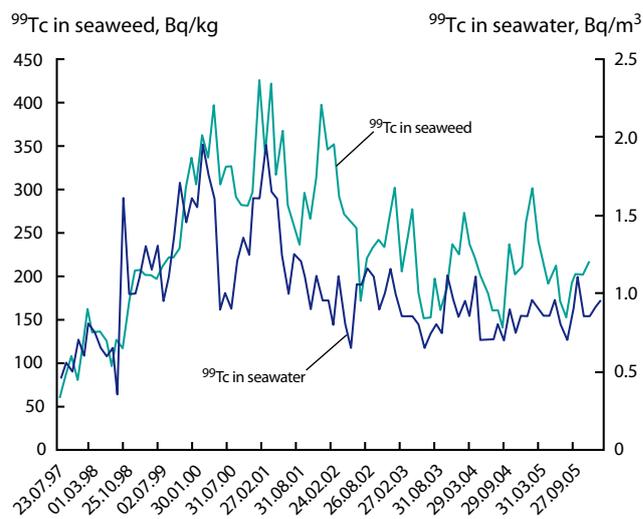


**Figure 4.16.** Variability in activity concentrations of ^{99}Tc in seawater and seaweed *(Fucus vesiculosus)* sampled at Hillesøy in northern Norway since the late 1990s.

## 4.2.4. Fish

Commercially important fish species, including farmed salmon, have been collected from various marine waters and analyzed with respect to [137]Cs.

Activity concentrations for ^{137}Cs in fish caught off Greenland fall within the range 0.1 Bq/kg ww to 0.3 Bq/kg ww, with higher levels on the east coast of Greenland at Ittoqqortoormiit (Scoresbysund) than on the west coast at Qaanaaq. This supports the pattern of seawater transport. Activity concentrations of ^{137}Cs in cod and haddock from the Faroe Islands over recent years were within the range 0.1 Bq/kg ww to 0.2 Bq/kg ww. A time series for ^{137}Cs in cod and haddock

caught off the Faroe Islands since 1960 (Figure 4·17) shows a progressive decrease from around 3 Bq/kg ww to around 0.1 Bq/kg ww and that activity concentrations appear to be slightly higher for cod than haddock.

Activity concentrations of $^{137}$Cs in cod from the Barents Sea have been analyzed every year since 1992 (Figure 4·18). All the data are below 1 Bq/kg ww, and with most below 0.5 Bq/kg ww, and there appears to be a slightly decreasing trend in the period 1992 to 2005.

Activity concentrations for $^{40}$K in shorthorn sculpin (*Myoxocephalus scorpius*) at Greenland were within the range 70 Bq/kg ww to 90 Bq/kg ww. These levels are directly related to the potassium content (which is under strict homeostatic control in animals) and therefore show very small variation regardless of variation in environmental levels.

Russian data for artificial radionuclides in marine fish

obtained near the Teriberka settlement on the Barents Sea coast in 2006 are shown in Table 4.4. All values are very low.

## 4.2.5. Seabirds

Cesium-137 has been determined in muscle for a range of seabirds from the Svalbard archipelago. Activity concentrations of $^{137}$Cs were either low or, as in the majority of cases, below detection limits. Where $^{137}$Cs activity concentrations were above detection limits, observed values ranged from $0.08 \pm 0.02$ Bq/kg ww to $0.18 \pm 0.05$ Bq/kg ww (NRPA, 2007a).

Polonium-210 is known to concentrate in marine organisms to a higher extent than other naturally-occurring alpha emitters, particularly in certain organs and is typically the greatest contributor to natural radiation doses to humans via ingestion of seafood. However, little is known about the trophic transfer and resulting activity concentrations of $^{210}$Po and $^{210}$Pb in seabirds, an important consumer group in marine food webs (NRPA, 2007a).

Activity concentrations of $^{210}$Po in muscle showed some variation between Arctic species of seabirds, with mean activity concentrations ranging from $1.10 \pm 0.48$ Bq/kg ww in glaucous gulls (*Larus hyperboreus*) to $13.67 \pm 7.24$ Bq/kg ww in little auks (*Alle alle*). The observed variation in $^{210}$Po activity probably reflects the differences in diet between the species. For example, little auks feed mainly on copepods which are known to accumulate high levels of $^{210}$Po (Carvalho, 1988), whereas glaucous gulls typically predate on other seabirds. Activity concentrations of $^{210}$Pb in muscle were negligible. Activity concentrations of $^{210}$Po in pooled seabird kidney samples were between 7- and 21-fold higher than mean muscle values and generally showed the same trend across species as activity concentrations in muscle (Table 4.5).

## 4.2.6. Cetaceans

As top predators in the aquatic food chain, fish-eating seals are vulnerable to the accumulation of contaminants. Measurements of $^{210}$Po and $^{210}$Pb activity concentrations in grey seals (*Halichoerus grypus*) from the Baltic Sea, in ringed seals (*Phoca hispida*) from Lake Saimaa and in ringed seals from the Arctic Ocean have been undertaken by the Finnish Radiation and Nuclear Safety Authority (STUK). Activity concentrations of $^{210}$Po and $^{210}$Pb in seals were determined in muscle, liver, kidney, bone and spleen (Table 4·6). The highest $^{210}$Po and $^{210}$Pb concentrations were found in liver and kidney samples, it is clear that Arctic seals have very high $^{210}$Po concentrations (Solatie et al., 2005).

Concentrations of naturally-occurring and anthropogenic radionuclides were determined in liver and muscle of seals and whale flesh from Greenland. Activity concentrations of $^{137}$Cs in seals from 2004 were within the range 0.1 Bq/kg ww

$^{137}$Cs activity concentration, Bq/kg ww
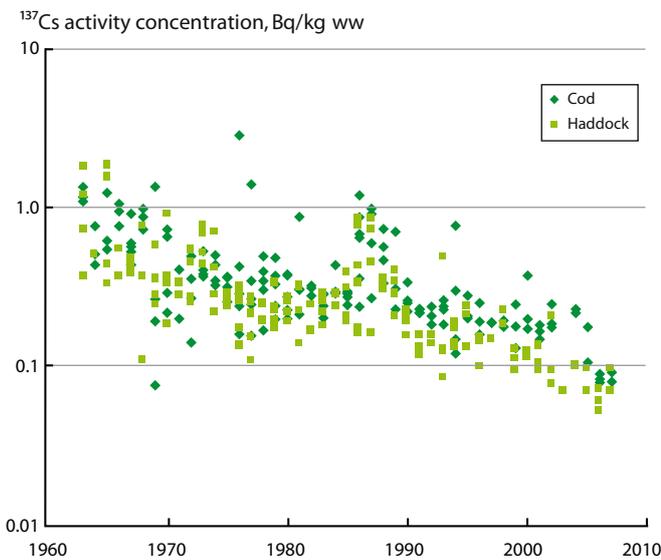


**Figure 4·17.** Activity concentrations of $^{137}$Cs in cod and haddock from the Faroe Islands since 1960.

$^{137}$Cs activity concentration, Bq/kg ww
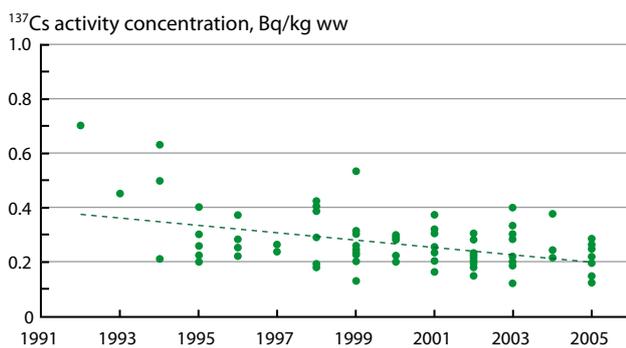


**Figure 4·18.** Activity concentrations of $^{137}$Cs in cod from the Barents Sea since the early 1990s.

**Table 4·5.** Activity concentrations of $^{210}$Po and $^{210}$Pb in various tissues (Bq/kg ww) for different seabird species. Source: NRPA (2007a).

| | n | $^{210}$Po | | | $^{210}$Pb | |
| --- | --- | --- | --- | --- | --- | --- |
| | | Muscle | Kidney | Liver | Kidney | Liver |
| Kittiwake (*Rissa tridactyla*) | 5 | $3.89 \pm 1.48$ | $31.9 \pm 0.3$ | – | $0.42 \pm 0.02$ | – |
| Brunnich's guillemot (*Uria lomvia*) | 6 | $11.6 \pm 5.1$ | $131.4 \pm 1.0$ | $42.9 \pm 0.3$ | $0.26 \pm 0.02$ | $0.19 \pm 0.01$ |
| Glaucous gull (*Larus hyperboreus*) | 2 | $1.10 \pm 0.48$ | $22.9 \pm 0.3$ | $7.18 \pm 0.1$ | $0.39 \pm 0.02$ | $0.016 \pm 0.002$ |
| Northern fulmar (*Fulmarus glacialis*) | 3 | $6.48 \pm 3.32$ | $47.0 \pm 0.6$ | – | $5.75 \pm 0.20$ | – |
| Little auk (*Alle alle*) | 5 | $13.7 \pm 9.9$ | $94.0 \pm 0.7$ | – | $0.67 \pm 0.04$ | – |

**Table 4.6.** Activity concentrations of $^{210}$Po and $^{210}$Pb (Bq/kg dw) in seal samples from the Baltic Sea and NE Svalbard in 2003 and 2004.

| | $^{210}$Po | $^{210}$Pb | dw % of ww | $^{210}$Po/$^{210}$Pb |
|---|---|---|---|---|
| Grey seal, Baltic Sea, 19 Sept. 2003 | | | | |
|    Muscle | 17 | 0.5 | 27 | 34 |
|    Liver | 82 | 2.6 | 27 | 32 |
|    Kidney | 68 | 3.4 | 23 | 20 |
|    Foreleg bone | 3.7 | 1.2 | 76 | 3.1 |
| Grey seal, Baltic Sea, 25 Sept. 2003 | | | | |
|    Muscle | 12 | 0.7 | 27 | 17 |
|    Liver | 57 | 3.3 | 29 | 17 |
|    Kidney | 67 | 2.7 | 22 | 25 |
|    Foreleg bone | 6.2 | 4.6 | 73 | 1.3 |
| Ringed seal, NE Svalbard, 15 May 2004 | | | | |
|    Muscle | 81 | 1.6 | 28 | 51 |
|    Liver | 490 | 3.1 | 29 | 160 |
|    Kidney | 610 | 2.6 | 23 | 230 |
| Ringed seal, NE Svalbard, 14 May 2004 | | | | |
|    Muscle | 100 | 0.5 | 28 | 200 |
|    Liver | 740 | 6.8 | 29 | 110 |

The data uncertainties are about 10%.
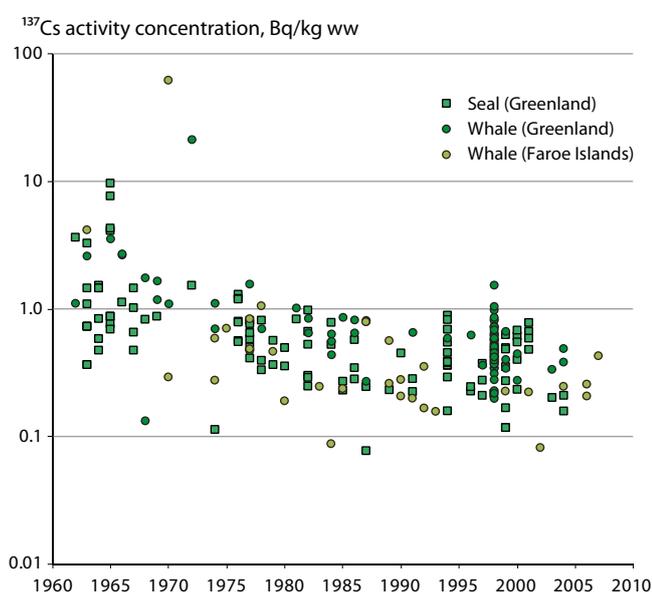


**Figure 4.19.** Trends in $^{137}$Cs activity concentrations in seals and whales from Greenland and in whales from the Faroe Islands since 1960.

to 0.6 Bq/kg ww, with the highest concentrations from the east coast of Greenland. Activity concentrations of $^{137}$Cs in minke whale (*Balaenoptera acutorostrata*) were within the range 0.4 Bq/kg ww to 0.5 Bq/kg ww. Concentrations of the naturally-occurring radionuclides $^{40}$K, $^{210}$Po and $^{210}$Pb in seal and whale were within the ranges 33 Bq/kg ww to 127 Bq/kg ww, 11 Bq/kg ww to 69 Bq/kg ww and 0.1 Bq/kg ww to 2.8 Bq/kg ww, respectively. Activity concentrations of $^{40}$K showed considerably less variation than $^{210}$Po and $^{210}$Pb. Activity concentrations of $^{210}$Po in seal muscle were within the range 10 Bq/kg ww to 11 Bq/kg ww and in seal liver within the range 29 Bq/kg ww to 69 Bq/kg ww. Activity concentrations of $^{210}$Pb in seal muscle are within the range 0.04 Bq/kg ww to 0.2 Bq/kg ww and in seal liver within the range 0.1 Bq/kg ww to 2.8 Bq/kg ww (Nielsen and Joensen, 2009).

Activity concentrations of 0.2 Bq/kg ww of $^{137}$Cs and 92 Bq/kg ww of $^{40}$K were found in samples from pilot whale (*Globicephala melas*) caught off the Faroe Islands. Figure 4.19 shows activity concentrations of $^{137}$Cs in seal and whale from Greenland and in whale from Faroe Islands since 1960.

The highest values were reported in the 1960s and early 1970s. There was a decreasing trend at the start of this period but levels have since stabilized at around 0.1 Bq/kg ww to 1 Bq/kg ww.

## 4.3. Radionuclides in the terrestrial and freshwater environments

### 4.3.1. $^{137}$Cs in soil

The previous AMAP assessment (AMAP 2004) reviewed $^{137}$Cs activity concentrations in the upper soil resulting from fallout following the Chernobyl accident. Additional humus layer samples are shown in Figure 4.20. Sampling was undertaken across large areas of northeastern Europe – northwest Russia, Finland, Norway and the Baltic countries – and represented several vegetation zones and highly variable soil profiles (Ylipieti et al., 2008). Activity concentrations in the humus layer reflect those in plants and mushrooms, and thus activity concentrations in food chains. The radioactive nuclide $^{137}$Cs is still the most significant fallout radionuclide in the environment. The figure shows that fallout from the Chernobyl accident can still be detected in the uppermost part of the soil in northeastern Europe.

### 4.3.2. Lakes, rivers and fish species

Table 4.7 shows mean annual activity concentrations for $^{90}$Sr and $^{3}$H in the rivers flowing into the White and Barents seas. For $^{90}$Sr there is a decreasing trend in activity concentrations, except for the Ob where the highest value was reported in 2002. For $^{3}$H there is no apparent trend, but the values are between 2 Bq/L and 3 Bq/L for all rivers.

Samples of lake water from Narsaq in southern Greenland and from Toftavatn in the Faroe Islands show that activity concentrations for $^{137}$Cs in lake water were higher from Toftavatn (3.6 Bq/m³) than from Narsaq (1.2 Bq/m³), whereas activity concentrations of $^{90}$Sr were higher from Narsaq (4.0 Bq/m³) than from Toftavatn (2.5 Bq/m³). The $^{90}$Sr and $^{137}$Cs in the lake water were due to fallout from both atmospheric
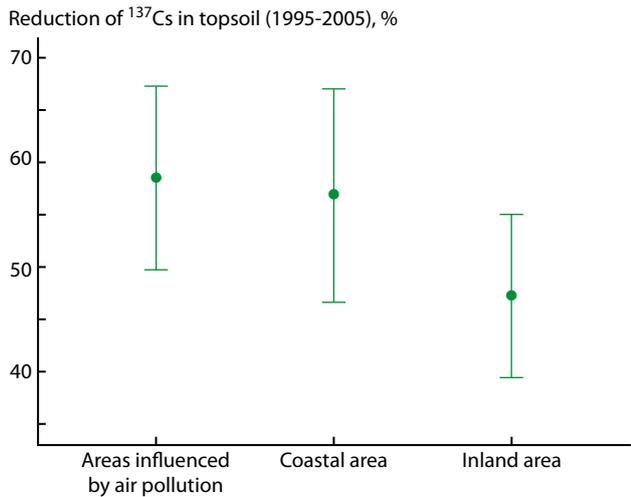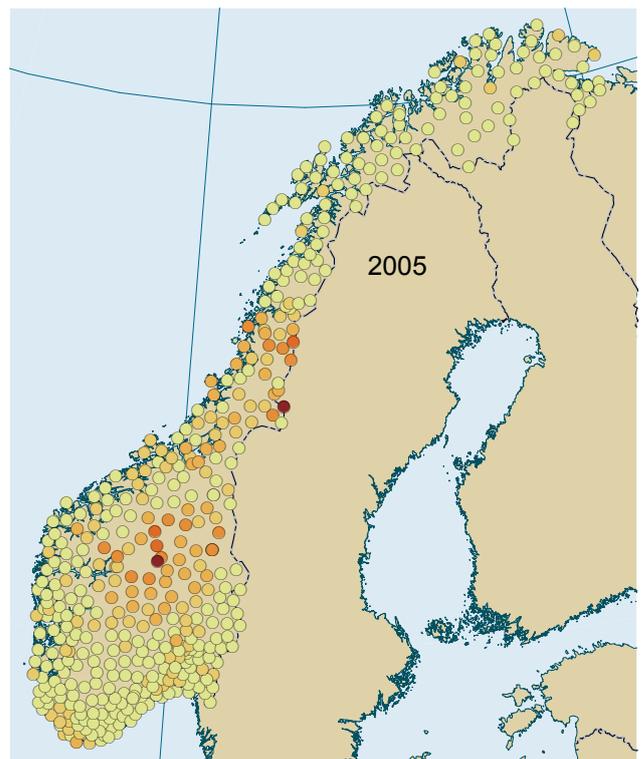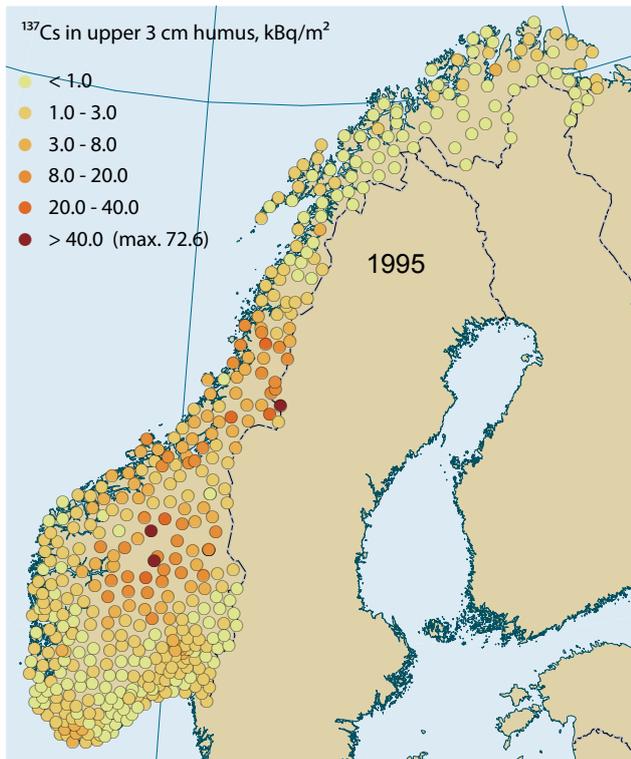
**Figure 4·20.** Activity concentrations of $^{137}$Cs in the upper 3 cm humus layer between 2000 and 2005.

nuclear weapons testing and the Chernobyl accident. Significantly higher activity concentrations of $^{90}$Sr relative to $^{137}$Cs in lake water from Narsaq compared to Toftavatn are probably due to more fallout at Toftavatn from the Chernobyl accident (Aarkrog, et.al, 1988). The level of $^{137}$Cs in landlocked Arctic char (*Salvelinus alpinus*) of 20 Bq/kg ww from the lake at Narsaq is relatively high compared to the level in water. A high uptake of radiocesium in freshwater fish is characteristic for oligotrophic lakes.

Lake Inari is the third largest lake in Finland (1040 km²). This is an oligotrophic Arctic lake rich in fish species and important for freshwater fishing. Lake Apukka is a very different type of lake; it is small (0.48 km²), shallow and highly eutrophic. Activity concentrations of $^{137}$Cs in fish from the two lakes are shown in Figure 4·21. Each point represents the
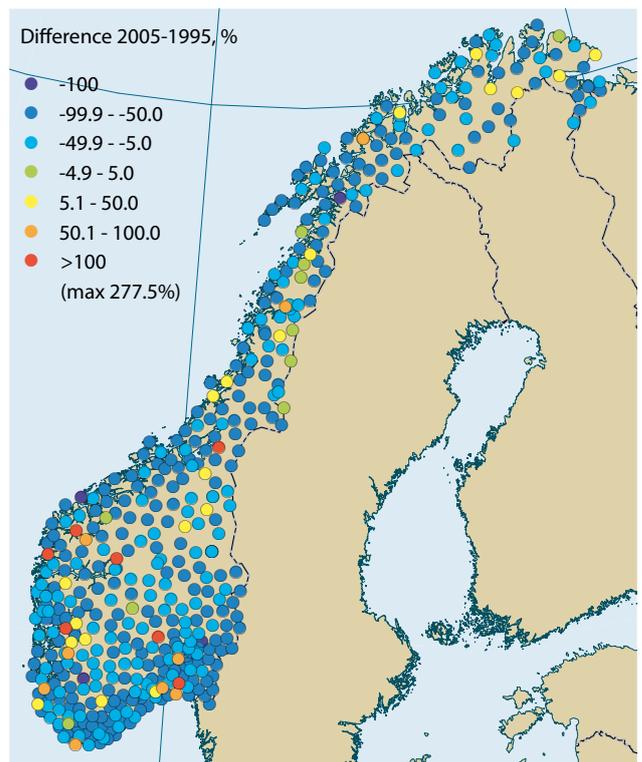
mean value of $^{137}$Cs activity concentrations in the relevant species per year. Following the Chernobyl accident in 1986 the highest $^{137}$Cs activity concentrations were found in pike (*Esox lucius*), which is the top predator (i.e., is piscivorous) in lake ecosystems and in species which are both predators and non-predators (i.e., are omnivorous) such as perch (*Perca fluviatilis*) and trout (*Salvelinus* spp.). The lowest $^{137}$Cs activity concentrations were found in whitefish (*Coregonus* spp.) and roach, which are non-predators (i.e., are non-piscivorous).

**Table 4.7.** Mean annual activity concentrations of $^{90}Sr$ (mBq/L) and $^3H$ (Bq/L) in rivers flowing into Russian Arctic Seas.

| | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|
| $^{90}Sr$ | | | | | | | |
| Severnaya Dvina | 6.3 | 8.8 | 7.6 | 7.0 | – | 7.3 | 5.6 |
| Pechora | 5.4 | 5.5 | 3.6 | 3.9 | – | – | – |
| Ob | 8.5 | 9.5 | 12.7 | 8.4 | – | – | – |
| Yenisey | 10 | 5.5 | 4.8 | 5.1 | – | – | – |
| Lena | 6.0 | 4.2 | 2.5 | 3.7 | – | – | – |
| $^3H$ | | | | | | | |
| Severnaya Dvina | 1.9 | 2.4 | 2.0 | 2.4 | 2.0 | 1.9 | 2.5 |
| Pechora | 2.4 | 2.7 | 2.6 | 2.0 | 2.2 | 2.3 | 2.9 |
| Ob | – | – | – | 2.7 | 2.3 | 2.7 | 2.6 |
| Yenisey | 3.2 | 3.7 | 3.3 | 2.7 | – | – | 2.8 |
| Lena | 3.5 | 3.7 | 3.1 | 2.0 | 2.8 | 3.2 | 3.0 |
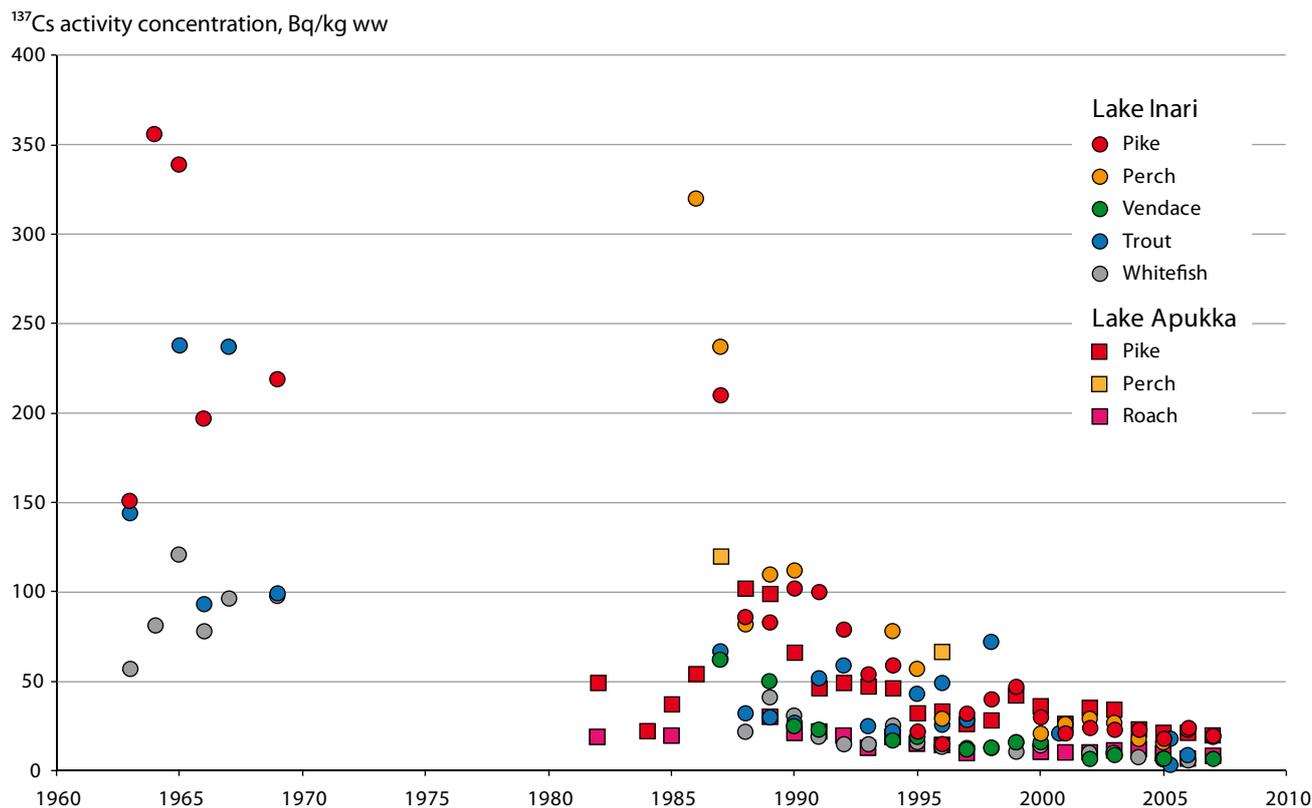


**Figure 4.21.** Activity concentrations of $^{137}Cs$ in various fish species in Lake Inari and Lake Apukka, Finland, since 1963.

The highest $^{137}Cs$ activity concentrations were found in the 1960s as a result of the atmospheric nuclear weapons tests. Although the number of fish samples was limited before the Chernobyl accident, it is still clear that $^{137}Cs$ activity concentrations were much higher compared to samples after the accident. The variation in $^{137}Cs$ activity concentrations in fish, especially in the first years after the accident was considerable (Figure 4.21). In the three years following the Chernobyl accident (1987, 1988, 1989) mean values in fish were between 116 Bq/kg ww and 237 Bq/kg ww in Lake Inari.

**Table 4·8**. Estimated ecological half-lives (y) and $R^2$ values (in brackets) in fish from two lakes in Finland after the Chernobyl accident.

| | Lake Inari | | | Lake Apukka | | |
| --- | --- | --- | --- | --- | --- | --- |
| | Pike | Perch | Whitefish | Pike | Perch | Roach |
| 1987 – 1996 | 3.0 (0.79) | 4.5 (0.78) | 2.2 (0.60) | 4.6 (0.86) | 3.2 (0.91) | 4.3 (0.59) |
| 1997 – 2007 | 9.9 (0.62) | 19.9 (0.23) | 6.7 (0.54) | 24.6 (0.35) | 28.2 (0.26) | 20.9 (0.47) |
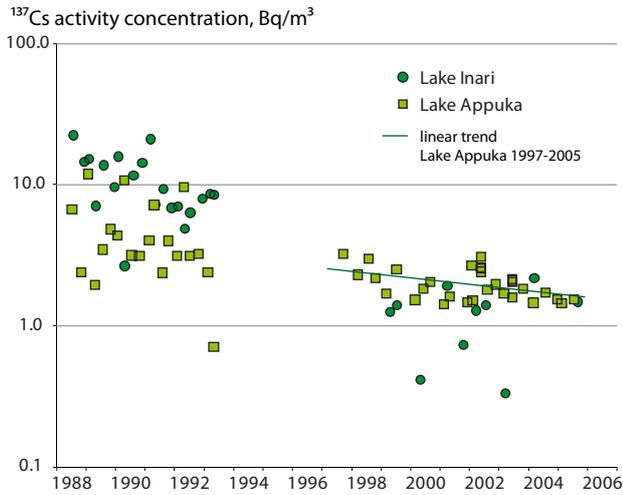


**Figure 4·22.** Activity concentrations of $^{137}Cs$ in water samples from Lake Inari and Lake Apukka, Finland, following the Chernobyl accident in 1986.

Ecological half-lives were calculated in three fish species in both lakes (Table 4.8). Pike and perch were studied both in Lake Inari and Lake Apukka, but whitefish only in Lake Inari and roach only in Lake Apukka. Whitefish and roach were classed as a non-piscivorous species. Ecological half-lives varied between 2.2 years and 4.6 years in the period 1987 to 1996 in all species in both lakes. In the second period, 1997 to 2007, the ecological half-lives varied widely between the lakes. Ecological half-lives were near the physical half-lives (30 years) in piscivorous and omnivorous species in Lake Apukka (Ylipieti and Solatie, 2008).

Freshwater data were available from 1988 to 2005 in both lakes. Variability in $^{137}Cs$ activity concentrations was large a few years after the Chernobyl accident; values ranged between 0.3 Bq/m³ and 22 Bq/m³ in Lake Inari and between 0.7 Bq/m³ and 12 Bq/m³ in Lake Apukka. Activity concentrations in Lake Inari decreased from a mean value of 10 Bq/m³ for the earlier period to 1 Bq/m³ for the later period, and in Lake Apukka from a value 5 Bq/m³ to 1 Bq/m³ (Figure 4·22). Thus, $^{137}Cs$ activity concentrations decreased more rapidly in Lake Inari than in Lake Apukka.

Relative to earlier studies, $^{137}Cs$ activity concentrations in different fish species in lakes in southern Finland increased to a maximum in the second, third or even fourth year after the Chernobyl accident. This effect was not observed in Lake Apukka and Lake Inari in northern Finland, where the fallout was much lower.

Differences in ecological half-lives in fish were very small during the first years after the accident (depending on the type of the lake or fish species). However, big differences were apparent eight years after the accident. In Lake Inari, which is oligotrophic, ecological half lives were much lower in the later period than in Lake Apukka, which is eutrophic, especially in species such as pike and perch. The main reason for long half-lives in fish in Lake Apukka is because conditions in the lake catchment help maintain $^{137}Cs$ levels in the water.

### 4.3.3. Wild berries

Berry pickers using wild berries for their own needs as well as for selling to industries that process berries for other consumers represents an Arctic food chain. Berries are used within households for desserts or jam and are mostly offered fresh or defrosted from frozen.

Samples of bilberry (*Vaccinium myrtillus*), lingonberry (*V. vitis idaea*), cloudberry (*Rubus* spp.) and cranberry (*Oxycoccux* spp.) were collected between 1980 and 2006 in the provinces of Lapland and Oulu, Finland (Table 4·9; Ylipieti and Solatie, 2007). The average $^{137}Cs$ activity concentration in lingonberry was slightly under 10 Bq/kg ww and in bilberry was over 10 Bq/kg ww. The highest $^{137}Cs$ activity concentration was measured in cloudberry. Mean activity concentrations in cloudberry before and after the Chernobyl accident in 1986 were both around 40 Bq/kg ww. The highest $^{137}Cs$ activity concentrations for cloudberry reached almost 250 Bq/kg ww. The cranberry samples were collected in 2006 from four locations in the municipalities of Puolanka, Savukoski, Kemijärvi and Hyrynsalmi. The statistical representativeness of the $^{137}Cs$ activity concentrations in this species is low due to the small number of samples.

Activity concentrations in all berries were under 600 Bq/kg ww, which is the recommendation of the European Commission (2003/120/EC). The average annual consumption of wild berries per capita in northern Finland is 13 kg. In the worst case: the consumption of berries picked with the highest $^{137}Cs$ activity concentration area (from the municipality of Kuhmo) would result in an annual dose to a consumer from bilberry of 2 µSv, from lingonberry of 6 µSv, and from cloudberry of 8 µSv. In total, the annual dose to a consumer of all wild berries would be a maximum of 16 µSv, which is 0.4% of the average whole annual dose to Finish people. In the rest of Finland the annual dose is much lower.

Data for radionuclides in terrestrial and marine foods, including berries and other wild foods adjacent to Andreeva Bay and Gremikha are available through collaborative work between the Norwegian Radiation Protection Authority and the Russian Federal Medical-Biological Agency (Shandala et al., 2008b). Table 4·10 shows the levels in berries in the environment near both STSs, which are about the same in the different areas listed here except at Gremikha SZ where the level is significantly lower. Compared to the mean level found in the Finish berries, these levels are slightly higher than levels found in Finish bilberry and lingonberry, and slightly lower than found in cloudberry (see Table 4·9).

### 4.3.4. Fungi

STUK has monitored $^{137}Cs$ activity concentrations in various species of fungi in the Kivalo research area in northern Finland since 1983. Activity concentrations increased rapidly in most fungi species after the Chernobyl accident in 1986, although the average $^{137}Cs$ deposition was low in northern

**Table 4.9.** Activity concentrations of $^{137}$Cs in wild berries collected in Lapland and Oulu, Finland, before and after the Chernobyl accident. Source: Ylipieti and Solatie (2007).

| | n | Mean | Standard Deviation | Activity, Bq/kg fw | | |
| --- | --- | --- | --- | --- | --- | --- |
| | | | | Minimum | Maximum | Median |
| Bilberry | | | | | | |
|    Pre-Chernobyl | 44 | 15.0 | 3.5 | 11 | 19 | 15 |
|    Post Chernobyl | 122 | 10.1 | 7.48 | 1.67 | 44.7 | 8 |
| Lingonberry | | | | | | |
|    Pre-Chernobyl | 5 | 8.6 | 3.13 | 6 | 12 | 7 |
|    Post Chernobyl | 119 | 8.4 | 9.92 | 0.78 | 78.9 | 6.11 |
| Cloudberry | | | | | | |
|    Pre-Chernobyl | 6 | 38 | – | – | – | – |
|    Post Chernobyl | 98 | 41.1 | 35.2 | 10 | 245 | 30 |
| Cranberry | | | | | | |
|    Pre-Chernobyl | 1 | 1.41 | – | 1.41 | 1.41 | 1.41 |
|    Post Chernobyl | 4 | 15.6 | 14.7 | 3.6 | 36.45 | 11.2 |

**4.10.** Activity concentrations of radionuclides in berries (Bq/kg) picked adjacent to Andreeva Bay and Gremikha. Source: Shandala et al. (2008b).

| | STS in Andreeva | | STS in Gremikha | | Norm or background | Radiological hazard assessment |
| --- | --- | --- | --- | --- | --- | --- |
| | SSZ | SA | SSZ | SZ | | |
| $^{90}$Sr | 9 | 3 | 9 | 0.3 | 6 | Not more than acceptable specific activity |
| $^{137}$Cs | 24 | 23 | 27 | 2 | 60 | Not more than acceptable specific activity |
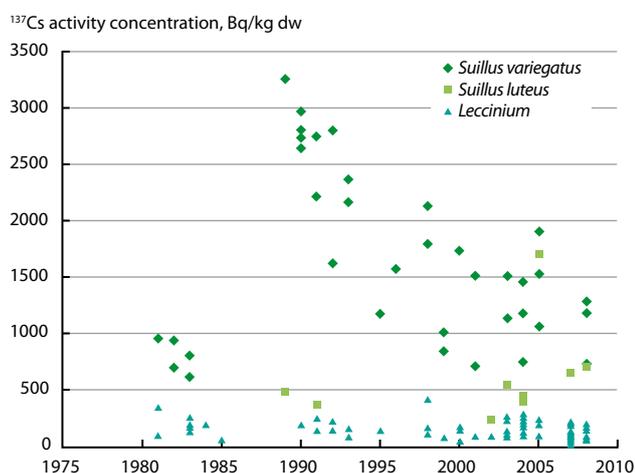


**Figure 4.23.** Changes in $^{137}$Cs activity in three fungi species over time.



**Figure 4.24.** Measured $^{90}$Sr and $^{137}$Cs activity in deposition at Apukka, Rovaniemi, Finland.

Finland. Recent studies show that cesium retention varies between species. For example, $^{137}$Cs activity concentrations have decreased slowly in the species *Suillus variegatus* over the last 20 years, whereas those in *Leccinium* have changed little. $^{137}$Cs activity concentrations have increased in *Suillus luteus* (Figure 4.23). Mushrooms are important indicators of cesium in forest ecosystems; a 'good mushroom year' could indicate increased cesium levels in reindeer meat and other animals that consume large amount of mushrooms.

### 4.3.5. $^{90}$Sr and $^{137}$Cs in deposition, grass and milk

As a result of the atmospheric nuclear weapons testing and the Chernobyl nuclear power plant accident in 1986, $^{90}$Sr and $^{137}$Cs have entered the environment (Figure 4.24) and consequently, the food chain. Because $^{90}$Sr has very similar chemical behavior to calcium, and $^{137}$Cs has very similar behavior to potassium, both follow the paths of these elements in the food chain and so enter the human body. Grass is an efficient collector of atmospheric contaminants, and radionuclides are rapidly transferred from grass to milk.

Soil→grass→milk is one of the main food chains introducing radioactivity into humans in Finland. Dairy milk (milk from several farms that has usually been processed, e.g., homogenized, pasteurized) and farm milk (milk taken straight from the milk tank on a single farm) have been analyzed for radionuclides since 1963, as has atmospheric deposition at various sites in northern Finland since 1972. In Finland milk is the main source of $^{90}$Sr to humans. Figure 4.25 and Table 4.11 (Solatie et al., 2008) show half-lives of $^{90}$Sr and $^{137}$Cs in different samples. The half-lives of these radionuclides in deposition represent the input to the soil→grass→milk food chain while the half-lives in milk represent the output. A comparison of the two data sets reveals that, for milk, for longer time intervals, the half-lives of $^{90}$Sr are greater than for $^{137}$Cs. Similarly, half-lives in deposition and milk can be observed for $^{137}$Cs in the 1980s prior to the Chernobyl accident. Longer half-lives could be expected in milk at present because contaminants are still enriched in the soil→grass→milk food chain.

**Table 4.11.** Half-lives and activity concentrations for $^{90}$Sr and $^{137}$Cs in atmospheric deposition at different periods in Finland.

| | Time period | Half life, y | Fallout, Bq/m² |
|---|---|---|---|
| $^{90}$Sr | | | |
| nuclear era | Oct. 1972 – June 1981 | 29.9 | 1 – 12 |
| middle years | June 1981 – Mar. 1986 | 2.7 | 0.2 – 4.0 |
| post Chernobyl | Aug. 1986 – Mar. 1999 | 2.5 | 0.01 – 3.3 |
| $^{137}$Cs | | | |
| nuclear era | Oct. 1972 – June 1981 | 11.4 | 0.4 – 15 |
| middle years | June 1981 – Mar. 1986 | 2.7 | 0.3 – 6.0 |
| post Chernobyl | Aug. 1986 – June 2007 | 2.4 | 0.01 – 3.3 |
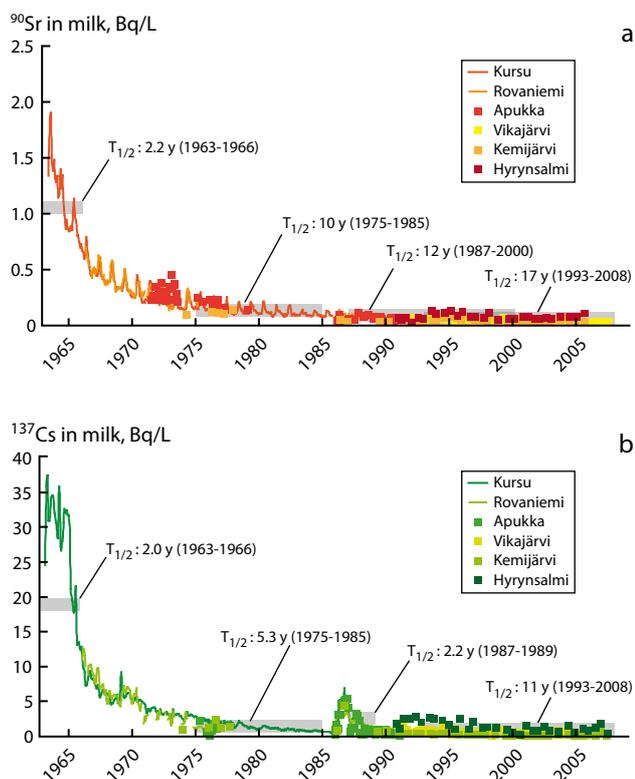


**Figure 4.25.** Activity concentrations and ecological half-lives of (a) $^{90}$Sr and (b) $^{137}$Cs in milk in Lapland since the early 1960s.

## 4.3.6. The lamb food chain

Activity concentrations of $^{137}$Cs in soil, mixed grass and lamb meat have been monitored at several sites in the Faroe Islands since 1990 (Figure 4.26). This is because lamb meat is an important food component for local people. Soil and grass samples were taken from 0.25 m² plots in late July or early August and four plots were chosen at each site. The grass was cut from each plot, after which three soil cores were taken, each 5.7 cm in diameter and 10 cm in depth. The meat samples are neck muscle collected at the time of slaughter, typically in first half of October. Soil data are presented in Table 4.12. The low pH and high loss on ignition are conditions that favor high uptake of radiocesium. The main pathway for $^{137}$Cs deposition to the terrestrial environment is by precipitation. Precipitation data are available at some of the sites (Table 4.13). There is an approximate 4-fold difference between the lowest and highest precipitation rate. The geographic variability in the soil parameters and the precipitation rate imply regional variation in the activity concentrations of $^{137}$Cs in soil, mixed grass and lamb meat (Figure 4.27).

The $^{137}$Cs content in soil, mixed grass and lamb meat decreased at most sites between 1990 and 2005, but not monotonically (Figure 4.27). The highest values occurred at the site with the highest precipitation rate, Hvalvík (Table 4.13). The data enabled the effective ecological half-life of
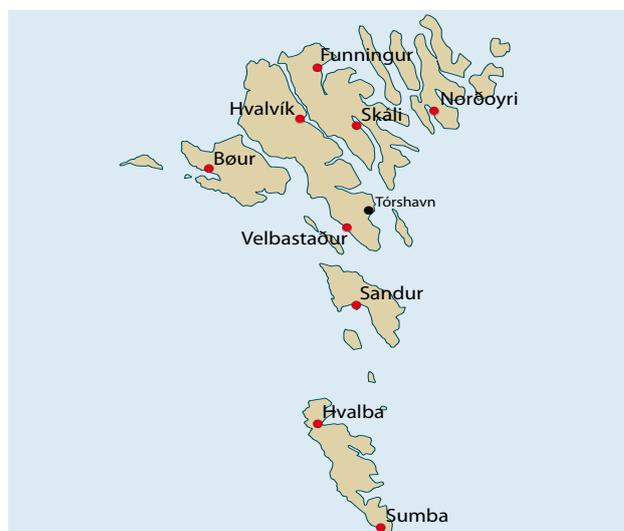


**Figure 4.26.** Sampling locations in the Faroe Islands as referred to in Figure 4.27.

**Table 4.12.** Soil characteristics for the top 10 cm soil layer in the Faroe Islands. Average (± 1SE) for the eight years between 1990 and 1997. Source: Joensen (1999).

| | Bøur | Velbastað | Hvalvík | Skáli | Funnings-fjørður | Norðoyri | Sandur | Hvalba | Sumba |
|---|---|---|---|---|---|---|---|---|---|
| pH | 4.9 ± 0.05 | 5.1 ± 0.11 | 4.8 ± 0.09 | 4.9 ± 0.06 | 4.8 ± 0.07 | 4.7 ± 0.10 | 4.8 ± 0.11 | 5.1 ± 0.07 | 4.9 ± 0.07 |
| Loss on ignition, % | 52 ± 4.1 | 30 ± 3.1 | 67 ± 3.8 | 56 ± 3.6 | 51 ± 3.1 | 67 ± 4.0 | 52 ± 4.4 | 63 ± 2.7 | 56 ± 5.2 |

**Table 4.13.** Precipitation rate (mm/y) in the Faroe Islands. Source: Cappelen and Laursen (1998).

| Bøur (1988 – 1997) | Hvalvík (1988 – 1997) | Norðoyri (1961 – 1990) | Sandur (1961 – 1990) | Sumba (1961 – 1990) |
|---|---|---|---|---|
| 1555 | 3261 | 2710 | 1193 | 884 |

**¹³⁷Cs in soil (0-10cm), Bq/m²**   a



**¹³⁷Cs in grass, Bq/kg dw**   b



**¹³⁷Cs in lamb meat, Bq/kg ww**   c
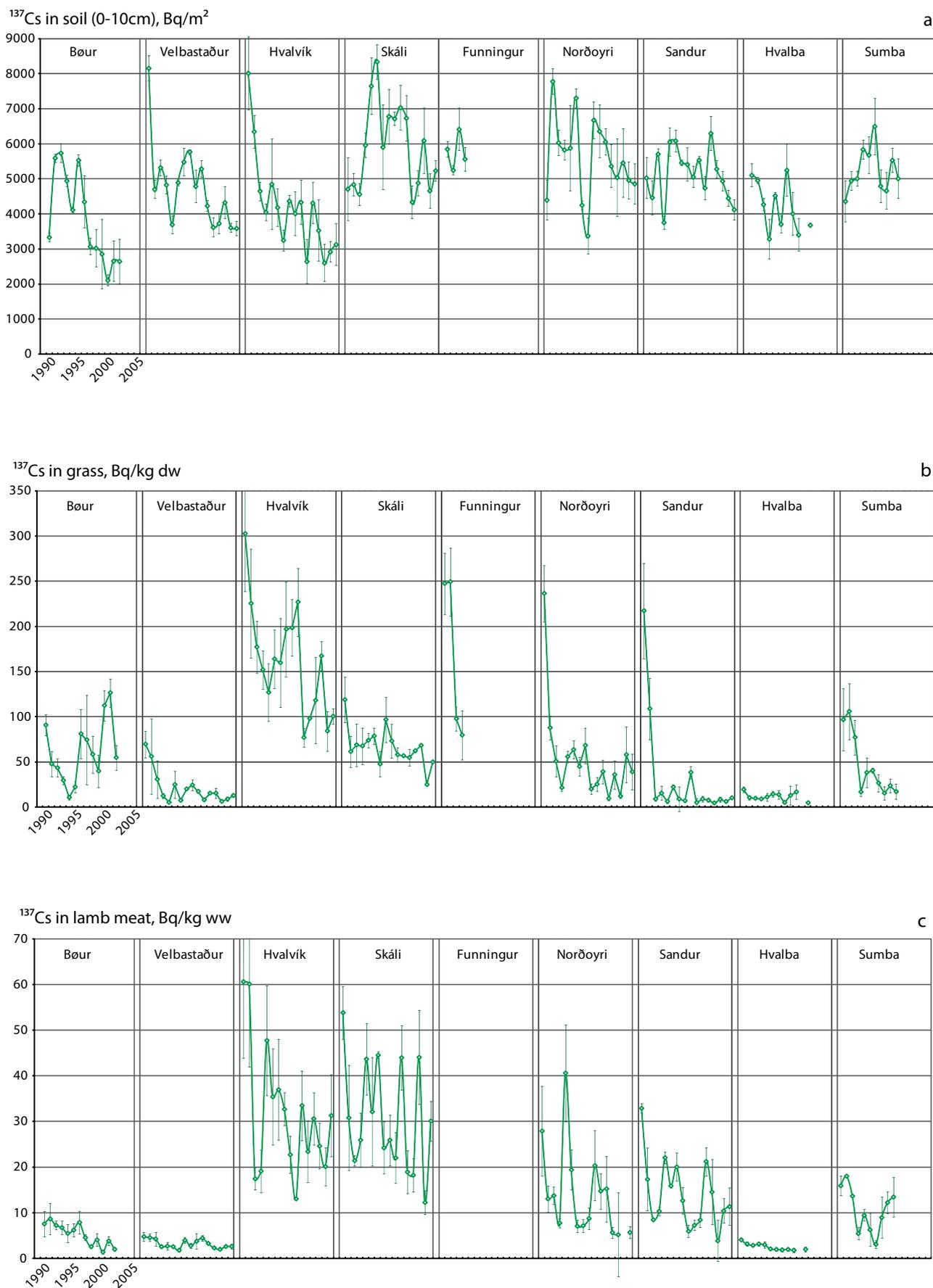


**Figure 4·27.** Activity concentrations for ¹³⁷Cs in (a) the upper soil layer, (b) in mixed grass, and (c) in lamb meat at various locations in the Faroe Islands since 1990.

**Table 4·14.** Effective ecological half-life (y) of $^{137}$Cs in soil, grass and lamb meat in the Faroe Islands based on measurements between 1990 and 2005. Time series do not all cover the same time period. Numbers in brackets represent $R^2$ from a linear regression between time and natural logarithm of $^{137}$Cs activity concentration in the samples. No estimates are given when $R^2 < 0.300$.

|                    | Bøur    | Velbastað | Hvalvík | Skáli   | Norðoyri | Sandur  | Hvalba  | Sumba   |
|--------------------|---------|-----------|---------|---------|----------|---------|---------|---------|
| Soil, Bq/m²        | 11.4    | 21.7      | 13.9    | –       | –        | –       | –       | –       |
|                    | (0.528) | (0.462)   | (0.636) | (0.005) | (0.044)  | (0.015) | (0.253) | (0.035) |
| Grass, Bq/kg dw    | –       | 8.1       | 12.8    | 16.5    | 7.8      | 5.0     | –       | 3.6     |
|                    | (0.027) | (0.329)   | (0.437) | (0.357) | (0.302)  | (0.384) | (0.005) | (0.667) |
| Meat, Bq/kg fw     | 5.1     | –         | –       | –       | –        | –       | 9.9     | –       |
|                    | (0.668) | (0.194)   | (0.162) | (0.148) | (0.295)  | (0.202) | (0.781) | (0.069) |

$^{137}$Cs to be estimated in some cases (Table 4·14); 11.4 years to 21.7 years in soil (3 sites), 3.6 years to 16.5 years in grass (6 sites) and 5.1 years to 9.9 years in lamb meat (2 sites).

## 4.3.7. Reindeer and their forage

The transfer of radionuclides, especially $^{137}$Cs, in the lichen →reindeer/caribou→human food chain has been studied extensively in the Arctic since the nuclear weapons tests era in the 1950s and 1960s. Lichens effectively absorb nutrients and contaminants from air and precipitation, and were more contaminated than green plants after the nuclear fallout from the nuclear tests and the Chernobyl accident. Because lichens have no roots, external contamination is removed by weathering or decay of the lichens, and concentrations of internally incorporated contaminants will be diluted by growth. Long-term studies of $^{137}$Cs in lichens in Scandinavia after the Chernobyl fallout have shown effective half-lives of about 3 years (Rissanen et al., 2005; Lehto et al., 2008), as illustrated in Figure 4·28. On the other hand, green plants have an initially low uptake but will continue to absorb contamination from the soil, and activity concentrations of $^{137}$Cs in plants have thus been found to decrease more slowly than in lichens (Gaare et al., 2000; Rissanen et al., 2005).

Previous AMAP assessments have presented overviews of radionuclide contamination in reindeer/caribou in the Arctic, for example, in northwestern Russia, northern Norway and Iceland (AMAP, 2004a). In a recent review, Macdonald et al. (2007) summarized available measurements of $^{137}$Cs and $^{134}$Cs in the caribou herds of northern Canada, Alaska, and Greenland between 1958 and 2000. Figure 4·29 shows mean activity concentrations of $^{137}$Cs for all Canadian caribou herds for each calendar year from 1958 to 2000. Also shown is the estimated annual deposition of $^{137}$Cs. It is apparent that $^{137}$Cs activity concentrations have fallen by an order of magnitude since the peak in atmospheric deposition in the early 1960s and are now generally less than 100 Bq/kg. Using the annual deposition estimates, it was possible to derive an effective half-life of 6.1 years for the disappearance of radiocesium from caribou meat. The geographical distribution of radiocesium in Western Hemisphere caribou herds shows increasing activity concentrations from west to east, presumably due to higher precipitation in the east (Figure 4·30).

Figure 4·31 presents updated data on $^{137}$Cs activity concentrations in reindeer in northern Finland and Norway, in areas of low Chernobyl fallout. In the first years after the Chernobyl fallout the activity concentrations in reindeer in Kemin Sompio and Paistunturi (both northern Finland) decreased with effective half-lives of about 3 years. Thereafter the decrease had a half-life of about 9 years. The effective

$^{137}$Cs in lichen, Bq/kg dw
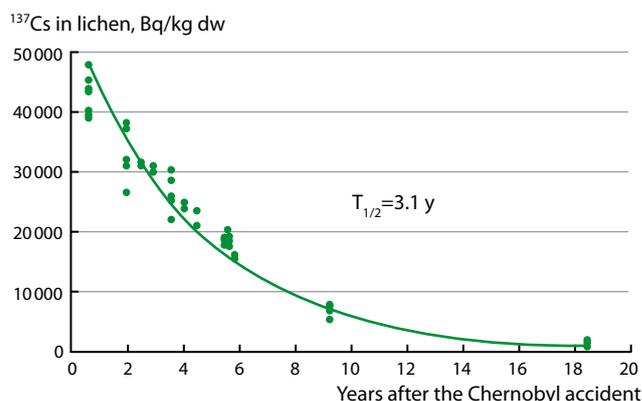


**Figure 4·28.** Levels of $^{137}$Cs activity in reindeer lichens (mainly *Cladina stellaris*) from a dry pine forest heath northwest of Uppsala, central Sweden. Source: Rissanen et al. (2005).

$^{137}$Cs in Caribou muscle, Bq/kg dw; $^{137}$Cs in deposition, Bq/m²
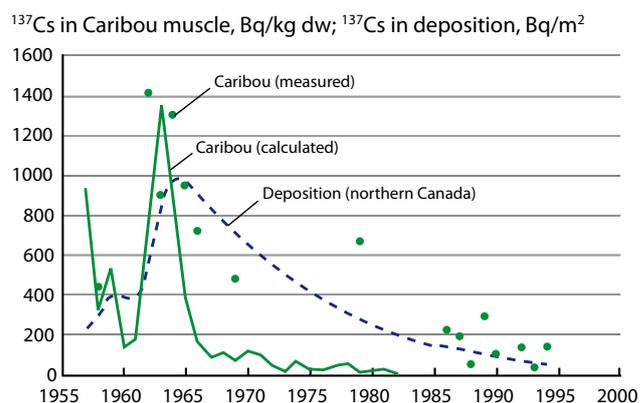


**Figure 4·29.** Levels of $^{137}$Cs activity in Canadian caribou herds since the mid-1950s. Source: Macdonald et al. (2007).

half-life in Ivalo (northern Finland) has been about 8 years during the whole post-Chernobyl period and has been about 9.5 years in Kautokeino (northern Norway) since 1990.

Seasonal changes in the diet of reindeer and caribou, from predominantly green plants during summer to a diet dominated by ground and arboreal lichens during winter, causes pronounced seasonal differences in radiocesium activity concentrations in the animals as long as lichens are significantly more contaminated than green plants (see example from Ivalo in Figure 4·31). The difference due to diet is also amplified by slower metabolism in reindeer and caribou during winter. Seasonal differences in $^{137}$Cs activities have not been particularly important issues in relation to reindeer and caribou in the Arctic. However, south of the Arctic Circle, in central Norway and Sweden, where the Chernobyl accident caused significantly higher radiocesium activity concentrations in reindeer, the lower $^{137}$Cs activities during early autumn made
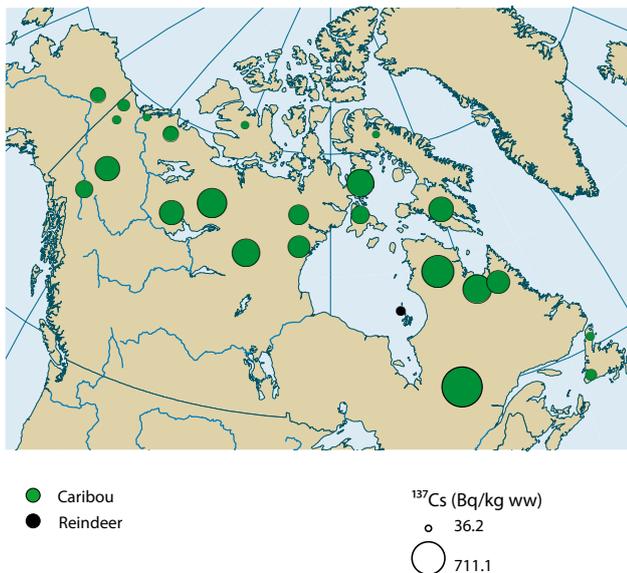
**Figure 4.30.** Geographical distribution of ^{137}Cs activity in caribou and reindeer between 1986 and 1989. The area of each circle is proportional to the pooled mean of all winter/spring concentrations for a given herd during this period. The single black circle represents a domestic reindeer herd on an island in Hudson Bay. Source: Macdonald et al. (2007).

possible the slaughtering of relatively low-contaminated reindeer. However, as illustrated in Figure 4.32, the differences in activity concentrations between early autumn and winter have reduced, and are not significant for recent years. This corresponds to the non-significant differences in ^{137}Cs activity concentrations in lichens and green plants observed in plants during 2001 to 2003 (Skuterud et al., 2005b). The decline in ^{137}Cs activity concentrations appears to be slower with time and varies between different areas. The proportion of pre-Chernobyl cesium in the pasture seems to influence the long-term rate of the decline (Åhman et al., 2001). A relatively larger contribution of green plants (with longer half-lives) to the intake of ^{137}Cs in reindeer will also gradually prolong the effective half-life in reindeer.

## 4.3.8. Humans

Previous AMAP assessments have presented overviews of radionuclide contamination in population groups that depend on reindeer and caribou for food. Following the atmospheric nuclear weapons tests fallout, these population groups had the highest activity concentrations of ^{137}Cs due to their intake of contaminated reindeer meat. Monitoring of ^{137}Cs in some of these groups is still ongoing, and contamination levels are falling (see examples from Finland and Norway in Figure 4.33).

Following the significant Chernobyl fallout in central Scandinavia, reindeer herders in central Norway and Sweden have had much higher activity concentrations of ^{137}Cs than reindeer herders in northern Fennoscandia. The northern reindeer herders have also had lower activity concentrations than population groups in the Chernobyl-affected regions that consume significant amounts of local products such as game, freshwater fish, berries and mushrooms (Rahola and Muikku, 2004). Around 2000, the mean body burden of the study population of local-product consumers in central Fin-
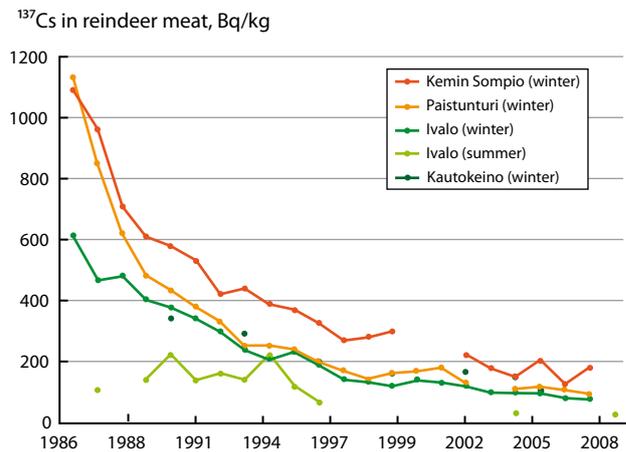


**Figure 4.31.** Activity concentrations for ^{137}Cs in reindeer meat from areas of northern Finland (Kemin Sompio, Paistunturi, Ivalo) and Norway (Kautokeino). Source: Rissanen et al. (2003), Solatie (pers. comm.) and Skuterud et al. (2005a).
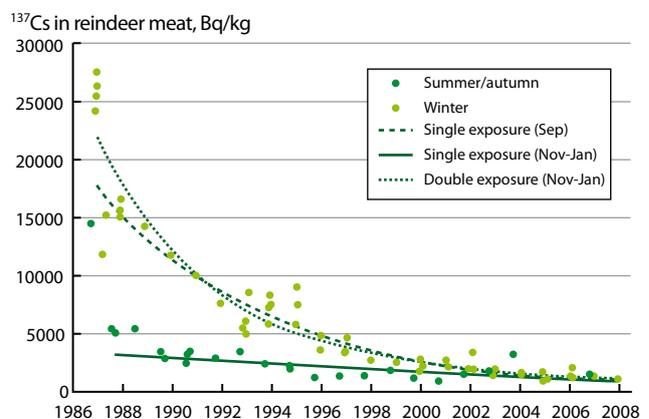


**Figure 4.32.** Levels of ^{137}Cs activity in reindeer meat from a Chernobyl-affected area in Norway (update of data presented by Skuterud et al., 2005b). The decrease in summer/autumn from 1987 onwards has a half-life of about 12 years. During winter, concentrations decreased initially by a half-life of about 3 years, increasing to current 12 years in the double exponential model.
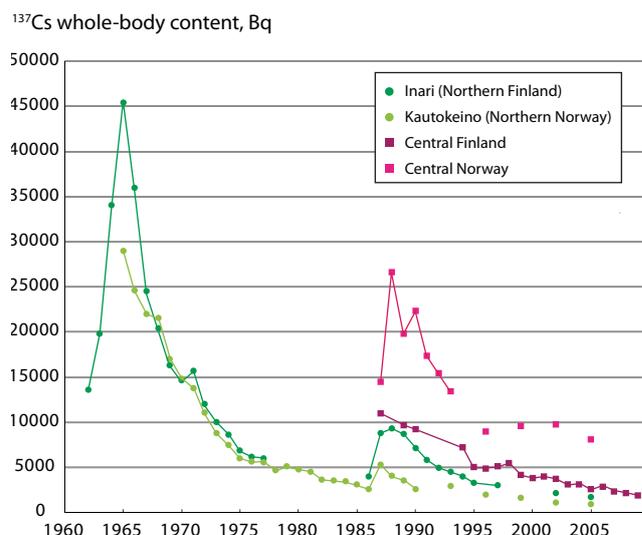


**Figure 4.33.** [137]Cs whole body content in reindeer herders in northern Finland and northern and central Norway (average values for males and females; from Rahola and Muikku (2005) and Skuterud et al. (2005a)).

land was more than ten-fold higher than the mean body burden of the general Finnish population (Rahola and Muikku, 2004). In the Chernobyl-affected areas in central Norway there is still a need for measures to reduce the intake of $^{137}$Cs by reindeer herders.

## 4.4. Concluding comments

Monitoring of radionuclides in the atmosphere in Finland and in seawater near Greenland and the Faroe Islands shows that traces of atmospheric weapons tests in the 1950s and 1960s are still detected, but have declined over time. Air monitoring data from Canada highlight how some of the fallout that has been incorporated into vegetation can be re-released into the environment through forest fires; cesium levels exceeding the detection limit have been shown to coincide with summer forest fires. Other data show that in spite of the peak of weapons testing have taken place over 50 years ago, the radiocesium from the fallout remains in the top layer of the soil. This is because processes that would normally favor mobility are slower in colder environments. Past fallout is thus likely to remain a source of radioactive contamination for grazing wildlife and for humans.

The 1986 accident at the Chernobyl nuclear power plant added further radiocesium to the environment, even though fallout in the Arctic was much less than further south in Fennoscandia and near to Chernobyl. The additional contamination is evident as small peaks in the atmospheric record as well as in the monitoring data for deposition and levels in vegetation and food products such as milk and meat. Monitoring data have been used to estimate the effective ecological half-lives of radionuclides in different environments and food webs. A food chain of major importance in the Arctic is lichen→reindeer/caribou→people. This has been studied extensively because it has been a major source of radionuclide intake by humans. Long-term studies in Scandinavia after the Chernobyl accident show that the effective ecological half-life for reindeer has increased from about 3 years shortly after the accident to 8 years to 9 years at present. Internal contamination in humans shows the same trend.

The monitoring data for terrestrial and freshwater environments show slow ecological half-lives, some also increasing with time. It is thus important to maintain the monitoring activities to be able to make predictions about the long-term consequences of radioactive contamination in the Arctic.

For the marine environment, the detection of $^{129}$I of European origin in the deep North Atlantic shows that radionuclide and other contaminant releases into the oceans can quickly become globally distributed. In general, the levels of anthropogenic radionuclides in the Arctic seawater are low, although they vary according to distance from sources and annual discharge rates. Some radionuclides may, however, concentrate in biota; best illustrated by the elevated levels of $^{99}$Tc in seaweed along the Norwegian coast. Concentrations in fish species are generally low and slowly decreasing with time. For seabirds, seals and whales that prey mostly on fish, there is a clear higher concentration of radionuclides in kidney and liver compared to muscle for the natural radionuclides $^{210}$Po and $^{210}$Pb.

*Chapter 5*

# Protection of the Arctic Environment

## 5.1. Background

### 5.1.1. Environmental protection – Arctic legal regime

The Arctic consists of territories of various nations, and as such has no overall and binding legal regime. As elsewhere, the framework for environmental protection of the Arctic is constituted by national laws. However, global treaties and norms influence national laws to an increasing extent – something that is undoubtedly linked to the special status of the Arctic environment. In particular, marine treaties have influenced the domestic laws, and much of the focus of environmental protection of the Arctic has therefore been marine conservation (Brown et al., 2003).

### 5.1.2. Special considerations for the protection of the Arctic environment

In the public perception, the Arctic might be considered as such an 'untouched' environment that, in many areas, any introduction of pollutants would be seen as adverse (Oughton, 2002). However, at the scientific level, there are other considerations that make the Arctic an interesting case study. There is evidence to suggest that the physical conditions in the Arctic may hypothetically alter radionuclide transfer to biota (Kryshev and Sazykina, 1986, 1990; Sazykina, 1995, 1998), at least in the case of poikilotherms. Indeed, it has been suggested that the slower digestion and metabolism of cold water animals, resulting in slower efflux rates than in warm water species, may result in differences in biological uptake within Arctic marine environments (Fisher et al., 1999). The modifying influence of Arctic climatic conditions upon the expression of radiation-induced effects has also been hypothesized by Sazkyina et al. (2003). The development of radiation effects in poikilothermic Arctic organisms is expected to occur more slowly because of low environmental temperatures. However, low temperatures, extreme seasonal variations in incoming solar radiation and lack of nutrients are physical and chemical environmental stressors of Arctic organisms that limit biodiversity. These also make Arctic ecosystems potentially more vulnerable to contaminants than organisms in other climatic regions (AMAP, 1998). In addition, the Arctic contains several potential radionuclide sources (Strand et al., 1997b).

### 5.1.3. Recent developments

Methodologies to assess the impact of exposure to ionizing radiation on flora and fauna in European temperate and Arctic environments were developed in two European collaborative projects: FASSET (Framework for Assessment of Environmental Impact; Larsson, 2004) and EPIC (Environmental Protection from Ionizing Contaminants; Brown et al., 2003). These studies have been superseded by the project

ERICA (Environmental Risk from Ionising Contaminants: Assessment and Management) wherein risk assessment methodologies have been developed and issues relevant to decision-making within the context of the management of environmental impacts of radioactivity have been addressed (Beresford et al., 2007a).

Within the last few years, the International Commission on Radiological Protection (ICRP) has begun to formulate ideas concerning protection of the environment (ICRP, 2003) and initial considerations with respect to a framework for environmental protection have been included in the new Basic Recommendations of the ICRP (2007a). It should be noted that the framework by design is highly generic
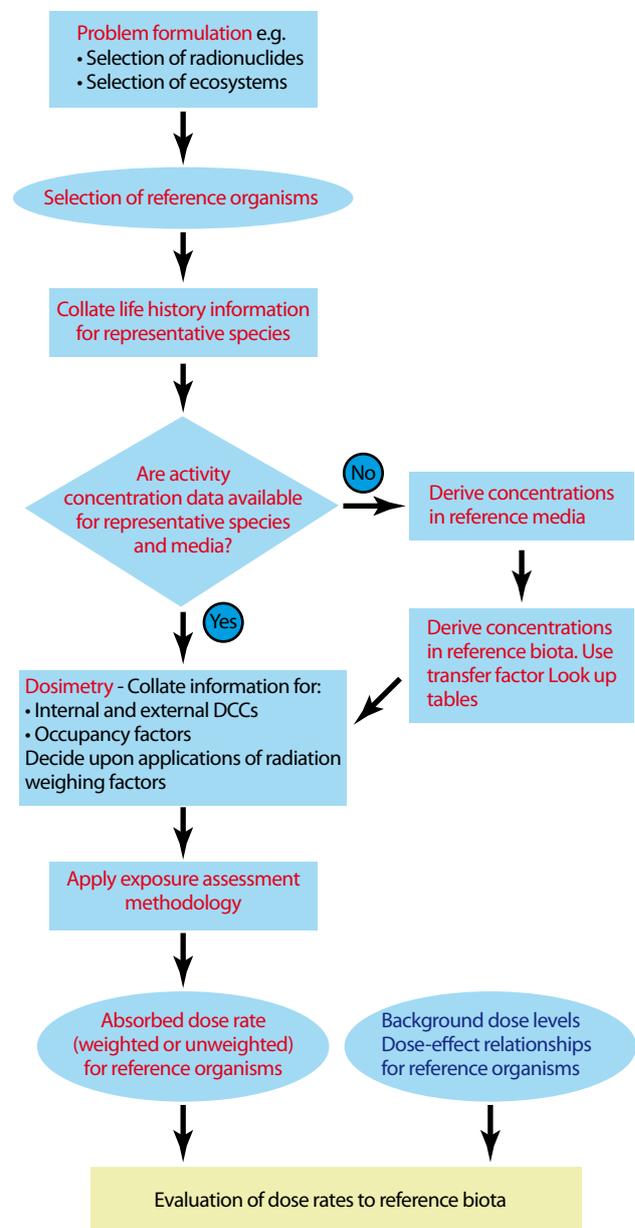


**Figure 5.1.** Flow diagram showing stages in the EPIC assessment methodology.

because the components constituting the system need to have relevance at a global level. The aspiration of the ICRP is to provide high level guidance for demonstration of compliance corresponding with existing/emerging national and international legislation, stating explicitly that the approach is being developed to provide a framework for more applied and specific numerical approaches. The ICRP will provide a primary set of reference values that can be related in a transparent way to the parameters applied in case- or site-specific assessments.

## 5.1.4. Emerging framework

Recent activities on the development of a protection framework for ionizing radiation have culminated in the reports of the ICRP, initially ICRP (2003) and then various draft documents. The ICRP's framework for non-human species is being designed such that it is harmonized with the ICRP's proposed approach for the protection of humans. To this end, an agreed set of quantities and units, a set of reference dose models, reference dose-per-unit-intake data, and reference organisms are in the process of being developed. As a first step, the ICRP has proposed a limited number of Reference Animals and Plants (RAPs). A RAP is defined as *a hypothetical entity, with the assumed basic characteristics of a specific type of animal or plant, as described to the generality of the taxonomic level of family, with precisely defined anatomical,*

*physiological, and life-history properties, that can be used for the purposes of relating exposure to dose, and dose to effects for that type of living organism.* The ICRP reports are likely, as is the case for other areas of radiation protection, to form the seminal reference with the view that others can then develop more area- and situation-specific approaches to assess and manage risks to non-human species.

Larsson et al. (2002) provided an overview of the elements typical of an environmental assessment and management procedure in a general sense. The overall system is typical of the Ecological Risk Assessment (ERA) approach promoted by U.S. Environmental Protection Agency, based primarily on pathway-based assessment systems (Suter, 1993). The system is divided into five steps: 1) planning; 2) problem formulation (to guide further assessment, i.e. to define the assessment context); 3) assessment, using the appropriate methods according to the assessment context; 4) risk characterization; and 5) decision and management.

Elements from this generic approach were extracted in order to develop the EPIC assessment framework (Brown et al., 2003; Hosseini et al., 2005), specifically with exposures from ionizing radiation for Arctic environments in mind. The scope of the EPIC assessment methodology consisted primarily of an assessment methodology that would enable an assessor to quantify the probable effect of radiation exposure on selected biota following a defined release of radionuclides (Figure 5.1). Although aspects of planning, for example, com-

**Table 5.1.** Selected radionuclides with generalized characteristics. Source: adapted from Whicker and Schultz (1982).

| Radionuclide (periodic group) | Principal radioisotopes, $T_{1/2}$ | Sources | Nutrient analogues | Principal biospheric reservoirs | Environmental mobility |
|---|---|---|---|---|---|
| K (Ia) | $^{40}$K ($1.3 \times 10^9$ y) | Primordial | K | Lithosphere | High |
| Cs (Ia) | $^{134}$Cs (2.06 y) $^{137}$Cs (30 y) | Fission | K | Soil, sediments | High |
| Sr (IIa) | $^{89}$Sr (50.5 d) $^{90}$Sr (28.5 y) | Fission | Ca | Soil, biota | High |
| Tc (VIIa) | $^{99}$Tc ($2.13 \times 10^5$ y) | Fission | None | Biota, soil | High |
| Po (VIb) | $^{210}$Po (138 d) | $^{238}$U decay series | None | Soil, sediment | High |
| Pu (actinide series) | $^{238}$Pu (88 y) $^{239}$Pu ($2.4 \times 10^5$ y) $^{240}$Pu ($6.5 \times 10^3$ y) $^{241}$Pu (14.4 y) | Activation, neutron capture | None | Soil, sediment | Very low |
| Am (actinide series) | $^{241}$Am (432 y) | Activation, neutron capture, decay of $^{241}$Pu | None | Soil, sediment | Very low |
| I (VIIb) | $^{129}$I ($1.57 \times 10^7$ y) $^{131}$I (8.04 d) | Fission | I | Biota, soil | High |
| Ra (IIa) | $^{226}$Ra (1600 y) | $^{238}$U decay series | Ca | Lithosphere | Moderate |
| H (Ia) | $^3$H (12 y) | Cosmic, fission, activation | H | Hydrosphere (tritiated water) | High |
| C (IVb) | $^{14}$C (5600 y) | Cosmic, activation | C | Atmosphere ($CO_2$) | High |
| Th (actinide series) | $^{227}$Th (18.7 d) $^{228}$Th (1.9 y) $^{230}$Th ($7.7 \times 10^4$ y) $^{231}$Th (25.5 h) $^{232}$Th ($1.4 \times 10^{10}$ y) $^{234}$Th (24.1 d) | Natural, U and Th series decay chains | None | Lithosphere | Very low |
| U (actinide series) | $^{234}$U ($2.45 \times 10^5$ y) $^{235}$U ($7.04 \times 10^8$ y) $^{238}$U ($4.47 \times 10^9$ y) | Natural | None | Lithosphere | Low-moderate |

patibility check with underlying principles and international regulation, were deemed necessary to facilitate compatibility with legislative requirements at national levels, it was recognized that any system needed to be sufficiently generic to allow broad applicability. Thus, standards and limits were not integrated into the system, because these were likely to be imposed through national regulation. However, it was envisaged that the EPIC framework might be used to structure information in a way that could allow standards to be developed.

Drawing on the additional components and methodologies recently developed within the ERICA project, and using EPIC as a foundation and by way of example, subsequent sections will address the question of how a full environmental assessment including risk characterization might be conducted for the Arctic.

## 5.2. Problem formulation and pre-assessment considerations

The starting point for most assessments, following an extrinsic planning phase, will be problem formulation. This will clearly differ depending on many factors including, among others, the source of the radioactivity and constituent radionuclides, the type and nature of the contaminated environment or receiving medium, legislative requirements, the involvement

| Concentration increase with trophic level | Critical organ (vertebrates) | Biological half-life (mammals) |
| --- | --- | --- |
| Approaches 1 | Total body | Moderate (weeks) |
| Approaches 3 | Total body | Moderate (weeks to months) |
| < 1 | Bone | High (years) |
| < 1 | Gastrointestinal tract, lung | Low (days) |
| < 1 – 10 | Spleen, kidney, lung | Moderate (weeks) |
| < $10^{-2}$ | Bone, lung | High (years) |
| < $10^{-2}$ | | High (years) |
| Up to $10^3$ (thyroid/plants) | Thyroid | Moderate (weeks to months) |
| < 1 | Bone | High (years) |
| Approaches 1 | Total body | Low (days) |
| Approaches 1 | Total body | Low (days) |
| < $10^{-2}$ | Bone, lung | High (years) |
| < 1 | Gastrointestinal tract, kidney, lung | Moderate (months) |

of stakeholders, assessment criteria and a consideration of uncertainty within the assessment (Beresford et al., 2007a). Often, the problem formulation can be supported by a conceptual model, which might, for example, describe what is known about a discharging site, its geographical limits, radioactive substances of interest, potential pathways and receptors, and the likelihood of exposure, as well as any data gaps.

Because the EPIC framework was more generic in the sense that the assessment was not concerned with any one particular source of contamination, nor was it specifically designed to be directly relevant to any particular authorized discharge regime, many of the factors listed above become redundant. The problem formulation within the context of the EPIC framework is reduced to the three points: 1) the geographical setting – the spatial coverage for which the analysis is applicable; 2) the radionuclides to be considered within the assessment; and 3) reference Arctic biota that can be used to evaluate potential dose rates to biota in terrestrial, freshwater, and marine environments.

### 5.2.1. Radionuclides considered

The EPIC assessment methodology was limited in terms of radionuclides considered. The initial list of 13 radionuclides (see Table 5.1) was broadly representative of: 1) routine release scenarios from power plants and reprocessing facilities, 2) accidental releases, and 3) naturally-occurring or technologically-enhanced naturally-occurring (TENORM) radionuclides. The selected radionuclides covered a broad range of environmental mobility and biological uptake and so the system should be flexible enough to allow other radionuclides to be assessed with the provision of appropriate parameters.

### 5.2.2. Reference organisms

The EPIC framework identified a number of reference organisms, defined as *a series of entities that provides a basis for the estimation of the radiation dose rate to a range of organisms that are typical, or representative, of a contaminated environment. These estimates, in turn, would provide a basis for assessing the likelihood and degree of radiation effects.* For the sake of consistency, the EPIC reference organism lists might be considered as secondary reference organism sets that have some relation to the ICRP RAPs but which have been modified to account for site- or case-specific characteristics. Many selection criteria are available for the selection of a reference organism suite (see for example Pentreath and Woodhead, 2001). The criteria applied in EPIC were described by AMAP (2003).

Generic reference organism lists have been used as a basis for deriving appropriate environmental transfer data and selecting suitable target geometries/phantoms for dosimetric modeling. In this respect, it soon became apparent that the identification of actual species (or in some cases families or classes of organisms) representing each of the broadly defined groups would sometimes be helpful. This was the case for deriving food-chain model parameters where detailed information was often required, beyond a generic consideration, with respect to organism characteristics. It was also the case for geometry construction where quantitative information on size, shape and density were required and could be derived,

**Table 5·2.** Reference organisms and representative families/species for marine ecosystems.

| Reference organism | Representative species |
|---|---|
| Benthic bacteria | Not applicable |
| Macroalgae | *Fucus* spp. |
| Phytoplankton | Not applicable |
| Zooplankton | *Pandalus borealis* |
| Polychaetes | *Lumbrineris* spp. |
| Pelagic planktotrophic fish | *Boreogadus saida* (polar cod) [a] *Mallotus villosus* [b] *Clupea harengus* (herring)[c] |
| Pelagic carnivorous fish | *Gadus morhua* (cod) |
| Benthic fish | *Pleuronectes* spp. (e.g., *Pleuronectes platessa,* plaice) |
| Carnivorous mammal | 'Seals' (*Erignathus barbatus* and *Phoca hispida*) |
| Benthos-eating bird | *Somateria mollissima* |
| Fish egg | Not applicable |

[a] High and low Arctic; [b] low and sub-Arctic; [c] sub-Arctic.

simply and transparently, from a consideration of real flora and fauna. Examples of suitable representative species of selected reference organisms were subsequently chosen giving preference to species ubiquitous throughout the European Arctic and to the availability of appropriate data. Table 5·2 illustrates the set of reference organisms and their representative species for marine ecosystems, selected within EPIC.

The choice of a particular geographical region has fundamental implications for the selection of reference organisms. Although some EPIC reference organisms have a distribution that covers the entire Arctic, for example caribou/reindeer, this is not the case for all biota groups, for example European mink. Therefore, a selection based on a consideration of the entire Arctic might differ from one based on a limited region within the Arctic as exemplified by the EPIC framework. Because different groups of organisms may exhibit quite dissimilar transfer and dosimetric parameters (the latter reflecting size and shape) a pan-Arctic treatment of the reference organism selection process would be preferable. This has not, however, been attempted to date.

## 5.3. Exposure assessment

The derivation of exposure involves two basic steps: 1) estimation of the activity concentrations in biota and environmental media, and 2) estimation of the dose rates to biota. The basic equations and parameters used in these calculations are addressed in this section.

### 5.3.1. Radionuclide transfer to biota

In relation to analyses of the transfer of radionuclides from the point of release/input to the resultant activity concentration observed within reference flora and fauna, the main focus of EPIC was on biological uptake. This simplification was made with a view to the generic applicability of the system, assuming that reference media activity concentrations would be predictable or measurable. This originally removed the requirement for a consideration of environmental (physical) transport models (e.g., IAEA, 2001), although more recent work has shown how such components can be easily integrated into the approach (e.g., Brown et al., 2008).

In the absence of monitoring data, it is assumed that the assessor will have access to appropriate models to allow activity concentrations in abiotic compartments of the environment to be calculated. However, a good starting point for

the assessment can be the assumption of a specified concentration in the organisms' habitat, for example, a specified activity concentration per liter of water in the case of the aquatic environment and a specified activity concentration per kilogram of soil or unit deposition per square meter in terrestrial environments (with the exception of radiocarbon and tritium where measurements are related to air concentrations).

The approach used in EPIC and the more recent ERICA project for the derivation of biota activity concentrations from known media activity concentrations involves the use of equilibrium Concentration Ratios (CRs).

For terrestrial ecosystems the CR is defined as:

$$CR_{b,i} \ (dimensionless) = C_{b,i}/C_{soil,i} \tag{5.1}$$

where, $CR_{b,i}$ is the concentration ratio for reference organism $b$ and radionuclide $i$; $C_{b,i}$ is the activity concentration of radionuclide $i$ in the whole body of reference biota (Bq/kg ww); and $C_{soil}$ is the activity concentration of radionuclide $i$ in surface soil (Bq/kg dw).

For aquatic ecosystems the CR, also commonly known as the Concentration Factor (CF), is defined as:

$$CR_{b,i} \ (dimensionless \ or \ l \ kg^{-1}) = C_{b,i}/C_{aq} \tag{5.2}$$

where, $CR_{b,i}$ is the concentration ratio for reference organism $b$ and radionuclide $i$; $C_b$ is the activity concentration of radionuclide $i$ in the whole body of reference biota (Bq/kg ww); and $C_{aq}$ is the activity concentration of radionuclide $i$ in the aqueous phase (Bq/L or Bq/kg) – normally filtered water.

Three broad ecosystem categories (terrestrial, freshwater, and marine) were selected for further consideration within EPIC, and extensive review work was undertaken to collate all transfer relevant data available in the literature for these three categories (Beresford et al., 2003).

#### 5.3.1.1. Transfer in terrestrial environments

The majority of collated terrestrial transfer data in EPIC are for natural radionuclides from the U decay series and $^{137}$Cs and $^{90}$Sr from global fallout and, to a lesser extent, the Chernobyl accident. This means that for many reference organism-radionuclide combinations there are no reported data.

To derive soil-biota transfer values where measured data are missing, use can be made of allometric-based approaches. Beresford et al. (2003) used the approach suggested by

Higley et al. (2003), essentially a biokinetic-allometric type approach, in combination with suitable soil-plant transfer values for dietary components. These calculations were revised by including soil ingestion in order not to underestimate values for the radionuclides with low plant uptakes (Beresford et al., 2008a).

Estimates were made for animals of average age for each species. Predicted transfer values for Cs, U and Sr were generally comparable with the range of observed data, although predicted values for Ra were high compared with observed data. The inclusion of soil ingestion improved comparisons with the observed data for Pu, Am and Th.

For $^{14}$C, a specific activity approach was used to derive transfer parameters (Galeriu et al., 2003; Beresford et al., 2003). For tritium ($^{3}$H) an approach was developed (including limited Arctic-specific parameters) enabling (unlike other biota assessment frameworks) organically-bound and body-water $^{3}$H activity concentrations to be derived (Galeriu et al., 2003; Beresford et al., 2003). For both $^{14}$C and $^{3}$H, CR values represent the ratio of activity concentrations in biota to those in air (Bq/m$^3$).

### 5.3.1.2. Transfer in freshwater environments

Concentration ratio data for the Arctic freshwater environment that have been collated within EPIC are limited to a few species and a few radionuclides. For many radionuclides and organism types, other methodologies needed to be applied in the derivation of transfer information.

For the freshwater environment, EPIC used the dynamic model ECOMOD (as described by Sazykina, 2000), by way of demonstration, to simulate the behavior of selected radionuclides in freshwater food chains. For some radionuclides (Cs, Sr, P, Mn, Zn, I, Co) rates of uptake by fish are modeled using temperature-dependent parameters and ECOMOD includes some parameters derived from northern Russian lakes. These aspects of ECOMOD can therefore be said to be applicable to the Arctic. However, for other radionuclides and for invertebrates and aquatic plants, non-Arctic specific empirical transfer ratios must be used. Aquatic mammals and birds are not considered within the existing model.

Although these modeling approaches have only been applied to Arctic lakes they can theoretically be adapted to Arctic rivers in combination with an appropriate river transport model. It was apparent when EPIC was completed, that there were many gaps in the exposure assessment framework for freshwater ecosystems. For many of the reference organisms selected there was little or no information on transfer within Arctic systems. A similar situation existed at the time this assessment report was drafted (2008).

### 5.3.1.3. Transfer in marine environments

Site-specific radionuclide CR values for Arctic marine biota have been collated within EPIC for European Arctic sea areas including the Norwegian, Barents, White, Kara, and Greenland Seas (Beresford et al., 2003).

By comparing region-specific data sets with recommended generic values for marine CRs, the hypothesis that transfer to Arctic biota differs from that observed in temperate areas, was tested for $^{90}$Sr, $^{137}$Cs, $^{239,240}$Pu and $^{99}$Tc by Brown et al. (2004a). The authors noted that little could be concluded on the effect of Arctic environmental conditions upon radionuclide uptake due to the limited amount of data available and problems associated with compatibility of generic and Arctic data sets. Although, in most cases, well-founded recommendations could not be made with respect to the application of Arctic-specific CR values instead of generic values, for some radionuclides, distinct differences were apparent between region-specific and generic values. In the case of Sr, for example, Arctic CRs for fish and crustaceans appeared to be higher than corresponding world-generalized values. For Pu uptake to mollusks, Arctic values were distinctly below those recommended for generic application. Speculative reasons for some of these differences were provided by Brown et al. (2004a) and may, among others, relate to differences in the suites of species considered under any particular generic group and/or the physico-chemical form of radionuclides between temperate and Arctic regions. It was considered that the use of region-specific CR data might be justified in some cases. In the context of environmental impact assessments, the authors also noted that data pertaining to uptake to specific organs are very poorly characterized even though such data may be crucial in the derivation of robust exposure, i.e., dose-rate, estimates.

Where data are lacking on some of the parameters required for simulation of transfer, allometric relationships may provide surrogate values. The allometric approach is based on the observation that metabolic parameters, including basal metabolic rates, ingestion rates, biological half-lives etc., are proportional to the mass of an organism (West et al., 1997). Such models have been applied to Arctic environments for the purpose of simulating transfer. The application of a food-chain model, parameterized using allometrically-derived values where appropriate, has allowed the derivation of Cs and Pu CRs for several Arctic marine trophic levels (Figure 5·2). The preliminary estimates agreed well with
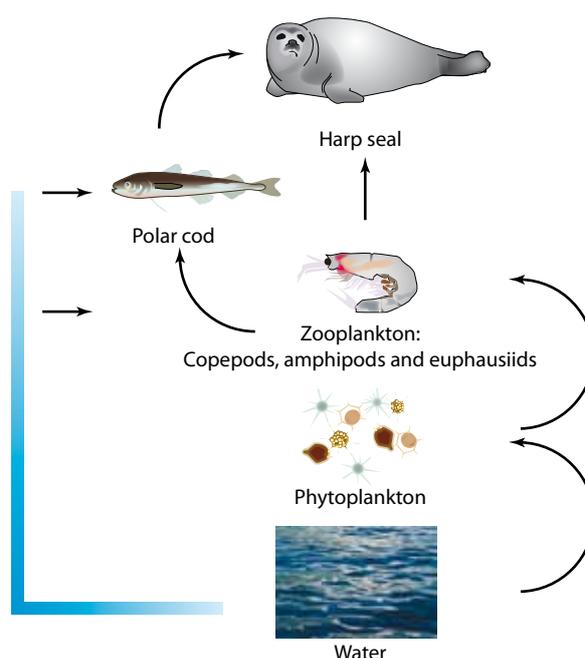


**Figure 5·2.** Food-chain model for harp seal in the Barents Sea. Source: simplified from Dommasnes et al. (2001).

**Table 5.3.** Examples of concentration ratios for terrestrial reference organisms. Source: extracted from Brown et al. (2003).

| Representative species | Bq/kg organism : Bq/kg soil (dw); Best estimate (range) | | | |
|---|---|---|---|---|
| | Cs | $^{90}$Sr | $^{210}$Po | $^{226}$Ra |
| Herbivorous mammals – all species | 7 | 2.9 | 4.2 | $4.8 \times 10^{-2}$ |
| | 0.01 – 76 | $(0.03 – 80) \times 10^{-2}$ | 0.4 – 14.3 | $(0.2 – 19.5) \times 10^{-2}$ |
| Herbivorous mammals – excluding reindeer | 1.0 | 1.1 | | $4.1 \times 10^{-2}$ |
| | 0.01 – 76 | $(0.05 – 62) \times 10^{-2}$ | – | $(0.2 – 19.5) \times 10^{-2}$ |
| Reindeer | 9.9 | 3.5 | 4.2 | $6.1 \times 10^{-2}$ |
| | 0.07 – 45 | 0.03 – 8.4 | 0.40 – 14 | $(0.31 – 16) \times 10^{-2}$ |
| Lemmings and voles | 3.5 | 1.9 | | $6.9 \times 10^{-2}$ |
| | 1.7 – 4.4 | | – | 0.01 – 0.2 |
| Carnivorous mammals – all species | 2.8 | 0.7 | 1.7 | $3.5 \times 10^{-2}$ |
| | 0.1 – 13 | 0.12 – 1.9 | 1.5 – 1.9 | $(0.4 – 9.6) \times 10^{-2}$ |
| Fox | 0.65 | 12.5[a] | | $4.0 \times 10^{-3}$ |
| | 0.1 – 1.7 | | – | (–) |
| Herbivorous bird – all species | 0.9 | Data for | – | $3.4 \times 10^{-2}$ |
| | 0.02 – 9 | *Lagopus* spp. only | | $(0.2 – 19.5) \times 10^{-2}$ |
| *Lagopus* spp. | 0.8 | $3.5 \times 10^{-2}$ | – | $2.5 \times 10^{-2}$ |
| | 0.02 – 3.2 | $(0.2 – 22) \times 10^{-2}$ | | $(0.9 – 5.1) \times 10^{-2}$ |
| Herbivorous bird – egg | $6.4 \times 10^{-2}$ [b] | – | – | – |
| | – | | | |

[a] Allometrically derived by Beresford et al. (2004); [b] estimated from dietary transfer to domestic hen eggs and CR values describing transfer to herbivorous bird whole-body.

**Table 5.4.** An overview of the ERICA marine concentration-ratio database. Source: Hosseini et al. (2008). The numbers given in the blank cells refer to the list of options presented in section 5.3.3 and indicate the method used to provide data for the given radionuclide-biota intersect. The tinted cells represent cases for which data are available and the color codes indicate the amount of data found.

| Element | Malg | Biv | Fish | Phy | Crus | Zoo | Worm | Mam | Anem | VasP | Bird | Rept |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Cs | | | | | | | | | | | | 1 |
| Pu | | | | | | | | | | 1 | | 1 |
| Co | | | | | | | | | | 1 | 2 | 2 |
| Sr | | | | | | | | | | 1 | 2 | 2 |
| Mn | | | | | | | | | 1 | | 2 | 2 |
| Po | | | | | | | | | 2 | 1 | 2 | 2 |
| Cd | | | | | | | | | 1 | 1 | 2 | 2 |
| Ag | | | | | | | | | 1 | 1 | 2 | 2 |
| Zr | | | | | | | 1 | 2 | | | 2 | 2 |
| Se | | | | | | 3 | | | | 1 | 2 | 2 |
| C | | | | | | | 1 | 1 | 1 | | 2 | 2 |
| Ni | | | | | | | | 2 | 1 | 1 | 2 | 2 |
| P | | | | | | | 1 | 1 | | | 2 | 2 |
| Pb | | | | | | | 1 | | 2 | 1 | 2 | 2 |
| I | | | | | | | 1 | | 1 | 1 | 2 | 2 |
| Ce | | | | | | 3 | 1 | 2 | | | 2 | 2 |
| S | | | | | | 3 | | | 1 | 1 | 2 | 2 |
| Am | | | | | | 3 | 1 | 5 | | 1 | 5 | 6 |
| Ra | | | | | | | 1 | 8 | 5 | 1 | 2 | 2 |
| Ru | | | | | | 3 | 1 | 2 | | 1 | 2 | 2 |
| U | | | | | 3 | 3 | 1 | 8 | | | 8 | 11 |
| Sb | | | | 3 | | | 1 | 2 | | 1 | 2 | 2 |
| Tc | | | | | | 3 | 1 | 8 | 2 | 1 | 2 | 2 |
| Th | | | | | 3 | | 1 | 8 | 5 | 1 | 8 | 11 |
| Cl | | | | 3 | | 3 | 1 | 1 | 1 | 1 | 2 | 2 |
| Np | | | 3 | | 3 | | 1 | 8 | 5 | 1 | 11 | 11 |
| Eu | | | | 5 | 3 | 3 | 1 | 2 | 5 | 1 | 2 | 2 |
| Cm | | | 3 | 5 | 5 | 5 | 5 | 5 | 5 | 1 | 5 | 6 |
| Nb | | | 5 | 3 | | 5 | 1 | 7 | 5 | 1 | 7 | 7 |
| Te | 3 | 3 | 3 | | 3 | 3 | 1 | 2 | 2 | 1 | 2 | 2 |
| H | 3 | 3 | 3 | 3 | 3 | 3 | 1 | 2 | 2 | 1 | 2 | 2 |

- ■ N ≤ 10
- ■ 10 < N < 20
- ■ N ≥ 20

Malg = Macroalgae
Biv = Bivalve molluscs
Fish = Fish
Phy = Phytoplankton

Crus = Crustacea
Zoo = Zooplankton
Worm = Polychaete worms
Mam = Mammals

Anem = Sea anemones/true corals
VasP = Vascular plants
Bird = (Wading) birds
Rept = Reptiles

| Pu | Th | U |
|---|---|---|
| $1.01 \times 10^{-3}$ | 0.64 | $1.8 \times 10^{-3}$ |
| (–) | $(0.2 - 47) \times 10^{-2}$ | $(0.1 - 2.8) \times 10^{-3}$ |
| | $7.7 \times 10^{-3}$ | |
| – | $(0.2 - 1.3) \times 10^{-2}$ | – |
| – | 0.4 | $9.4 \times 10^{-3}$ [a] |
| | 0.2 – 0.5 | |
| – | $7.7 \times 10^{-3}$ | $2.6 \times 10^{-3}$ |
| | $(0.2 - 1.3) \times 10^{-2}$ | $(2.4 - 2.8) \times 10^{-3}$ |
| – | $5.5 \times 10^{-3}$ | $7.1 \times 10^{-4}$ |
| | $(0.1 - 1) \times 10^{-2}$ | (–) |
| $1.7 \times 10^{-4}$ | – | – |
| | | |
| – | $3.9 \times 10^{-4}$ | $5 \times 10^{-4}$ |
| | $(3.1 - 5.4) \times 10^{-4}$ | $(4.1 - 6.8) \times 10^{-4}$ |
| – | $3.5 \times 10^{-4}$ | $4.1 \times 10^{-4}$ |
| | – | – |
| – | – | $2 \times 10^{-3}$ [b] |
| | | – |

empirical datasets and demonstrated that, in some cases, the application of an equilibrium CR was highly inappropriate (Brown et al., 2004a). In the context of the EPIC project, a biokinetic modeling approach, drawing on allometrically-derived parameters where appropriate was used in the derivation of some marine CR values (Brown et al., 2003).

The utility of using a biokinetic-allometric approach was explored by Gwynn et al. (2006) in the context of deriving a whole-body biological half-life for $^{210}$Po in adult ringed seals. The equation applied in this study is as follows:

$$C_s = \frac{\sum_1^n x_i . AE_{r,i} . IR . C_i}{k_{es}} \qquad (5.3)$$

where $x_i$ is the fraction of the diet associated with dietary component $i$ for seal; $AE_{r,i}$ is the assimilation efficiency (dimensionless) for dietary component $i$; $IR$ is the ingestion rate per unit mass of seal (kilogram fresh weight per day, per kilogram fresh weight); $C_i$ is the activity concentration in the $i$th dietary component (Bq/kg ww); $C_s$ is the activity concentration in (the 'whole body' of) seal (Bq/kg ww); $k_{es}$ is the effective loss rate from seal (per day) – from the excretion rate and physical decay of the radionuclide.

The initially calculated effective half-life of 562 days was greater than the physical half-life of $^{210}$Po (i.e., producing a negative biological half-life) suggesting that either the underlying methodology was at fault or that one or more of the parameters used in the model were not valid for the purposes of deriving biological half-lives for $^{210}$Po. One possible source of error was considered to arise from the use of a Po assimilation factor recommended for man (ICRP, 1979). Increasing the assumed assimilation efficiency of $^{210}$Po from 0.1 to 1 reduces the effective half-life to 56 days resulting in a whole-body biological half-life of 95 days. An additional consideration is the validity of using an equation which assumes that all $^{210}$Po originates from the diet of the seal; as in man $^{210}$Po mainly arises from the *in situ* decay of $^{210}$Pb (ICRP, 1979).

## 5.3.2. EPIC transfer look-up tables

The review and modeling conducted within EPIC provided mean CR values that may be applied in an exposure assessment. Look-up tables, with recommended radionuclide-specific CRs for reference organism groups in Arctic environments are provided by Brown et al. (2003) and Hosseini et al. (2005). Transfer data for the Arctic terrestrial environments are reported and discussed by Beresford et al. (2005). An example is presented in Table 5.3 for the terrestrial environment.

## 5.3.3. Identification and management of transfer data gaps

The approach presented above relies on comprehensive underpinning databases providing information on transfer through the use of CRs. Data gaps may arise because a reference organism (as applied in the EPIC and ERICA approach) does not adequately represent the organism(s) of particular interest within a given study or because the radioisotope to be assessed does not form part of the default databases. The flexibility available within the recently developed ERICA Tool (see Brown et al., 2008) allows these potential obstacles to be navigated in a reasonably robust manner.

Following developments within the ERICA project, a methodology was developed by Beresford et al. (2008a) for filling gaps in transfer data. According to this guidance, the options used to provide default CR values, when values could not be derived from the literature, are as follows:

1. Use an available CR value for an organism of similar taxonomy within that ecosystem for the radionuclide under assessment (preferred option).
2. Use an available CR value for a similar reference organism (preferred option).
3. Use CR values recommended in previous reviews or derive them from previously published reviews (preferred option).
4. Use specific activity models for $^3$H and $^{14}$C (preferred option).
5. Use an available CR value for the given reference organism for an element of similar biogeochemistry.
6. Use an available CR value for biogeochemically similar elements for organisms of similar taxonomy.
7. Use an available CR value for biogeochemically similar elements available for a similar reference organism.
8. Use allometric relationships, or other modeling approaches, to derive appropriate CRs.
9. Assume the highest available CR (least preferred option).
10. Use a reference organism from a different ecosystem (least preferred option).
11. Combination of approaches/options (least preferred option).

By applying this type of approach, it is usually possible to fill all data gaps for organism-radionuclide combinations. Table 5.4 shows an overview of the ERICA marine CR database and it is used as an example of applying this gap filling method. The numbers in the blank cells indicate the option used to provide data for a given radionuclide-biota combination and correspond to those listed above.

Of particular note with regard to the characterization of transfer data within the default transfer databases for ERICA, is that the derivation method is cited allowing the assessor to identify how each default value was selected. This allows the assessor to rapidly identify data of this type and the concomitant derivation method applied, during an assessment using the ERICA Tool (see Brown et al., 2008).

As illustrated by Table 5.4 the data coverage for temperate-generic marine ecosystems is poor in many cases. The data coverage for Arctic environments appears to be even more poorly characterized. This should be reflected in uncertainty introduced by extrapolating from temperate environments.

The data gap filling methods have not been applied in any structured way to the Arctic environment. Use of such methods would provide a more robust foundation for future assessments in the Arctic and would harmonize methods for the Arctic environment with those used for temperate or world-generic systems.

## 5.3.4. Some criticisms of the concentration-ratio approach

There is a general scientific concern that the assumption of instant equilibrium between the radioactivity in the medium and the biota (mediated by the concentration ratio) may in some cases result in poor predictions of the activity concentrations in organisms (see Coughtrey and Thorne, 1983; Brown et al., 2004a). This is because, in reality, organisms will retain radionuclides in their bodies and return them to the medium over timescales that can range from days to years. Furthermore, changing conditions in aquatic systems that may be initiated by, among others, a perturbed environmental regime, may lead to transfer of contaminants within food chains quite unlike the present situation. Understanding the influence of physical and chemical factors on uptake and depuration at various trophic levels may therefore prove indispensable for the requirement for making long-term robust prognoses with respect to contaminant behavior and fate.

Alternative methods to the use of concentration ratios have been developed that allow the derivation of biota activity concentrations under non-equilibrium conditions (e.g., Thomann, 1981). It is possible to develop relatively simple biokinetic models based on exchange between the medium and the organism with feedback, based on knowledge of the biological half-life of elimination and the CR, which allows deduction of exchange rates between the medium and the organism. Complex environment systems, including delayed excretion and retention by specific organs, require parameterization through rigorous, resource-intensive experimental work (Vives i Batlle et al., 2004; Wilson et al., 2007). This observation and that few data exist in the open literature to allow parameterization of biokinetic models for most organisms and radionuclides renders this type of approach currently challenging in many cases. However, this is a subject under development and significant progress has been made recently, for example, in deriving parametric values for biokinetic models through the application of allometric methods (Vives i Batlle et al., 2007a).

## 5.3.5. Absorbed dose rates

The basic components of information that are required to derive dose rates to organisms are: 1) dose conversion coefficients (DCCs) for mapping activity concentrations onto a dose rate and 2) occupancy factors defining the time spent by biota in various surroundings within their habitats for the parameterization of external dose calculations.

Many models exist for deriving absorbed dose-rates to individual organisms, including the analyses and solution of dose-distribution functions, conservative approaches (whereby all radiation emitted by radionuclides within the organism are absorbed) and Monte Carlo methodologies. Examples of dose calculation methodologies include those by the IAEA (1979), Copplestone et al. (2001), USDOE, (2002), and Pröhl et al. (2003).

In the case of internal irradiation arising from the presence of γ-emitting radionuclides, calculations are often required to derive absorbed fractions, that is, the quotient of the energy emitted by the target to the energy absorbed by the source, for each of the characteristic photons emitted. Absorbed fractions may also be used in the derivation of external dose-rates through simplifying assumptions relating to density differences and the source geometry (e.g., Vives i Batlle et al., 2004a). Alternatively, more complex models can be applied when media density differences are important, through the application of photon and electron transport simulations using Monte Carlo methods (e.g., Taranenko et al., 2004).

### 5.3.5.1. EPIC methodology for deriving dose conversion coefficients

The EPIC approach used reference organisms as the basis for further dosimetric modeling. The actual dimensions of the organisms were based, in most cases, on the adult form of representative organisms (see Table 5.5) and were specified in the look-up tables presented by Golikov and Brown (2003). For the derivation of DCCs, ellipsoids were used to represent the various geometric forms of representative plants and animals.

Owing to the complexity of the processes involved and the enormous variability of organisms and their natural habitats, it was not possible to derive external DCCs for all possible exposure conditions. Therefore, typical exposure situations appropriate to and based around the geometries for reference organisms were selected for detailed consideration. These are:

- For DCCs pertaining to species living *in the soil*, two source descriptions were assumed: 1) a uniformly contaminated volume source for natural radionuclides and 2) a planar isotropic source, located at a depth equivalent of 0.5 g/cm² in the soil (this represents a thin surface layer contamination selected to represent a period shortly after a deposition episode) for artificial radionuclides.
- For DCCs pertaining to species living *on the ground*, two source descriptions were assumed: 1) a semi-infinite volume source for natural radionuclides and 2) a planar isotropic source located at a depth equivalent of 0.5 g/cm² in the soil for artificial radionuclides.

- For DCCs pertaining to aquatic species at the sediment/ water interface, two source descriptions were assumed: 1) a volume source with a depth of 5 cm for artificial radionuclides (a depth of 5 cm was arbitrarily selected to represent common artificial radionuclide profiles – bioturbation and post depositional migration of radionuclides often lead to the rapid development of a finite layer of contamination) and 2) a semi-infinite volume source for natural radionuclides.

The method for deriving absorbed doses as applied in the EPIC approach was based on an approximation defining the dose distribution of radiation within an organism's body. This distribution can be defined using two functions:

1. The dose attenuation function, which describes the dose at any point along the path length for radiation travelling through matter. Equations can be solved using exact numerical methods.
2. The chord distribution function, which describes many possible path lengths within the body. This can be calculated using a Monte Carlo methodology for each specific geometry.

External doses to organisms from radionuclides present in the soil or the water column were calculated using a variant of the simple formula for uniformly contaminated isotropic infinite absorbing medium. This approach approximates the dose rate to an organism immersed in an infinite contaminated medium but neglects density differences between the organism and the medium. Furthermore, it allows for self-shielding by the organism itself, and averages the dose rate throughout the volume of the organism. This approach has been used to calculate the external dose from β- and γ-radiation for organisms buried in soil or free swimming in the water column; the relevant activity concentrations being those in the soil or water media as appropriate.

The estimation of external exposures at the interface of environments with different densities is more complex than cases pertaining to infinite, uniformly-contaminated environments. A two-step method was used for the calculation of dose rates at the interfaces. First, the kerma in a specified location (above the soil/air interface, in soil at the given depth) was derived. Second, the ratio of the dose in an organism and the kerma was calculated for the different energies characteristic of different radionuclides. Further details on these methods are provided elsewhere (Golikov and Brown, 2003).

The full set of DCCs derived in the EPIC project was reported by Brown et al. (2003) and Hosseini et al. (2005). The dose-rate derivation methodologies developed in the EPIC project were subject to analyses through intercomparison with other dosimetric modeling methods by Vives i Batlle et al. (2007b).

### 5.3.5.2. ERICA's dosimetric approach

The dosimetric methods developed during the EPIC project have been superseded by the calculation tools that are now available through the ERICA project. Within the ERICA tool (Brown et al., 2008) it is possible to derive user-defined geometries representing a given plant or animal and so to

**Table 5.5.** Summary of natural background dose rates for various organisms groups. Source: adapted from Sazykina et al. (2003) and Brown et al. (2003).

| Ecosystem | Organism | Dose rate, µGy/d |
|---|---|---|
| Marine | Phytoplankton [a] | 0.5 − 2.1 |
| | Zooplankton [a] | 0.6 − 4.1 |
| | Crustaceans [a] | 2.7 − 14 |
| | Molluscs [a] | 2.7 − 13 |
| | Macrophytes [a] | 1.7 − 12 |
| | (Benthic) Fish [a] | 1.3 − 10 |
| | Waterfowl [a] | 0.5 − 1.6 |
| | Seal [b] | ~ 4.5 |
| Freshwater | Fish | 1.4 − 2.2 |
| Terrestrial | Generic vertebrate [c] | ~ 3.2 |

[a] Derived for the Kara Sea – it is assumed that phytoplankton, zooplankton and waterfowl receive all external irradiation from the water column whereas crustaceans, molluscs, macrophytes and benthic fish receive all external irradiation from sediment; [b] from Gwynn et al. (2006) – internal and external irradiation from naturally occurring $^{210}$Po and $^{40}$K; [c] generic terrestrial vertebrate in a temperate environment (Whicker and Shultz, 1982).
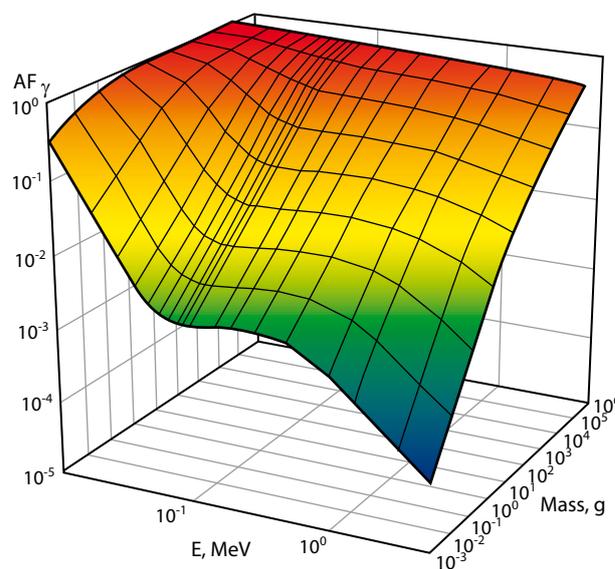


**Figure 5.3.** Absorbed fractions as a function of mass and photon energy (MeV)

derive radionuclide-specific DCCs. These are the methods that have currently been adopted by the ICRP in the derivation of DCCs specifically for RAPs. For aquatic environments, the absorbed fractions for photon and electron sources uniformly distributed in soft-tissue spheres and ellipsoids immersed in infinite aquatic medium have been calculated by Monte Carlo simulation. The calculations covered an energy range of 10 keV to 5 MeV, a mass range from $10^{-6}$ kg to $10^3$ kg and shapes from sphere to ellipsoids with varying degree of non-sphericity (Ulanovsky and Pröhl, 2006). From the computed absorbed fractions, a set of re-scaling factors (i.e., the ratio of the absorbed fraction for a non-spherical body to that of an equivalent spherical geometry) have been derived. A function has been found relating re-scaling factors to the non-sphericity of the organism's body. This allows interpolation between points and absorbed fractions to be defined for user-defined organism size and shape within cer-

tain limits. An example of the types of interpolation surface formed from a collation of this type of information is provided in Figure 5.3.

For terrestrial reference organisms, estimation of external exposures is more complex than for organisms in the aquatic environment. Soil, air and organic matter differ considerably in composition and density, which means that radiation transport through matter, cannot be adequately taken into account by the application of analytical solutions. Therefore, the derivation of DCCs is based on radiation transport simulated for mono-energetic photons using Monte Carlo techniques (see Taranenko et al., 2004). Generalized, representative cases as defined by energy, contaminated media and organism sizes were selected for detailed consideration. Exposure conditions, for which detailed calculations are not available, can then be deduced by an interpolation between these cases. From the calculations for mono-energetic radiation sources, nuclide-specific DCCs are derived for external and internal exposure, taking into account the type of radiation as well as energy and intensity of the radiation emission.

However, these newly available dosimetric methods have not been applied for the purpose of Arctic radiation protection frameworks. Little effort would be required to run the dosimetric module for the lists of reference organisms generated in the EPIC project in order to bring the DCC values into line with those used by ERICA and moreover by the ICRP.

## 5.3.6. Dose rate calculation

The whole body absorbed dose rate is used as a measure of the reference organism's exposure to ionizing radiation, normally expressed in units of μGy/hr, and is the sum of internal and external absorbed dose rates. (Originally, dose rates in the EPIC project were reported in units of Gy/y. More recent approaches, such as the ERICA project, have tended to use units of μGy/hr.)

The practical application of the system of dosimetry based around the absorbed dose (in units of Gy) forces consideration of the empirical observation that the same absorbed dose of differing radiations can produce differing degrees of effect in the same biological endpoint. That is, the radiations can differ in their *qualitative effect*. For example, there is a very substantial body of experimental evidence to indicate that the absorbed dose of high linear energy transfer (LET) radiation (α-particles) required to produce a given biological effect is less than that of low LET radiation (β-particles and γ-rays) – the relative biological effectiveness (RBE) phenomenon. For human radiological protection practice, this phenomenon is taken into account by applying dimensionless radiation weighting factors ($w_r$) to the absorbed doses from the different radiations, and summing, to give a quantity termed the *equivalent dose*. It should be emphasized, however, that values of $w_r$ defined for the purpose of human radiation protection cannot be applied without reservation to other organisms and biological endpoints.

It may nonetheless be appropriate to introduce radiation weighting factors to take account of the differing biological effectiveness of different types of ionizing radiation for biota. It has been noted that the final choice of radiation weighting factor for α-particles will depend on the selection of reference organism, endpoint and dose (or dose-rate) range (Tra-

cy and Thomas, 2002). This means that consensus on appropriate radiation weighting factors is not easily attained. It has therefore been considered appropriate by some workers (Pröhl et al., 2003) that calculations of absorbed dose should be split into low LET and high LET components to facilitate the incorporation of a radiation weighting factor. This is also consistent with the upper bound on the range of variation reported by Chambers et al. (2006) for a-radiation weighting factors in relation to population relevant deterministic endpoints (mainly mortality). For this reason, the radiation emission types for each radionuclide have been split into the categories: α, low-β and β,γ.

The weighted DCCs (split into the categories internal and external) for a given radionuclide and reference organism become:

$$\left[ DCC_{i,low\beta}^{j} \right]_w = DCC_{i,low\beta}^{j} * w_{low\beta} \tag{5.4}$$

$$\left[ DCC_{i,a}^{j} \right]_w = DCC_{i,a}^{j} * w_a \tag{5.5}$$

$$\left[ DCC_{i,Total}^{j} \right]_w = \left[ DCC_{i,low\beta}^{j} \right]_w + \left[ DCC_{i,a}^{j} \right]_w + DCC_{i,\beta\gamma}^{j} \tag{5.6}$$

where, $\left[ DCC_{i,low\beta}^{j} \right]_w$ ; $\left[ DCC_{i,a}^{j} \right]_w$ and $\left[ DCC_{i,Total}^{j} \right]_w$ are 'weighted' DCCs for low β, α and all radiation types respectively. They are specific to radionuclide *i* and reference organism *j*. $w_\beta$ and $w_a$ are radiation weighting factors.is the DCC for β, γ radiation for radionuclide *i* and reference organism *j*.

In view of the way in which DCCs were presented by Golikov and Brown (2003) in terms of the components of α-, β- and γ-radiation, a choice was made within the EPIC approach not to apply a weighting factor for low β in most cases. However, ^{3}H is known to emit a large component of low β-radiation and earlier studies (e.g., Straume and Carsten, 1993) showed that a radiation weighting factor in excess of unity might be appropriate for this particular radionuclide. Within the EPIC approach, therefore, weighted DCCs were derived using provisional weighting factors of 3 for [3]H (all other β-emitters were assigned a radiation weighting factor of 1) and 10 for α-radiation. In contrast other approaches, e.g., FASSET and ERICA, apply a default of 3 for the low-β component of the dose rate in all cases.

The external dose rate, averaged over different habitats, can be determined by the following equation:

$$\dot{D}_{int}^{j} = \sum_i C_i^j * DCC_{int,i}^{j} \tag{5.7}$$

where, $C_{zi}^{ref}$ is the average activity concentration of radionuclide *i* in the reference media of a given habitat *z* (Bq/kg soil or sediment, or Bq/L water), $DCC_{ext,zi}^{j}$ is the DCC for external exposure defined as the ratio between the average activity concentration of radionuclide *i* in the reference medium corresponding to the habitat *z* and the dose rate to organism *j* (μGy/h per Bq/kg or Bq/m³), $v_z$ is the occupancy factor, that is, the fraction of the time that the organism *j* expends in habitat *z*.

The internal dose rate (for biota in both aquatic and terrestrial environments) can be derived from the activity concentration in the selected reference organism using the following equation:

$$\dot{D}^{j}_{ext} = \sum_{z} v_{z} \sum_{i} C^{ref}_{zi} * DCC^{j}_{ext,zi} \qquad (5.8)$$

where, $C^{j}_{i}$ is the average activity concentration of radionuclide $i$ in reference organism $j$ (Bq/kg ww), $DCC^{j}_{int,i}$ is the radionuclide-specific DCC for internal exposure defined as the ratio between the average activity concentration of radionuclide $i$ in organism $j$ and the dose rate to the organism (µGy/h per Bq/kg ww).

Using similar methodologies, the dose rates derived in the study by Gwynn et al. (2006) for an adult ringed seal were dominated by the internal components of the naturally-occurring radionuclides $^{210}$Po and $^{40}$K, while the dose contribution from anthropogenic radionuclides was small.

## 5.4. Placing the exposure estimates in context

Arguably, two points of reference may be used for the purpose of assessing the potential consequences of exposures to radiation on non-human biota. These are natural background dose rates and dose rates known to have specific biological effects on individual organisms (Pentreath, 2002).

The ICRP suggests bands of derived consideration levels for reference fauna and flora (ICRP, 2003). These can be compiled by combining information on logarithmic bands of dose rates relative to normal natural background dose rates, simply as a means of presentation, plus information on dose rates that may have an adverse effect on reproductive success, or result in early mortality (or cause morbidity), or are likely to result in scorable DNA damage for such organisms (ICRP, 2003). Such a banding could be essentially on the same basis as proposals made for humans (ICRP, 2001). By way of example, and still a debatable issue, dose rates that were only fractions of their background might be considered to be trivial or of low concern; those within the normal background range might need to be considered carefully; and those that were one, two, three or more orders of magnitude greater than background would be of increasingly serious concern because of their known adverse effects on individual fauna and flora (Pentreath, 2002).

### 5.4.1. Background dose rates in Arctic environments

In the Arctic, as everywhere on Earth, terrestrial and aquatic organisms are exposed to natural sources of ionizing radiation, including cosmic rays, radionuclides produced by cosmic ray interactions in the atmosphere, and radiation from naturally-occurring radionuclides, which are ubiquitous in all living and non-living components of the biosphere (Whicker and Schultz, 1982). Typical dose rates of natural background exposure for different types of organisms in the Arctic were discussed by Sazykina et al. (2003). These dose rates were derived using data on the activity concentrations of natural radionuclides in the Arctic aquatic ecosystems for several reference organism groups and representative species. The doses were estimated by the methods described in earlier studies (IAEA, 1976, 1979; Kryshev and Sazykina, 1990, 1995; Kryshev et al., 2001, 2002), taking into account geo-

metrical characteristics of organisms and ionizing radiation sources (Table 5.5). Terrestrial background radiation dose rates have been particularly poorly characterized. Within the EPIC project, typical annual doses to terrestrial vertebrates under generic conditions were obtained from Whicker and Shultz (1982) in order to provide an indication of dose rate in the Arctic.

Some new information on terrestrial background dose rates has become available through the work of Beresford et al. (2008b) but the dose rates derived are based primarily on information from temperate environments. Whether these values are directly relevant for Arctic systems is debatable. Brown et al. (2004b) addressed the issue of assessing dose rates arising from naturally-occurring radionuclides for European aquatic environments. The majority of the calculated absorbed dose, for both marine and freshwater organisms was seen to arise from internally incorporated α-emitters, with $^{210}$Po and $^{226}$Ra being the major contributors. Calculated doses were somewhat higher for freshwater compared to marine organisms, and the range of doses was also much greater in the latter. This was believed to reflect much greater variability of radionuclide activity concentrations in freshwaters as compared to seawater, as well as variability or uncertainty in concentration ratio values (as applied for freshwaters only). The work revealed a number of substantial gaps in published empirical data, especially for European aquatic environments. The situation in terms of gaps in characterizing Arctic freshwater environments is arguably more serious (see Table 5.5).

### 5.4.2. Effects of radiation within the Arctic

Climatic conditions in the Arctic are, in general, unfavorable for organic life. Low temperatures and extreme seasonal variations in light are some of the physical and chemical characteristics which cause environmental stress to organisms in the Arctic and make them potentially more vulnerable to contaminants (AMAP, 1998).

The following observations were made by Sazykina et al. (2003) upon considering the effects of radiation under Arctic conditions:

- Severe climatic conditions are factors of natural environmental stress, restricting the number of biological species which are able to survive in the Arctic. Low biodiversity is a negative ecological factor associated with the low capacity of Arctic ecosystems to adapt in the case of any environmental changes.
- The development of radiation effects in the Arctic poikilothermic (or hibernating) organisms is expected to occur more slowly, because of low environmental temperatures. On the other hand, repair of radiation damage in cells and tissues is not effective at very low temperatures. Lesions in the cooled (poikilothermic or hibernating) organisms are latent. However, if organisms become warm, lesions are rapidly revealed. As a result, radiation effects may not appear during the winter period, but may manifest themselves intensively during the warm season.
- Development of embryos and young poikilothermic organisms in the Arctic occurs slowly; for example, the development of roe of some Arctic fish species takes more

than 200 days, whereas in temperate regimes fish eggs usually develop over 8 to 10 days. At the same dose rate, Arctic fish eggs receive much higher doses during the radiosensitive stages of ontogenesis compared to fish eggs in the temperate climate. Long-lived species, depending on their reproductive strategy, may be more vulnerable to radioactivity because of the potential for integration of dose in the reproductive organs with time.

- High concentrations of lipids in Arctic animals may be expected to increase their radiosensitivity, because chemical products of lipidoperoxidation produced by irradiation are toxic for organisms.
- Long-distance migrations of Arctic animals, in general, are favorable for survival, because animals do not stay within any contaminated local area for long periods; thus accumulated doses to migratory animals are expected to be lower than those for sedentary organisms. Furthermore, the thick skin of Arctic mammals can protect effectively not only from cold and ice, but also from external α- and β-radiation.

### 5.4.2.1. Compilation of data on dose-effect relationships

Data compilation within the EPIC project focused on the effects of chronic radiation exposure at dose rates well below those that are known to cause mortality of organisms (see Sazykina et al., 2003). And, from the wide variety of radiation effects reported in the open literature, emphasis was placed on those which were important for the survival and reproduction of organisms in the wild. Furthermore, information was organized in a form that would facilitate the development of appropriate Arctic dose limits, providing a scientific basis for regulations in the radiation protection of the environment. To this end, the construction of a preliminary scale of the severity of radiation effects at different levels of chronic exposure was considered useful to aid decision-making. Data concerning dose-effect relationships for radiation effects in reference (or related) Arctic biota available from Russian and other former Soviet Union sources were collated. The compiled data were focused on effects in radiosensitive species in terrestrial and aquatic ecosystems, such as mammals, fish, and sensitive groups of plants (e.g., pines). Less attention was given to radioresistant species (e.g., lichen and moss). Effects data were organized under 'umbrella' endpoint categories, namely:

- morbidity (e.g., worsening of physiological characteristics of organisms; effects on immune system, blood system, nervous system);
- reproduction (negative changes in fertility and fecundity, resulting in reduced reproductive success);
- mortality (shortening of lifetime as a result of combined effects on different organs and tissues of the organism);
- cytogenetic effects (radiation effects at the cellular level);
- ecological effects (changes in biodiversity, ecological successions, predator-prey relationships);
- stimulation effects (radiation hormesis, low dose stimulation effects); and
- adaptation effects (responsive adjustments of organisms to the conditions of chronic irradiation).

The biological endpoint 'reproductive success' was of particular interest because this tends to be the most radiosensitive endpoint, one that ultimately influences the viability of a defined population and relates the assessment to the underlying principle of sustainability.

In order to underpin the approach outlined above, a database in Microsoft ©EXCEL was constructed – the so-called EPIC database on radiation effects. The database includes data on radiation effects in wild organisms, which were observed from field studies in the northern areas of Russia, including sub-Arctic regions. These areas included the East Ural radioactive trace (following the Kyshtym accident – a thermal explosion in a high-level radioactive waste tank at the Mayak PA in 1957), local areas with enhanced levels of natural radioactivity in the Komi Republic, and some others. Data on radiation effects in the Low Arctic referred mostly to cold-water fish. The database also included data from laboratory experiments with boreal organisms, and data from several other relevant experimental studies. Considering the great importance of the radiobiological studies of wildlife in the Chernobyl-contaminated areas, these data were also included in the EPIC database. In total, the EPIC database contained about 1600 records from 435 research papers and books. The information covers a very wide range of radiation dose rates to wild flora and fauna: from below $10^{-5}$ Gy/d to more than 1 Gy/d. Dose reconstructions were made, in some cases, by the authors of the database using data on levels of radioactive contamination in the organism/environment and standard dose derivation methodologies (IAEA, 1976, 1979; Kryshev and Sazykina, 1990; Kryshev et al., 2002).

Parts of the database have been published and discussed in detail (see Sazykina and Kryshev, 2003, 2006). For example, the approximate threshold levels of chronic exposure above which specific types of effects can be detected in fish are, according to Sazykina and Kryshev (2003):

- dose rates of 0.5 mGy/d to 1 mGy/d (ca. 21 μGy/h to 42 μGy/h) with accumulated doses above 0.05 Gy to 0.2 Gy are threshold levels for the appearance of the first negative changes in fish blood, and early signs of decrease in the immune system; at lower dose rates (< 0.5 mGy/d) the organisms seem able to adapt provisionally to radiation with a gradual restoration of health parameters;
- dose rates of 2 mGy/d to 5 mGy/d with accumulated doses above 1.5 Gy are threshold levels for the appearance of negative effects on the reproduction system; and
- dose rates of 5 mGy/d to 10 mGy/d of chronic lifetime exposure lead to life shortening of adult fish.

These observations appear to be in apparent contradiction to earlier, internationally debated considerations on this subject where it was, for example, concluded that *in the aquatic environment it would appear that limiting chronic dose rates to 10 mGy/d or less to the maximally exposed individuals in a population would provide adequate protection for the population* (IAEA, 1992). Nonetheless, the implications of the occurrence of morbidity effects in individuals, such as negative changes in blood and detrimental effects on the immune system, for the integrity of the population as a whole is unclear rendering a direct comparison between the two sets of conclusions far from straightforward.

Although the EPIC database mostly includes information on plants and animals that are not indigenous to the Arctic, it still constitutes an important source of reference in the evaluation of any environmental impact arising from radiation exposure in the Arctic. However, the extrapolation of dose-effects data from predominately temperate conditions to the severe conditions associated with the Arctic may require some caution and, in the case of deriving limits, the application of appropriate safety factors.

Many of the data on radiation effects generated in the EPIC project have been included within the ERICA effects database, FREDERICA (Copplestone et al., 2008).

### 5.4.2.2. Effects and climate change

Increase in temperature, as follows from the Arrenius's Law, increases the rates of biochemical reactions within the biological range of temperatures. Temperature affects both the reactions of radiotoxins with biomolecules, as well as diffusion of toxic compounds to neighboring tissues. Lowering the temperature of biological tissues/organisms helps to prevent the development of radiation effects. For example, the survival time of fish (*Carassius* sp.) kept at different temperatures between 3 °C and 25 °C after an exposure of 18 Gy was much longer at low temperatures and seemed to follow a decrease in oxygen consumption; i.e., time of survival was inversely proportional to metabolic rate (Keiling et al., 1958). Development of radiation effects in Arctic poikilothermic (or hibernating) organisms might therefore be expected to occur more slowly under low ambient temperatures. However, repair of radiation damage in cells and tissues is not effective at very low temperatures. At temperatures of about 2 °C to 4 °C, the reparation process is practically inoperative (Kudryashov and Berenfeld, 1982; Kuzin, 1986; Mettler and Upton, 1995). Lesions in 'cooled' organisms (e.g., poikilothermic or hibernating animals) are effectively latent, whereas, in organisms that become warm, lesions may be rapidly revealed.

### 5.4.2.3. Possible multi-stressor effects

Research on the combined effects of environmental pollutants and radiation for circumstances of specific relevance to the Arctic are rare. The most authoritative overview to date of combined effects on humans has been published by UNSCEAR (2000b). This concluded that there is little evidence for combined effects of radiation and other environmental pollutants at the levels typically encountered in the environment, based on epidemiological data or studies. At the same time, there exist studies of interactions which indicate that, at least at high exposure, the action of one agent can be influenced by simultaneous exposure to other agents (UNSCEAR, 2000b). This influence can be positive (synergistic), negative (antagonistic) or zero (concentration-additive).

Even though the primary molecular and cellular effects of various persistent organic pollutants (POPs), heavy metals and radionuclides are often diverse, their toxicity may be compared using suitable (umbrella) endpoints. The umbrella endpoints used in EPIC were described in section 5.4.2.1. A similar, but more detailed approach is being developed for POPs using, in addition to mortality and reproduction

effects, biological markers based on subtle, low dose effects (e.g., on liver enzymes). Carcinogenic effects are also considered – stating whether a POP is mutagenic or functions as a tumor promoter. In the Arctic, the major concern about POPs concerns long-term chronic exposures, i.e., organisms being exposed to low levels over their entire lifetime. In this context, the main effects of concern are those that may affect reproduction and survival at both the individual and population level. An overview of toxic properties of important POPs was undertaken by AMAP (2004b). Endpoints such as mortality and effects on reproduction are also important in connection with heavy metals (AMAP, 1998).

According to UNSCEAR (2000b), to address and assess potential combined effects the following parameters must be considered first: the mode of action of the agent (genotoxic or non-genotoxic); the shape of the dose-effect relationship for single-agent effects; the dose or concentration involved (low or high); the type of exposure (chronic or acute); and the sequence and time interval between exposures (simultaneous, before or after radiation exposure). Furthermore, to achieve a well-balanced conclusion on the combined actions of two agents, the dose-effect relationship of the combined exposure should be known and analyzed using a model in which the interactions can be consistently and quantitatively defined. The majority of studies on combined effects, including those with radiation, do not meet these conditions (UNSCEAR, 2000b).

Even though interactions between non-radioactive contaminants and radionuclides/radiation exposure have not been extensively studied for non-human biota, two separate, but connected, general influences may be distinguished: 1) effects of co-exposure to non-radioactive contaminants on accumulation kinetics and internal tissue distribution of radionuclides; and 2) the possible modifying influence of co-contaminants on the biological effects induced by the exposure to ionizing radiation (Woodhead and Zinger, 2003). To identify these influences, systematic investigations of combined effects are needed, particularly at low levels of exposure.

## 5.4.3. Criteria and standards

### 5.4.3.1. General

There have been several review publications on radiobiological effects in the environment (IAEA, 1976, 1992; Blaylock and Trabalka, 1978; NCRP, 1991; Polikarpov, 1977, 1998; Turner, 1975; Woodhead, 1984; UNSCEAR, 1996). Owing to a lack of information on lower intensity, chronic irradiation regimes, the existing reviews draw heavily on studies of radiation effects from acute exposure at high doses. The applicability of data pertaining to acute exposures to the conditions of interest under, for example, authorized discharges of radionuclides to the environment is debatable. In human radiological protection, a dose and dose-rate effectiveness factor (DDREF) is used to project cancer risk determined at high doses and high dose rates to the risks that would apply at low doses and low dose rates. However, a similar type of factor is not available for environmental assessments nor can this value for human radiological protection be used as a substitute, because the endpoints of concern are different, i.e., for envi-

**Table 5·6.** Scale mapping absorbed dose-rates onto effect. Source: Sazykina et al. (2003).

| Absorbed dose rate, Gy/d | Effect |
|---|---|
| $10^{-6}$ to $10^{-5}$ | Natural radiation background for Arctic/northern organisms. |
| $10^{-4}$ to $5 \times 10^{-4}$ | Minor cytogenetic effects. Stimulation of the most sensitive species. |
| $5 \times 10^{-4}$ to $1 \times 10^{-3}$ | Threshold for minor effects on morbidity in sensitive vertebrate animals. |
| $2 \times 10^{-3}$ to $5 \times 10^{-3}$ | Threshold for effects on reproductive organs of vertebrate animals, decrease in embryo survival. |
| $5 \times 10^{-3}$ to $10^{-2}$ | Threshold for life-shortening of vertebrate animals. Threshold for effects in invertebrate animals. Threshold for effects on growth of coniferous plants. |
| $10^{-2}$ to $10^{-1}$ | Life-shortening of vertebrate animals; chronic radiation sickness. Considerable damage to coniferous plants. |
| $10^{-1}$ to $1$ | Acute radiation sickness of vertebrate animals. Death of coniferous plants. Considerable damage to eggs and larva of invertebrate animals. |
| $> 1$ | Acute radiation sickness of vertebrate animals; lethal dose received within several days. Increased mortality of eggs and larva of invertebrate animals. Death of coniferous plants, damage to deciduous plants. |

ronmental protection there is likely to be most concern about endpoints that have direct relevance to population viability, such as fertility.

In the 1990s, international reviews of radiation effects on flora and fauna were published by IAEA (1992) and UNSCEAR (1996). The conclusions of the IAEA and UNSCEAR reports specified the ranges of chronic dose rates, which are of concern in the environmental protection of flora and fauna. None of these dose rate levels were intended as recommendations for radiation protection criteria, although they clearly could have implications for the development of such criteria and standards.

Dose limits have been applied in other situations as exemplified by the approach advocated by the US DOE (2002). The limits used by the US DOE were established earlier based on the findings of many reviews considering the effects of ionizing radiation on flora and fauna (e.g., NCRP, 1991; IAEA, 1992) and relate to the protection of populations of wild organisms. In their graded approach the US DOE (2002) used a dose limit of 10 mGy/d ($\approx$ 400 µGy/h) for indigenous aquatic animals and benchmarks of 400 µGy/h and 40 µGy/h for terrestrial plants and terrestrial animals, respectively.

A large database on radiation effects on biota has been compiled within the EC Project FASSET, based mainly on available English-language publications. In the FASSET (the so-called 'FRED') database, again, the majority of radiation effects data is for acute dose exposures (Woodhead and Zinger, 2003). In general terms, it seemed that although there might be minor effects at lower dose rates in sensitive species, the dose rates for statistically significant effects in most studies were about 0.1 mGy/h; the responses were then observed to increase progressively with increasing dose rate and usually became very clear at dose-rates > 1 mGy/h when these were delivered over a large fraction of the lifespan (Real et al., 2004).

Although general conclusions and, in some few cases, dose-limits exist in relation to environmental protection from ionizing radiation, their direct applicability within the context of the Arctic is limited. This is mainly because there are reasons to believe that Arctic climatic conditions influence the expression of radiation-induced effects and, furthermore, that Arctic ecosystems are potentially more vulnerable to contaminants than organisms in other European climatic regions. The dose limits/benchmarks derived for temperate environments may, therefore, be unsuitable for direct application to the Arctic.

Furthermore, these values relate to the protection of pop-

ulations of wild flora and fauna only. This is in accordance with the general consensus on this matter where prevention or limitation of effects on the population constitutes a standard protection goal (Copplestone et al., 2007). Nonetheless, it should be recognized that protection is afforded to a number of plants and animals at the individual level (see Pentreath, 1999) and that a system should therefore be flexible enough to assess effects on individuals as well as populations. For practical reasons, the approach taken by EPIC focuses on environmentally-relevant endpoints at the individual organism level, hence all data collation and subsequent analyses are made at the individual level.

### 5.4.3.2. Arctic

From the information compiled in EPIC, a preliminary scale which maps observed biological effects onto ranges of absorbed dose was constructed (Table 5·6). Dose-effect relationships were tabulated for the generic groups: terrestrial animals, terrestrial plants and aquatic animals. The table also includes the 'background' dose-rate range observed under natural conditions.

A general conclusion was drawn by Sazykina et al. (2003) that the threshold for minor effects in sensitive vertebrates lies somewhere within the range 0.5 mGy/d to 1 mGy/d for chronic low-LET radiation and effects on reproduction within the range 2 mGy/d to 5 mGy/d. This can be compared with the observation by UNSCEAR (1996) that although sterility has been induced in sensitive mammals at dose rates of around 4 mGy/d with exposure periods of a few months, no damaging response at dose rates around 0.9 mGy/d over the whole life of the same mammals could be observed. However, the extrapolation of biological effects observed at one level of biological organization to a higher level is not a simple matter. Although minor effects on morbidity in sensitive vertebrate animals are observed at the dose range specified above, populations of highly productive vertebrate organisms (mice, some ubiquitous fish species) are viable at dose rates of the order 10 mGy/d.

The establishment of dose limits may therefore depend not only on the types of organism that require protection but on the level of protection, for example, protection of viable populations versus protection of individuals from a particular radiosensitive species.

According to the ERICA approach (see Garnier-Laplace et al., 2006) that draws on information provided in EC recommendations (the so called Technical Guidance Docu-

ment; EC, 2003), two main methods can be used for effect analysis and the subsequent derivation of risk assessment benchmarks. The first, termed the Safety Factor method, uses expert judgment to define assessment/safety factors that ensure a margin of safety. These factors usually vary from 10 to 1000 depending on the quality and quantity of available effects data, and combine multiple sources of uncertainty with an unclear degree of conservatism. The second method is based on the construction of Species Sensitivity Distributions (SSDs) and the derivation of benchmarks according to a clearly defined set of rules. Although this method has the potential to provide a more transparent approach to dealing with uncertainty, it requires that the knowledge on dose-effect relationships is adequate with respect to the problem formulation.

The generalized conclusions within EPIC regarding the threshold dose-rates at which various effects are observed are consistent with earlier studies. From the available information it was, therefore, not possible to justify any Arctic-specific dose-standards at that time. Assumptions of Arctic vulnerability might provide justification for applying a more conservative safety factor to any derived dose limits. However, the data set upon which such a conclusion is drawn is limited in scope and the hypothesis relating to whether there is a unique expression of radiation-induced biological damage under Arctic conditions remains to be properly tested.

By applying the SSD method and combining terrestrial, freshwater and marine data, Garnier-Laplace et al. (2006) were able to derive a chronic exposure screening dose rate of 10μGy/h applicable to all ecosystems considered within ERICA. Ideally, a similar type of approach might be applied in the derivation of benchmarks or screening levels for Arctic environments specifically, but it is quite clear that the quality and coverage of appropriate data might render such a venture impracticable. For example, in many cases with the Arctic-relevant information as evidenced by the EPIC database, dose-effect relationships are often poorly characterized. In this case recourse could be made to a less data-demanding safety factor approach.

## 5.5. Risk characterization

Risk characterization is the last step of risk assessment. It has been defined as the integration of evidence, reasoning, and conclusions collected in hazard identification, dose-response assessment, and exposure assessment and the estimation of the probability, including attendant uncertainties, of occurrence of an adverse effect if an agent is administered, taken, or absorbed by a particular organism or population (IUPAC, 2001).

The management of uncertainty in the context of environmental impact assessment was considered by Zinger et al. (2007). For example, Walker et al. (2003) classified uncertainties in terms of their *location* (where they occur) and their characteristics – given dimensions of *level* (whether it can best be classified as statistical uncertainty, scenario uncertainty or recognized ignorance) and its *nature* (knowledge-related uncertainty or inherent variability). Van der Sluijs (2003) added dimensions on the *quantification of knowledge base* (identification of weak and strong parts in the assess-

ment) and *value-ladenness* of choices (biases that may shape the assessment). Practical tools such as an uncertainty matrix (van der Sluijs et al., 2005) have been developed to aid the assessor in classifying uncertainties.

### 5.5.1. Assigning probability distributions to input data and parameters

In order to estimate the probability, including attendant uncertainties, of effects in the environment from exposures to ionizing radiation, probabilistic approaches can be applied to propagate uncertainties through an assessment. Such methods require the assignment of probability distributions to uncertain variables.

Probability distribution functions are a convenient instrument for representing quantitative uncertainty in the inputs and parameter values because they enable the use of existing probabilistic techniques for uncertainty and sensitivity analyses. There are a number of ways of assigning a probability distribution depending upon the availability and quality of data, as described by Beresford et al. (2007a).

### 5.5.2. Undertaking uncertainty and sensitivity analyses

Once parameters have been assigned suitable distributions, an uncertainty or sensitivity analysis may be undertaken. There are a number of methods available for undertaking this type of analysis of which one is outlined in this section.

To estimate the uncertainty of the endpoints of the exposure assessment, the uncertainties in the inputs and parameters must be propagated through the model. A good discussion on this subject was had within the IAEA (1989). When simple analytical expressions for the probability distributions are available, variance propagation can sometimes be applied for propagating the uncertainties (Morgan and Henrion, 1990; Hoffman and Hammonds, 1994).

#### 5.5.2.1. Monte Carlo analysis

When analytical methods cannot be applied, Monte Carlo analysis may be used to propagate uncertainties in input data and model parameters through the model to provide a probability distribution of the endpoints. This may be used as a quantification of the uncertainties of the estimations. The basics of the Monte Carlo method are relatively straight-forward (see Vose, 1996): point estimates in a model equation are replaced with probability distributions, samples are randomly taken from each distribution, and the results tallied usually in the form of a probability density function or cumulative distribution. This process is illustrated in Figure 5.4 for the case of a simple model with one input, one parameter and one endpoint.

This type of approach has been applied within the framework of the ERICA project (Zinger et al., 2007; Beresford et al., 2007a). The approach has currently not been applied to the Arctic for lack of underlying information that might allow probability distribution functions to be defined for the parameters used in the risk characterization. Primarily, this relates to the lack of statistical information for the CR values collated within the EPIC project.
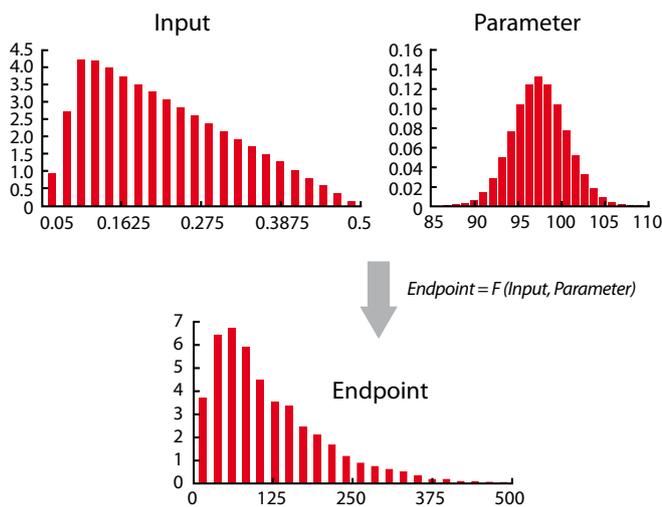
**Figure 5.4.** In the ERICA tool Monte Carlo probabilistic simulations are used for propagating the uncertainties in the inputs and parameters through the model. As result, a probability distribution of the endpoints is obtained, which can be used for quantifying the uncertainties of the estimations. In this example, the endpoint is calculated with a function F (the model) of one input and one parameter.

Uptake of radionuclides in Arctic seal species has been modeled using a biokinetic model applying the Monte Carlo type analysis (Brown et al., 2006b). Model results were compared with empirical data from samples taken within the Arctic region. Results indicated that the model performs well when estimating concentrations of $^{137}$Cs in two seal species for both median values and reproduction of the distribution of data values, but not as well for a third seal species. Likely factors affecting the results were considered to be the probability density functions used for the input parameters.

## 5.6. Available assessment tools and examples of their use

Software tools are available that allow environmental impact assessments to be conducted in a structured self-contained manner. The two most pertinent examples are the RESRAD-BIOTA code and the ERICA Tool.

- The RESRAD-BIOTA code (http://www.ead.anl.gov/resrad) was designed to be consistent with, and provide a tool for, implementing the graded approach for biota dose assessment (US DOE, 2002). The code includes a kinetic-allometric approach (Higley et al., 2003) to estimate the transfer of radionuclides to animals. The internal and external DCCs (relating unweighted absorbed dose to media or biota activity concentrations) were estimated using a Monte-Carlo transport code.
- The ERICA Tool is a computerized, flexible software system that has a structure based upon the ERICA Integrated Assessment tiered approach to assessing the radiological risk to biota (http://www.project.facilia.se/erica/download.html). The Tool guides the user through the assessment process, recording information and decisions and allowing the necessary calculations to be performed to estimate risks to selected biota. Tier 1 assessments are media-concentration based and use pre-calculated environmental

media concentration limits to estimate risk quotients. Tier 2 calculates dose-rates but allows the user to examine and edit most of the parameters used in the calculation including concentration ratios, distribution coefficients, percentage dry weight soil or sediment, dose conversion coefficients, radiation weighting factors and occupancy factors. Tier 3 offers the same flexibility as Tier 2 but allows the option to run the assessment probabilistically if the underling parameter probability distribution functions are defined. Results from the Tool can be put into context using incorporated data on dose-effects relationships and background dose-rates (Brown et al., 2008).

Either of these tools could be applied to Arctic environments for selected scenarios in a straightforward manner. This would require, however, an adaptation of existing parametric data to account for Arctic-specific conditions, primarily in relation to transfer datasets and dosimetric parameters reflecting Arctic reference organisms. Although a reasonable coverage of such information is available from the EPIC project, it is envisaged that further work might be necessary especially in relation to supplementing existing datasets with statistical information.

### 5.6.1. Case study: Integrated environmental management of the Barents Sea

An environmental impact methodology based on the methods developed within the EPIC project has been applied in Arctic marine environments (Brekken et al., 2004) as input to broader integrated environmental management plans under the auspices of the Norwegian Ministry of the Environment (RNMER, 2006). Scenarios involving the transfer of radioactive substances to the Barents Sea from external source areas were selected for further analysis. The scenarios included worst-case accidents at the Sellafield nuclear fuel reprocessing site and at the Kola Nuclear Power Plant. The Norwegian Radiation Protection Authority's marine box model (Iosjpe et al., 2002) was employed to simulate seawater and sediment activity concentrations with time. The total exposure was determined to be low for both accident scenarios. For the Sellafield scenario, the accumulated dose over a period of 20 years was calculated to fall within the range, for different organism types of 0.22 mGy to 5 mGy, while the doses for the Kola accident scenario were equivalent to about 10% of these levels (Figure 5.5). Cesium-137 contributed the most to the total dose for both scenarios with the exception of the category 'marine mammals' for the Kola scenario and 'seabirds' for the Sellafield scenario where $^{134}$Cs and $^{90}$Sr respectively contributed most. The authors of the report considered that the lack of information in relation to low dose chronic irradiation of plants and animals meant that definitive conclusions about possible effects in the environment were difficult to draw. However, it seems evident from the limited data available that significant detrimental effects for the predicted doses would have been unlikely.

### 5.6.2. Case study: Komi Republic

The Komi Republic is another boreal environment where some work has been conducted in relation to the applica-

Weighted absorbed dose rate, µGy/y
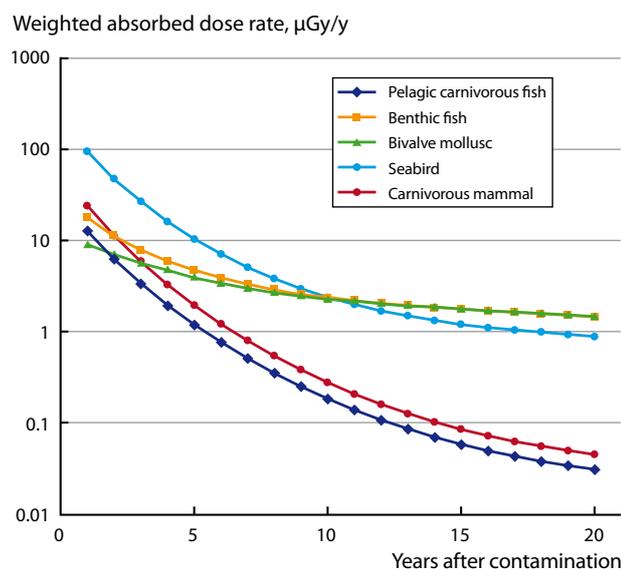


**Figure 5·5.** Predicted weighted dose rates ($^{137}$Cs, $^{134}$Cs, and $^{90}$Sr) to reference organisms from the Barents Sea for the Kola nuclear power plant scenario. A 'worst case' release scenario as described by Larsen et al. (1999) was selected for this example. Further details are provided by Brown et al. (2003).



**Figure 5·6.** Location map for the areas considered within the Komi Republic, Russia.



**Figure 5·7.** Approximate locations of the Vodnyi study sites: Obzhig (U-Ra site 1), Krokhal (Ra site 1), and Otvally (U-Ra site 2).

tion of environmental impact assessments (see Figure 5·6). Although the Komi sites technically fall outside the AMAP boundaries as defined by AMAP (1997) the northern part of the republic has a sub-Arctic climate. Figure 5·7 shows the approximate locations of the Vodnyi study sites.

The ERICA impact assessment methodology was applied to the area with a view to identifying problems at a time prior to the final release of the approach, thus allowing any critical problems to be addressed (Beresford et al., 2007b). Measured activity concentrations of $^{226}$Ra were within the ranges predicted using default ERICA CR values at the Komi case study areas for grasses, herbs and shrubs. In contrast, activity concentrations of $^{226}$Ra in trees were under-predicted by the ERICA assessment Tool. For $^{232}$Th and $^{238}$U there was reasonable agreement (mean values being within an order of magnitude) for all vegetation types at one of the selected sites. However, at another site predictions tended to be high compared to measured values, with the predicted mean often being in excess of the measured upper range. Beresford and co-workers considered that differences might be explained by the use of soil ash weight activity concentrations in assessments at all Komi sites because this would result in predictions being comparatively high. Very small datasets rendered a comparison between predicted and observed activity concentrations in mammals (tundra voles) within the Komi case study problematic. The limited comparison that could be undertaken inferred that while predictions for $^{226}$Ra were in reasonable agreement, $^{238}$U and $^{232}$Th were under-predicted. It was noted that the default CR values for $^{232}$Th and $^{238}$U for some reference organisms (e.g., mammals) are dominated by data from the United Kingdom, although for others (e.g., birds) they are derived from data from the Komi Republic (Beresford et al., 2005). The validity of applying transfer parameters derived primarily from temperate environments to Arctic systems has not been established at even a basic level. The quite limited analysis conducted above already suggests that significant discrepancies in transfer may occur.
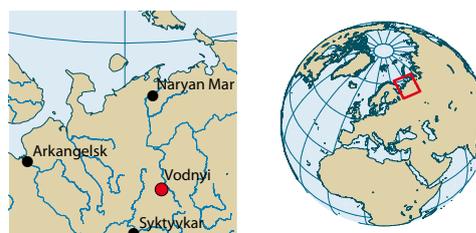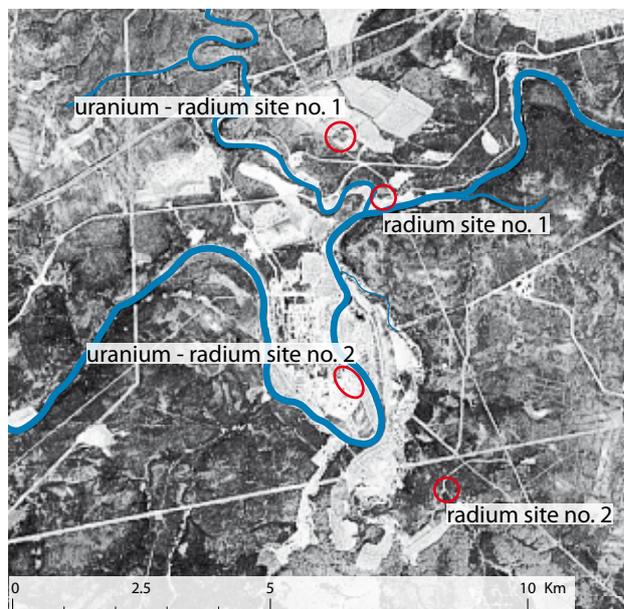
Predicted external dose rates for the ERICA suite of on-soil reference organisms at the sites Krokhal, Obzhig and Otvally within the Komi area compared favorably with the expected absorbed dose rates to organisms in the field as inferred from air measurements (Gamma air kerma rates). Radium-226 was found to be the main contributor to the predicted external dose rates at all three sites; about 90% at Krokhal and Otvally, but only 60% at Obzhig where the contribution from $^{228}$Th was higher.

The Komi sites offer a unique insight into environmental effects arising from the presence of radionuclides in Arctic environments. At Obzgig, biological effects in tundra voles were studied in the same year. Chromosome aberrations were documented at this site including the elevated appearance of 'bridges' and 'fragments' (Materyi et al., 2003). The predicted dose rate for Mammal (Rat) representing the most suitable reference organism for analyses using the ERICA Tool corresponds to about 70 µGy/h at this site (Beresford et al., 2007b). The Otvally site is of particular interest because data on radiation effects on plants have been collated here. Effects on bird vetch (*Vicia cracca*) were studied in 1980/81 (Popova et al., 1984, 1985; Bondar and Popova, 1989). Effects reported in these studies comprised cytogenetic disturbances and lower seed weights in exposed populations compared to controls. Simultaneous dose rates were not derived but can be assumed to be considerably lower than the predicted weighted total dose rate to 'Grasses and Herbs' (the reference organism that most closely represents *Vicia cracca*) of about 460 µGy/h that was

derived for this site about a decade earlier using the ERICA Tool (Beresford et al., 2007b).

The only conclusion that could be drawn following application of the ERICA methodology to the site using available activity concentration data (for both media and biota) was that the analyses could not provide any definitive validation of the effects prediction provided by the ERICA approach beyond the observation that the predictions made were not in contradiction to effects observed in the field. This reflected the limitations in relating effects observed for one set of endpoints (i.e., those predicted by the ERICA approach) to those observed for another (i.e., *in situ* observation). Comparison is further complicated by the recognition that many of the long-lived radionuclides present are also chemically toxic. U-238 and $^{232}$Th with their relatively low specific activities are two radionuclides for which their chemical toxicity is likely to dominate over their radiotoxicity.

## 5.7. Concluding comments

A framework for the protection of the Arctic environment has been presented within this chapter. The approach allows for the impact on plants and animals from exposure to radiation to be quantified in a robust manner through integration of current knowledge concerning transfer in the environment, dose-rates from the presence of radioactivity within biota and their media, and information concerning radiobiological effects on endpoints of concern from an environmental protection perspective. Methods are also applied to allow variability in parameters to be considered and risk in the sense of characterizing the probability as well as the severity of given effects.

In recent years, developments in frameworks for the protection of biota from ionizing radiation have been applied to temperate regions but a similar process has not occurred for the Arctic. More specifically, data concerning transfer in the Arctic environment have not been collated in such a way as to lend themselves to probabilistic approaches. Nor have the most recently developed dosimetric methods been applied to the situation in the Arctic. Hypotheses concerning the greater vulnerability of Arctic plants and animals to exposures to ionizing radiation have not been tested nor have background dose rates to biota in Arctic environments been derived in a robust way. Methods have been applied in temperate environments in order to derive screening dose rates designed to be protective of sensitive endpoints in generic ecosystems. Such methods might be adapted for use in the Arctic by selection of appropriate effects data and application of bespoke uncertainty factors. Some of the deficiencies could be addressed through application of exposure assessment tools that are presently available for general use by the scientific community.

There have been well founded arguments forwarded in the published literature that present concerns about a disproportionate reaction to the consideration of environmental protection from ionizing radiation, with a case presented for continuing with the original ICRP axiom as paraphrased in the term "if man is protected then the environment is protected" (Smith, 2004). Although a framework for protection of the environment from radiation has been long overdue for various reasons, including the requirement to fill a conceptual void and to place environmental assessment on a firm regulatory footing, the remarks concerning the need for a simple and cost-effective regulatory approach are highly pertinent.

The highly destructive nature of human activities is well documented. Every year, it is believed that between 17 000 and 100 000 species vanish from the earth, a level which has caused some scientists to suggest that the world is undergoing a sixth great period of extinction (Leakey and Lewin, 1995). Furthermore, humans exploit natural resources often to an extreme degree. In the ideal case, it can be hoped that the economic and societal benefits from these activities are counterbalanced by considerations for sustainability and environmental conservation but, as often happens, the latter are allocated a low priority. Protection of the environment from ionizing radiation needs to be seen within this context and it has been argued by Smith (2004) that ideally resources should be mainly allocated to causes of significant environmental concern. The ongoing work of many groups in recent years shows that the requirement to regulate for radiation exposure of plants and animals is an important issue but it needs to fit within an overall framework of environmental protection and management and the concomitant resource allocation should be proportionate to the risk of harm. The framework presented in its present form should fill these requirements. However, there is still a need to refine the system not only in terms of harmonizing with the approaches applied for other contaminants, but also with regard to the understanding and quantification of specific interactions and possible combined effects of these contaminants.

*Chapter 6*

# Climate Change

## 6.1. Introduction

Climate change and concomitant environmental change have been the subject of much scientific and political debate for some years although it is only relatively recently that in-depth assessments of climate change and its probable impacts on the environment have been published. The assessments conducted by the Intergovernmental Panel on Climate Change (IPCC) and the Arctic Climate Impact Assessment (ACIA) comprise the most thorough scientific evaluations of global and Arctic climate change, with both groups concluding that Arctic climate change is occurring, that the rate of change is accelerating faster than previously thought, and that the predicted changes will have long-lasting, extensive and fundamental impacts on Arctic ecosystems, their biotic/abiotic constituents and the populations and societal structures fundamentally linked to them. Arctic radioecology and the heightened vulnerability of Arctic populations to radioactive contamination have been defined by the Arctic-specific conditions and processes that control the behavior and fate of radioactive contaminants in this environment and it is these conditions and processes that appear to be most vulnerable to the changing Arctic climate.

### 6.1.1. The IPCC, ACIA and AMAP assessments

The first major scientific appraisal of Arctic climate change was the IPCC Third Assessment Report – *Climate Change 2001: Impacts, Adaptation and Vulnerability* (IPCC, 2001). The IPCC assessment presented Arctic climate change as a sub-section of the prognosis for global climate change while the second major assessment concerning climate change focused specifically on the Arctic and was carried out by ACIA. This was commissioned by the Arctic Council and conducted by two of its working groups: the Arctic Monitoring and Assessment Programme (AMAP) and the Conservation of Arctic Flora and Fauna (CAFF) in association with the International Arctic Science Committee (IASC). A distillation of the major conclusions of this assessment was published as *Impacts of a Warming Arctic* (ACIA, 2004). The findings of this assessment support the conclusions of the IPCC assessment and expound on the impact of Arctic climate change on Arctic peoples and the economics and societal impacts in general. The most recent report of the IPCC (IPCC, 2007) does not deviate significantly from the findings of its first assessment or that of ACIA in relation to potential impacts on the Arctic region.

Given the range of models and scenarios used in climate assessments, predicted changes in Arctic climate are typically provided as ranges for parameters such as temperature and precipitation. The IPCC (2007) predicted average temperature rises in the Arctic of between 2 °C and 9 °C, whereas ACIA (2004) predicted average rises of between 4 °C and 7 °C for the same period. Both assessments discussed the general probable consequences of such rises, including a reduction in sea ice cover of up to 30%, reduction in the area underlain by permafrost, increased ground temperatures and changes in precipitation patterns across the Arctic region. The impacts of these climatic changes on the Arctic environment include a thickening and extension of the active layer of permafrost throughout the Arctic with subsequent alterations in drainage, hydrology, terrestrial ecology and the physical alteration of Arctic and sub-Arctic landscapes. A projected reduction in the ice-jam flooding of major river systems will result in changes for Arctic riverbank ecosystems and freshwater ecology, especially in the highly productive large Arctic river deltas. The melting of snow and ice will potentially impact on the water cycle and hydrological systems of the region, while changes in precipitation patterns are predicted to lead to the Arctic runoff regime being precipitation-driven with reduced seasonal variation in runoff levels and drying out of Arctic peat lands, bogs and tundra due to increased evapotranspiration. Warming of the Arctic environment is likely to lead to an increase in biological production although the effects of increased precipitation on this parameter remain unclear. High-Arctic plants will probably display strong growth in response to summer warming although exposure to higher levels of ultraviolet-B (UV-B) radiation may result in damage to vegetation. As warming occurs, there will be general changes in species compositions and assemblages with a tendency for northward shifts and the ultimate loss of some significant Arctic species. Changes in sea ice will affect the seasonal distributions, geographic ranges, patterns of migration, nutritional status, reproductive success, and ultimately the abundance and balance of many Arctic species. Reductions in sea ice extent will cause an increase in the frequency and severity of Arctic storm events; affecting coastal regions through increases in the rate and extent of coastal erosion. Arctic climate change combined with other stresses will also affect human communities in the Arctic. The impacts of a changing climate will be particularly significant for the communities of indigenous peoples that pursue traditional lifestyles. Changes in sea ice, duration of snow season, and habitats and diversity of food species will affect hunting and gathering activities and could impact on longstanding traditions and ways of life. Indigenous peoples face significant impacts in relation to both the cultural and economic structures of their populations.

A fourth assessment related specifically to Arctic climate change was that by AMAP entitled *The Influence of Global Change on Contaminant Pathways To, Within, and From the Arctic* (Macdonald et al., 2003). The brief section of the report concerning the transport of radioactivity to the Arctic under climate change scenarios concluded that pathways of anthropogenic radioactive contamination to the Arctic and contaminant loads will not constitute any greater hazard to the Arctic population than is the case today. The assessment did not aim to provide a thorough assessment of how specific radioecological processes within the Arctic may alter as a result of climate change, but was largely focused on the physical transport of contaminants to the region. However the point was made that potentially the most significant increase in radiation exposure to Arctic residents may arise

from increased emanation of $^{222}$Rn from soil (with a resulting increase in environmental levels of $^{210}$Pb/$^{210}$Po as a result of thawing of permafrost and reduction in snow cover and that: '*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.*'

The AMAP assessment, although less wide-ranging in scope and content than the IPCC or ACIA assessments, is the only one to focus specifically on Arctic contaminants and so its conclusion concerning the potential effect of Arctic warming on the radiation dose to Arctic residents from $^{222}$Rn and its daughter products should be viewed with some significance.

## 6.1.2. Potential impacts on possible sources of radioactive contamination in the Arctic

Irrespective of the potential changes in the background radiation environment (see Macdonald et al., 2003), the predictions outlined within the ACIA and IPCC assessments have potential implications for some actual and potential sources of radioactive contamination that exist in or discharge to the Arctic. Although the consequences of a changing Arctic climate in relation to potential sources, such as the nuclear facilities on the Kola Peninsula or the Bilibino power plant, are difficult to predict they should still be the focus of some attention. Similarly, impact assessments related to possible future activities of significance with respect to radioactive contaminant levels can only be conducted satisfactorily if emerging predictions for the future status of the Arctic are recognized and integrated into assessment efforts. The focus of much attention in recent years, with respect to protection of the Arctic from radioactive contamination, has been coastal nuclear facilities (both military and civilian nuclear facilities and radioisotope thermoelectric generators). It is of some concern that such sources and potential sources lie in the Arctic zones that are likely to undergo rapid and significant physical change as a result of the increased coastal erosion / storm damage that are predicted to occur as a result of retreating sea ice. While climate change may affect the vulnerability of potential sources actually present in the Arctic, it may also have potential impacts on the transport of materials to the Arctic from sources that lie outside the region. Of most importance in this regard are the predicted changes in the hydrology of the catchments of the major Siberian rivers that constitute a link between sources of radioactive contaminants outside the Arctic region and the Arctic marine environment. At some coastal sites in the Arctic, contamination has already been released into the onshore and offshore coastal environment. Continuing management and long-term remediation of coastal sites and facilities need to take account of the scope for further migration of contamination; see for example discussions by Shandala et al. (2007) and Sneve et al. (2007b). Given the timeframe for site management, the optimum plan for contaminated land management needs to take account of how climate change may affect radionuclide migration potential, in terms of changed near-surface hydrogeochemistry, and weathering and erosion processes. Consideration may also need to be given to existing accumulations of radionuclides in marine sediments. Remobilization of long-lived contamination in bed sediments, by increased storm activity and coastal change effects may create an enhanced source term to the water column, including, for example, from relatively distant locations, such as the Irish Sea (into which the reprocessing facility at Sellafield discharges).

The Arctic has long been described as a region of particular vulnerability with respect to radioactive contamination due to a range of factors. For example, the number of potential sources in, or relevant to, the region and Arctic-specific processes that can result in Arctic residents being subject to higher exposures than an equivalent amount of environmental contamination would produce in more temperate regions. The latter can be reduced to both societal aspects and environmental aspects (such as elevated uptake of contamination by Arctic vegetation and relatively short Arctic food chains facilitating efficient transfer of contamination). To a great extent, Arctic radioecology and the radiological protection of both the Arctic environment and its human populations have been based on the recognition of those aspects that are unique to the Arctic. The findings of the IPCC and ACIA assessments appear to indicate that the tenets upon which Arctic radioecology and radioprotection are based are likely to undergo changes within the next century as a result of climate change. On a socio-economic level, the indigenous peoples of the Arctic that are most susceptible to radioactive contaminants also constitute the populations most vulnerable to the impacts of climate change. The social, agricultural and economic practices of these populations are already undergoing significant alteration due to the impacts of climate change. How subsequent shifts in these practices will affect the vulnerability of these populations and their exposure to radioactive contamination can only be determined by a detailed examination of the implications of climate change on the practices of most relevance to their exposure to radioactive contamination.

Current understanding of Arctic radioecology and the processes that are accepted as governing the behavior and fate of radioactive contamination within the Arctic environment is based on the Arctic environment that exists today. This understanding is not only relevant to our ability to elucidate radioecological processes with respect to current levels of contamination but also to our ability to ascertain the likely impacts of future events that may introduce contaminants to the region. The Arctic-specific factors of relevance to the radioecology of the region include, but are not limited to, the lack of mobile water in the Arctic terrestrial environment; the low nutrient status of the Arctic terrestrial environment; the ability of Arctic soils to constitute an effective sink for radionuclides; and the limited number of marine and terrestrial species in the Arctic. These factors are all products of the low temperature regime of the Arctic region that, according to all current assessments of Arctic climate change, is undergoing a rapid and significant alteration. It would be unreasonable to assume that predicted climate change scenarios will have no impact on the behavior of radionuclides in, in particular, the Arctic terrestrial environment. Of some pertinence also are the shifts likely to occur in relation to perhaps the most significant component of the Arctic terrestrial environment, the permafrost/tundra system. Tundra constitutes what is possibly the largest sink for radioactive contaminants in the Arctic and changes in the physico-chemical nature of Arctic tundra may have

direct implications for the understanding of, among others things, the retentive capacity of Arctic soils for radioactive contaminants and the uptake of such contaminants by vegetative species found there. The frozen state of Arctic permafrost and tundra has, in the past, contributed to the general perception of the Arctic environment as one that exhibits low mobility in relation to radioactive contamination, a situation which is likely to change with the thawing of permafrost, the drying out of the tundra, and the changes in precipitation patterns predicted for the Arctic. All assessments conducted to date indicate that changes in Arctic tundra will lead to alterations in vegetative species assemblages and the productivity of the terrestrial Arctic environment. Such changes will potentially disrupt current Arctic terrestrial food chains and subsequently the currently accepted predominant transfer pathways of radioactive contamination from the terrestrial environment to people.

The previous AMAP observation in relation to the potential for a significant and widespread change in the natural radiation background of Arctic residents is perhaps the clearest indication of how climate change may directly and precipitously affect the radioecology of the Arctic region. Radon emanation from soil is highly dependent on temperature and barometric pressure, both factors being vulnerable to climate change, and the movement of $^{222}$Rn and its daughter products $^{210}$Pb and $^{210}$Po (which are the dominant radiation dose contributors to Arctic residents) in the Arctic environment after escape from soil is to an extent a function of precipitation, snow cover, and soil conditions. In the context of the suite of Arctic contaminants and climate change, anthropogenic radioactive contamination is perhaps unique in that it is the only one which will be subject to changes in the baseline, i.e. natural background dose-rate, against which its levels and effects are evaluated and expressed. Irrespective of the direct potential effects of a tripling of the natural background dose on Arctic residents, it is also worth considering this potential increase in the presence of a variety of other stressors (such as increased UV-B exposure), elevated levels of or enhanced vulnerability to other contaminants (such as heavy metals and POPs) and the societal stresses imposed by changes in society and culture.

### 6.1.3. Arctic radioprotection

The last few years have seen a shift from the concept of radioprotection oriented towards humans to radioprotection of the general environment and much effort has been devoted to establishing relevant frameworks and functional methodologies to ensure adequate levels of environmental protection (see Brown and Dowdall, 2006). Arctic-specific radiation impact assessment methodologies and criteria have only recently been established and it seems clear that many of the parameters currently used in the underlying models may be subject to significant modification as the climatic conditions governing those parameters alter. Although determination of the exact role various factors play in the environmental behavior and fate of radionuclides is primarily a task to be undertaken as part of future empirical studies, some insight into the likely consequences for biota within a changing radiation regime may be gained at a rudimentary level by using the prognoses from climate research as input data to con-

ventional radioecological models. Such information could be usefully placed within the broader context of environmental impacts arising from physical disturbances, changing human practices and the effects of other stressors and contaminants within the Arctic as predicted by climate change scenarios.

## 6.2. Actual and potential sources of anthropogenic radioactivity

### 6.2.1. Nuclear facilities in the Arctic – vulnerability

A significant number of nuclear facilities of various types exist in the Arctic region, summary details being contained in the relevant AMAP reports (AMAP, 1998, 2004a). Such facilities vary widely with respect to infrastructural stability/integrity and hence their potential vulnerability to climatic or climate-related parameters such as storm surges, geocryological stability, and coastal erosion. The impact of the Arctic climate on such facilities is well- and long-evidenced in particular for the Andreeva Bay Shore Technical Base on the Kola Peninsula. The initial incident that resulted in extensive contamination of the Andreeva Bay site was caused by loss of integrity in a fuel storage facility due to climatic freeze-thaw action on the infrastructure. Continuing infrastructural degradation of facilities at the site has been exacerbated by severe climatic conditions and previous and ongoing dispersion of radioactivity from the site has largely been as a result of the ingress of precipitation to storage facilities with subsequent run-off of contaminated waters to the nearby marine environment or via the erosion and dispersion of contaminated soils from the site again via the action of precipitation. The vulnerability of nuclear facilities in northwest Russia and such radioactive materials as may be stored therein to the effects of climate change would seem to be largely determined by a range of parameters, among them the site's location, geomorphology, and design/construction. The impacts of the effects of climate change in so far as it has manifested itself to date have been noted for a variety of industrial installations along the Eurasian Arctic coast (ACIA, 2004). It is unlikely that nuclear facilities within the Arctic are any less vulnerable from an infrastructural point of view and it may therefore be argued that an assessment similar to those conducted for nuclear infrastructure in other (arguably less vulnerable) countries may be warranted for the Arctic.

### 6.2.2. Power plants

Two nuclear power plants are located within the Arctic region – the Kola nuclear power plant near Polyarny Zori on the Kola Peninsula and the Bilibino nuclear power plant in the Chuckchi region in eastern Russia. The Bilibino power plant is located in an area of high geocryological hazard potential as defined by ACIA for the B2 emissions scenario by 2050 (ACIA, 2004) and is specifically mentioned within the ACIA assessment as an example of an installation at risk from the effects of ground destabilization due to a warming climate.

## 6.2.3. Radioisotope thermoelectric generators

Given the nature of construction of typical RTG designs it is hard to envisage any climate-related scenario that would lead to enhanced risk of exposure or accidental release of radioactivity to the environment. Perhaps the only potential relevant scenario would be the loss of an RTG from a coastal site due to storm action or coastal erosion or a combination of the two, a scenario for which only a sub-set of the total number of RTGs present in the Arctic are vulnerable. In the event of a worst case scenario such as loss to sea of an RTG as a result of storm action or erosion, risk analysis indicates that such an event would result in negligible consequences for people or the environment (Standring et al., 2007).

## 6.2.4. Tundra

The terrestrial environment of the Arctic is composed to a large extent of organic soils and tundra soils high in organic materials and such matrices constitute the largest terrestrial Arctic sink for contaminant anthropogenic radionuclides and natural radionuclides of the uranium and thorium series. The potential implications of climate change with respect to tundra and organic soils and their radioactive contaminant loads most probably relate to possible alterations in the mechanisms controlling the behavior of such contaminants in the Arctic terrestrial environment. The Arctic terrestrial environment is most often characterized as a low-mobility environment for radionuclides (Dowdall et al., 2003) as the processes normally governing mobility and redistribution of radionuclides are retarded or non-existent due to the low-temperature regime. Warming temperatures, changes in precipitation types and amounts, and shifts in hydrological regimes may, in general, be expected to produce changes with respect to Arctic soils in terms of their organic matter contents, structure, pH, redox and nutrient status. The roles of these factors for a variety of radionuclides have been established to a large extent although specific information relating to Arctic conditions is lacking. Destabilization of tundra as a result of warming temperatures and other climatic effects has the potential to alter the ability of this compartment to function as an effective sink (see section 6.3.2) and significant changes in tundra or organic soils are a matter of potential consequence with respect to Arctic radioactive contaminants.

In recent years, elevated levels of dissolved organic carbon (DOC) have been observed in waterways in a number of countries and this is often regarded as a climate-related phenomenon. For example, an increase in concentrations of DOC in streams and freshwater systems in the United Kingdom has been reported by Evans and Monteith (2001), Evans et al. (2005), Freeman et al. (2001) and Worral et al. (2003); the latter two articles proposed increasing temperatures as a potential cause of extra leaching of carbon material from peat soils. Hejzlar et al. (2003) also reported increased DOC in a waterway in South Bohemia (Czech Repeblic); similar observations have been made in the Nordic countries and the continental United States (Schindler et al., 1997; Driscoll et al., 2003; Stoddard et al., 2003; Skjelkvåle et al., 2005). Elucidating the reasons for increasing concentrations of DOC in waterways, which indicates a loss of soil organic matter

(SOM) from the catchment soils, has been of interest because losses of carbon in this way from soil can have an effect in a variety of scientific disciplines. Insam (1990), among others, showed that the conversion of SOM to the low molecular weight organic compounds that constitute dissolved organic material (DOM), by microbially-derived enzymes, is controlled to greater or lesser extents by temperature, hydrology, nutrient status and vegetation species. Eivazi and Tabatabai (1990) also demonstrated the role of pH, redox conditions and certain inorganic species in this process. Despite evidence to suggest that the processes controlling DOM concentrations may be a function of temperature, Pastor et al. (2003) concluded that the increase in DOM concentrations observed in previous studies could not be fully explained by general temperature increases (presumably as a result of climate warming that may have occurred to date). Principle component analysis (PCA) of field data collected in Wales by Bonnett et al. (2006) led to their conclusion that 87% of the seasonal variation in DOM concentrations they observed was due to soil temperature, although they stopped short of asserting that long-term trends in increasing DOM concentrations were due to generally rising temperatures. Worrall et al. (2004) tested the theory that climate change alone could account for the observed trends in DOM concentrations and concluded that the climate change that has occurred to date (a purported rise of some 0.78 °C) could only account for some 6% of the total DOM increase observed. Given that seasonal temperature variations have, in some studies, been shown to affect DOM discharge, especially from organic and peat soils, and that this may be attributed to the modest climate change observed during the past 30 years, it would appear reasonable to assume that further changes in temperature and concomitant changes in other influential variables (hydrology, pH, redox, nutrient status) will influence DOM discharge in some manner, most probably increasing DOM concentrations in runoff waters.

Possible climate change induced increases in discharged DOM amounts from organic and peat soils in temperate European and Asian ecosystems are an interesting scenario from an Arctic radioecological point of view for a number of reasons. First, peat lands (upland and lowland), tundra and organic soils constitute a significant sink for a variety of radionuclides introduced via discharges from a variety of facilities, deposited during atmospheric nuclear weapons testing, and as a result of the Chernobyl accident, as well as being sinks for a variety of natural radionuclides. Any change of stability in these soils' ability to retain radionuclides is therefore a matter of great interest as it could affect the discharges of radionuclides to runoff waters. Second, in relation to soil-to-plant transfer, DOM plays a major role in the mobility and availability of metals (Lawlor and Tipping, 2003) and increased organically-associated Al and Fe have been observed at sites where increased DOM concentrations have been recorded in soil waters (Evans et al., 2005). The observations of elevated metal concentrations associated with increases in DOM discharges is relevant to radioecology as radionuclides often become associated with DOM which, in turn, will affect their mobility (and hence bioavailability) in soils. Bunzl et al. (1998) conducted studies on the association of Pu, Am and radiocesium with different molecular size fractions of DOM found in podsols and a peat soil col-

lected within 10 km of the Chernobyl nuclear power plant. Both Pu and Am were observed in all size fractions of DOM (between > 2000 and 560 Daltons) from a well decomposed soil (deep peat). However, radiocesium was only observed in one low molecular weight fraction of DOM, in a less humified soil. Bunzl and co-workers offered no explanation for this observation. Agapkina (2002) presented results relating to the occurrence of anthropogenic radionuclides in soil solutions and demonstrated that the association of radionuclides such as $^{90}$Sr, $^{137}$Cs, $^{238,239+240}$Pu and $^{241}$Am with different molecular weight fractions of DOM is dependent on the radionuclide and the soil type from which the solution is derived. However, $^{137}$Cs appeared to behave differently by selectively interacting with specific molecular size fractions of DOM. Thorium mobility in semi-arid soils has also been partly explained by association with DOM (Bednar et al., 2004). The fact that DOM plays a role in the mobility and bioavailability of radionuclides and that climate-influenced changes already appear to be occurring in DOM discharges from soil types which play a special role in the radioecology of many radionuclides indicates the potential significance of climate change on the radioecology of radionuclides in such soils.

## 6.2.5. Ice masses

The two main terrestrial-to-marine transfer pathways for radionuclides in the high Arctic are glacial melt water and terrestrial run-off. Both pathways are subject to climatic variability and long-term climate changes. Glaciers are known to be effective sinks of a range of contaminants including radionuclides, the accumulated contaminants being gradually released from glaciers as a result of glacier movements and transported from the terrestrial to the marine environment by glacial melt waters. The volume of glacial melt waters and the suspended sediment yields from glacierized catchments are controlled by air temperature and precipitation and are thus sensitive to environmental change (Svendsen et al., 2002). Terrestrial runoff is also influenced by air temperature and precipitation (Svendsen et al., 2002) and warming of the terrestrial environment can result in changes in the physical character of soil in ways (porosity, permeability, etc) that may affect the susceptibility of radionuclides to terrestrial run-off.

Radionuclide transfer and deposition processes in Arctic fjords are controlled by freshwater and terrestrial particle fluxes and by the extent and duration of sea ice. It has been shown that the presence of both low salinity water and terrestrially derived particles positively influence the deposition of particle-reactive radionuclides. This is clearly illustrated by the higher sedimentation rates and particle-reactive radionuclide inventories close to glacier fronts with a rapid decrease away from the fronts (Mitchell et al., 1999). All these controls are sensitive to climate parameters, and as such, radionuclide transfer and deposition processes in Arctic fjords are potentially vulnerable to climatic and environmental change.

As an example, two thirds of the Svalbard archipelago, lying between 70° N and 80° N, is currently permanently covered by ice and glaciers. Ice core data have shown accumulation and downward migration of radionuclides within the glacial accumulation area; this area above the mass equilibrium line accounting for an average 37.5% of the total glacier surface area on Svalbard (Kohler, J., Norwegian Polar

Institute, pers. comm., 2004). Radionuclides trapped in the accumulation area will migrate through the glacier following glacier transport pathways and will be gradually released in the glacier ablation area from which they are transported to the marine environment by melt water. The migration and release of radionuclides from glaciers is normally expected to occur over a timescale of several hundred years, but this is subject to the effects of global climate change. The Kongsfjorden catchment area is dominated by several tidal glaciers, of which Kronebreen with a surface area of some 700 km$^2$ and Kongsvegen (ca. 100 km$^2$) are the largest. The total glacial surface area in the Kongsfjorden catchment area is estimated at more than 1000 km$^2$, with an approximate accumulation area of 375 km$^2$. Preliminary calculations based on average integrated depositional values (UNSCEAR, 2000a) indicate a potential total inventory of the order of 400 GBq $^{137}$Cs, 300 GBq $^{90}$Sr, 5 GBq $^{239+240}$Pu and 2 GBq $^{241}$Am, which may be released into Kongsfjorden over the coming years should these glaciers undergo significant degradation as a result of Arctic climate change. While this example is hypothetical, it serves to demonstrate that relatively small local Arctic environments may witness significant inputs of radionuclides in coming years as disruption of sink terms such as glacial ice occurs as a result of a warming climate. This is exacerbated to some degree with respect to trying to predict the impacts of such potential inputs because the specificities of such environments and systems are neither well understood nor the subject of enough research. Further assessment of inventories of sink terms and their vulnerabilities to a changing Arctic climate would therefore appear to be warranted.

# 6.3. Terrestrial Arctic radioecology and climate change

## 6.3.1. Radon

The influence of climatic and meteorological conditions on radon evasion from soils has been well established for many years although relatively little work has been performed on the specificities of radon evasion/exhalation in Arctic conditions. A wide range of parameters affect both the occurrence of radon in soil gas and exhalation of radon to the atmosphere including soil type, texture, content of progenitor radionuclides, moisture content, temperature and atmospheric conditions (see, for example, Washington and Rose, 1990; Tanner, 1980). Recent work by Glover (2006) investigated the role of permafrost as a radon barrier with respect to a model unventilated dwelling and concluded that permafrost reduces domestic radon concentrations by between 80% and 90% (indoor levels of 5 Bq/m$^3$ to 10 Bq/m$^3$ for $^{226}$Ra soil levels of 40 Bq/kg), indoor levels increasing by two orders of magnitude after permafrost thawing due to climate change. This work supports the earlier findings by Sellmann and Delaney (1990) as to deepening of the active layer causing increases in levels of radon in soil gas. While it is safe to assume that warming temperatures and reduction of permafrost/ice will probably result in higher soil gas radon levels, the impact of such changes on both the level of radon in domestic dwellings and on the amount of daughter nuclides such as $^{210}$Po/$^{210}$Pb transferred to plants/animals/humans is more difficult to predict

due to the extensive range of variables involved in the environmental behavior of radon once it leaves the soil. Changes in building practices, dwelling designs, and insulation levels as a result of a changing climate may all impact on radon levels in domestic dwellings or other structures over and above such impacts as may be produced via changes in soil variables and it would appear to be difficult to predict any general shift of levels in such environments. Of significance however is the role played by exhalation of radon gas in the introduction of isotopes such as $^{210}$Po and $^{210}$Pb to the environment.

Pb-210 and $^{210}$Po constitute the largest internal radioactive dose contributor to Arctic residents and biota and this dose tends to be greater than for inhabitants of temperate regions due to the nature of Arctic food chains, diet and other factors. The primary source of $^{210}$Pb to the Arctic has long been considered to be long-range transport of the isotope via air masses that have passed over continental land masses. The presence of snow plays a role in the local availability of radon daughters via 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 role 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 soil itself due to its being frozen for long periods. The lack of work in the differentiation of local and long-range inputs of radon daughter isotopes to the Arctic environment and the elucidation of the role of Arctic-specific processes with respect to local inputs complicates the problem of determining the effect a changing climate may have on potential levels of $^{210}$Pb/$^{210}$Po in Arctic ecosystems. While determining the effect of any one parameter (e.g., soil moisture, particle size) on potential changes in a soils' gas radon levels or the exhalation of radon to the atmosphere is an apparently simple matter, determination of the effects of changes in a range of parameters is appreciably more complicated. Nevertheless, given the level of knowledge as to the influence of individual parameters on the behavior of radon in the environment, it would appear unreasonable to assume that Arctic climate change will not have an impact on the levels of radon gas and daughter products.

## 6.3.2. Soil-to-plant transfer

Lack of previous research on the specificities of Arctic radioecological processes in the terrestrial environment complicates attempts to hypothesize on the potential effects of climate change on, for example, soil-to-plant transfer in Arctic terrestrial systems. Some indications that climate can and does play both direct and indirect roles in soil-to-plant transfer are provided in studies conducted in other regions and usually focused on more general, but nonetheless climate relevant, aspects of soil-to-plant transfer. Seasonal variations in radionuclide uptake by vegetation have been investigated in studies that range in design from field trials conducted over periods of years to short-term laboratory-based activities and that report on both inter- and intra-seasonal variation in uptake to both above- and below-ground tissues. Sandalls

and Bennett (1992) conducted a multi-year study on radiocesium uptake in grass on upland soils and observed seasonal differences for all soil types, suggesting climatic conditions as a possible influence but with no specific climatic variable being identified as a controlling factor. Rafferty et al. (1994a) performed a one-year study on monthly concentration ratios for $^{40}$K and $^{137}$Cs in grasses and demonstrated that the seasonal variation was such that the greatest transfer of $^{137}$Cs occurred in winter while the highest transfer of $^{40}$K was during spring-summer. Seasonal variation in the transfer of particulate soil to vegetation surfaces was determined to be the most probable explanation, as was confirmed by Rafferty et al. (1994b). Ehlken and Kirchner (1996) observed seasonal fluctuations in $^{137}$Cs, $^{134}$Cs, $^{90}$Sr and $^{40}$K transfers in north German soils although no specific pattern emerged. Kirchner and Ehlken (1997) reported some weak trends with variables such as temperature and precipitation but concluded that the interplay of these variables with the hydrological properties of the soils caused the seasonal variations in transfer. Salt and Mayes (1991) observed significant correlations between uptake of radiocesium and radiostrontium and variables such as temperature and rainfall. Marked seasonal variations in soil-to-plant transfer as observed by Baeza et al., (1996, 2001) were hypothesized to be due to variations in temperature and humidity over the season as opposed to changes in the amount of available radionuclides present in the soil. While it is clearly evidenced in the literature that at least radiocesium exhibits some seasonal variation with respect to its transfer, it is the reasons for this seasonality that are of interest in trying to elucidate the potential effects of climate change on terrestrial Arctic radioecology. Soil moisture was proposed by Willey and Martin (1995) as a factor involved in seasonal variations in stable cesium uptake by plants. Salt et al. (1996) observed different patterns in uptake of $^{137}$Cs and $^{40}$K after eliminating water stress as a variable and concluded that declining $^{40}$K towards the end of summer was related to depressed plant growth. Kirchner and Ehlkin (1997) provided what is potentially the best evidence of the impact of short-term seasonal climatic changes on the uptake of cesium and strontium and reported that the impact of climatic variables may be obscured due to some soils exhibiting better retention capacities than others.

Further Arctic relevant information may be gleaned from studies where altitude may possibly serve as an appropriate surrogate for climate. Echevarria et al. (2003) studied the uptake of $^{99}$Tc at locations in France that differed in altitude but were otherwise geologically and topographically similar, in order to simulate the possible effect of climate on soil characteristics and $^{99}$Tc uptake. Higher uptake was observed under the more temperate regime than for the colder sites at the higher altitude. Lettner et al. (2006) demonstrated aggregated transfer factors (Tag) dependency with respect to $^{137}$Cs uptake and altitude (increase with altitude) at an Austrian site and concluded that combinations of climatic variables were involved in the observed variation.

Perhaps the most comprehensive comparative study of the effect of climate region on soil-to-plant transfer was that by the IAEA (2006a), which ultimately concluded that no systematic differences existed between transfer factors in temperate, subtropical and tropical regions. It should however be noted that Arctic regions were not included in these studies.

The fact that some ecosystems reported in the original study exhibited transfers greater or lower than the average values by over an order of magnitude, precipitated the second coordinated research project by the IAEA the aim of which was to study transfer of nuclides to vegetation in relation to the soil types found across different climatic zones. This primarily studied agricultural species under agricultural settings and so limits the usefulness of the findings in relation to the predominantly non-agricultural Arctic where semi-natural systems are of more significance. Uchida (2007) made the salient point that although no systematic differences were observed between the regions, the variation in transfer factor values around the averages determined by the study were greatest for sub- and tropical environments.

## 6.3.3. Specific climate vulnerabilities for Arctic radioecology

In contrast to more temperate ecosystems, the Arctic soil compartment and systems linked to it has not, with one obvious exception, been the focus of enough attention in relation to radioecological processes. The Arctic therefore is badly positioned with respect to attempting to predict the potential impacts of climate change on the behavior of contaminant radionuclides despite the fact that it is well established that the Arctic is more vulnerable to such contaminants than other climatic regions. The exception has been the large amount of effort regarding the deposition of $^{137}$Cs in the Arctic environment, and to a lesser extent $^{134}$Cs and $^{90}$Sr and their subsequent uptake into Arctic food chains. In most Arctic regions, $^{137}$Cs penetration in soil is minimal, despite the main deposition of this nuclide outside of Chernobyl-affected areas having occurred almost 50 years ago in the 1950s and 1960s. Cs-137 is usually detected in the Arctic terrestrial environment predominantly within upper soil layers (5 to 10 cm), in association with organic material (which is the most common soil material in most of the Arctic) rather than in the underlying mineral horizons, as is more typical for temperate soils (e.g., Baskaran et al., 1991; Schimmack and Bunzl, 1992; Strandberg, 1997). The downwards penetration of cesium in soils has been described as a function of latitude, the amount of precipitation at the time of deposition, the characteristics of the present vegetative cover, the edaphic nature of the soil and the extent of bioturbation by soil organisms (Bergman, 1994; Strandberg, 1997). For radionuclides other than $^{137}$Cs, there is little information concerning their behavior in Arctic soils and a dearth of contemporary data in general. Arctic vegetation and the food chain members that come after it, have received the most attention, due to the intrinsic relationship between Arctic indigenous peoples and the animals that feed upon this vegetation, within the context of the Chernobyl accident on areas within the Arctic. Certain peculiarities of Arctic flora have contributed to their being recognized as efficient bioaccumulators of contaminants and $^{137}$Cs activity levels in Arctic vegetative species are among the highest in the world. Short growing seasons have led to Arctic plants having evolved longer life spans to facilitate the accumulation of sufficient energy to reproduce. This long life span confers greater opportunity for the uptake of contaminants by Arctic plants as compared to temperate species. Arctic plants tend to absorb nutrients directly from the air due to a com-

bination of physical characteristics and the shallow nature of the active soil layer thereby leading to a greater susceptibility for airborne contamination. In addition, Arctic species tend to retain leaves over winter, so absorbed contamination is not purged by leaf fall, but rather stored within the plant tissues from season to season.

Specific chemical reactions, of radioecological consequence, are impacted by the extreme cold of the Arctic. Two of the most relevant are precipitation-dissolution and cation exchange. Ice formation with resultant solute exclusion can result in the formation of supersaturated soil solutions, promoting secondary mineral precipitation within the soil layers. This can result in increased weathering of the underlying lithology with subsequent alteration of soil chemistry processes as demonstrated by Sletten (1988) and Hallet (1978). Carbonate chemistry, of significance in the soil radiochemistry of U and Th series radionuclides among others (Ivanovich, 1994) is affected by the freezing of Arctic soils leading to precipitation of insoluble calcium carbonate ($CaCO_3$) and an increase in ratio of magnesium and calcium ions ($Mg^{2+}$ : $Ca^{2+}$) in the solution phase (Vlasov and Pavlova, 1969). Of all the soil processes affected by the Arctic climate, perhaps the most important in the field of radioecology, are ion-exchange reactions, responsible for pH buffering, ionic transport and metal binding. Cation exchange capacity (CEC) appears to be relatively unaffected by soil freezing (Hinman, 1970; Polubesova and Shirshova, 1992) although elevations in exchangeable K and ammoniacal nitrogen ($NH_4$-N) have been observed. Freeze-thaw cycles can result in the liberation of crystal lattice bound K (Graham and Lopez, 1969) and freezing has been observed to increase pH via adsorbed bases while thawing results in a lowering of pH (Fedorov and Basistyi, 1974). Furthermore, the mobility of radionuclides within Arctic soils is subject to climatic effects, the consequences of which can be contrary to what may be expected in temperate soils. Supersaturated solutions formed by solute exclusion during the freezing process can lead to freezing-point depression and increased levels of unfrozen water in the soils and liquid films on the surface of soil particles can be the dominant pathway for the flow of water and associated dissolved substances (Hoekstra, 1969; Murrman, 1973). Monovalent ions have been reported to be more mobile within frozen soils (Czurda and Schababerle, 1988), due probably to a reduced attraction to charged surfaces within the soil column. An aspect of the chemistry of metallic Arctic soil contaminants is their ability to move from the soil into overlying snow. This was demonstrated by Ostroumov et al. (1992) who determined via laboratory studies that solutes can move into snow layers in a process related to cation adsorption onto ice particles and Kadlec et al. (1988) who established that the freezing of wetlands and organic soils causes many solutes to move into the top soil layers. Evidence for upward migration of radionuclides has been suggested by the recent work of Schuller et al. (2002) describing the movement of $^{137}$Cs in frozen soils. The importance of the movement of contaminants between water or snow and soil during freeze-thaw processes becomes apparent during thawing, with the release of solutes in meltwater. During winters with little or no snowmelt before spring, Johannessen and Henriksen (1978) ascertained that 50% to 80% of the winter pollutant load is released with the first 30% of meltwater. Soil mobility is a feature of the

Arctic region that has a direct bearing on the distribution of deposited radionuclides within this matrix. Soil horizons in Arctic soils can undergo severe distortion due to the action of processes such as frost heave. Tedrow and Walton (1977) demonstrated that differential movement of soil particles from cyclical freeze-thaw action can disturb soil profiles, an effect that has implications for the vertical redistribution of the radionuclide load.

# 6.4. Arctic marine radioactivity and climate

The most prominent change anticipated for the 21st century in the Arctic Ocean is a significant increase in surface air temperature (SAT). Climate scenario experiments performed with coupled atmosphere-ocean sea ice models project this increase to be much larger than the global mean, leading to the so called 'Arctic amplification' (Serreze and Francis, 2006). For the end of the century, the average SAT increase north of 60° N from projections of 14 model ensembles participating in the fourth assessment report (AR4) of the IPCC (IPCC, 2007) ranges from 2.5 °C to 7 °C relative to the period 1980 to 1999 (Chapman and Walsh, 2007). The projected Arctic amplification of the SAT increase, as well as a projected decrease of sea level pressure (SLP) over the entire Arctic are both robust features in IPCC climate model experiments (ACIA, 2004; Chapman and Walsh, 2007). The SLP decrease over the Arctic in the 21st century projections is associated with an intensification of the Northern Annular Mode (NAM) and leads to an increase of atmospheric circulation patterns which are dominated by anti-clockwise motion. This corresponds to more intense low pressure systems in the North Atlantic storm tracks (Cassano et al., 2006).

## 6.4.1. Anticipating changes

### 6.4.1.1. Sea ice

With respect to Arctic sea ice, the IPCC-AR 4 simulation results confirm a strong reduction in seasonal ice cover. For a subset of 20 IPCC models the decrease in total ice area by 2050 is consistently larger than 40% in summer and in the marginal seas in winter (Overland and Wang, 2007). The consequence is an increase in the seasonal amplitude of ice cover during a warming climate. For the end of the century, half of 14 IPCC models show an ice-free Arctic Ocean in late summer (Arzel et al., 2006). The sea-ice cover retreat does not necessarily occur at a constant pace. Strong ice-loss events seem possible, such as short periods of very fast reduction driven by wind events or warm inflow events (Holland et al., 2006). Importantly not only ice extent, but also ice thickness is expected to shrink continuously (Arzel et al., 2006), which together with less multi-year ice occurrence leads to higher vulnerability of the ice cover (Serreze et al., 2007). An example for the combined detrimental effects of increased vulnerability due to thin sea ice occurred in 2007. Warm air temperatures and unusual wind fields lead to a record minimum of September ice extent (Overland et al., 2008; Zhang et al., 2008). The reduction of ice covered area, especially in summer, and the projected change from the thicker multi-year

ice to the thinner seasonal ice cover in large areas will lead to a strong reduction of ice export from the Arctic Ocean to the Nordic Seas (Holland et al., 2006; Bethke et al., 2006). A further consequence of the reduced ice cover is the increase of the open water fraction and thus an increase of wind-fetch on the ocean surface. This is supposed to intensify wave activity, coastal erosion and resuspension of sediment (Serreze et al., 2007).

### 6.4.1.2. Dense water formation

Less clear than changes in the sea ice cover are the anticipated changes in dense water production. This transformation of water masses is initiated by cooling of surface waters and/or ice formation associated with salt-brine release. Both processes increase the surface density and lead to vertical convection, a downward transport of surface water. Convection considerably affects the vertical distribution of radionuclides in the water column. The vertical displacement of dissolved radionuclides may shift them into a completely different circulation regime and alter the direction and speed of transport. There are presently three regions where the main dense water formation or convective overturning occurs: the Labrador Sea, the Greenland-Iceland-Norwegian Sea and the Barents Sea. Since dense water formation in these areas depends on the vertical stability of the water column, it is controlled by large-scale processes like atmospheric cooling and the transport of salty Atlantic water, freshwater and ice.

Anticipated changes in this complex system and associated consequences for radionuclide transport are difficult to assess, because the occurrence of convection depends on the relative importance of competing processes. For example, the anticipated reduction in ice export from the Arctic would counteract the expected increase in liquid freshwater export (Serreze et al., 2007).

Despite an anticipated decrease in deep overturning and a reduction in the Atlantic meridional overturning circulation (AMOC) intensity (IPCC, 2007), vertical exchange in the upper 600 m of the water column of the Nordic Seas is likely to increase (Bryan et al., 2006). For the Arctic Ocean, results indicate an increase in heat import by advection from the Nordic Seas and an increase in convection along Siberian shelves, where ice cover is reduced (Bitz et al., 2006). This is associated with an increase in overturning in the upper 1000 m locally in the Arctic Ocean. The overturning circulation reaches further into the Arctic Ocean and the northward volume transport of Atlantic Water is increasing (Bitz et al., 2006). As a consequence of the increased lateral heat flux, ice production in the Barents and Kara Seas decreases whereas in the central Arctic Ocean and the East Siberian Sea the ice production increases due to the thinner ice (Bitz et al., 2006).

### 6.4.1.3. Precipitation and river runoff

For precipitation the projections of Arctic climate change in the 21st century suggest an increase of 30% to 40%, with the largest relative increase over the central Arctic Ocean (IPCC, 2007). The anticipated increase of precipitation over the drainage basins results in an increase in river runoff to the Arctic Ocean of the order of 20% over the 21st century (Hol-

land et al., 2006; Bethke et al., 2006; Koenigk et al., 2007). Downscaling results with a regional coupled model indicate an increase in extreme precipitation events for a warmer climate, especially in the Barents Sea and the East Siberian Sea (Saha et al., 2006).

## 6.4.2. Arctic Ocean circulation and transport of radioactivity

A major marine pathway for input of artificial radionuclides to the Arctic Ocean is via the inflow of water from the Atlantic Ocean (AMAP, 2004a). Most important sources on this pathway are the European reprocessing facilities at Sellafield (UK) and La Hague (France) and the outflow of Chernobyl-fallout derived radionuclides from the contaminated Baltic Sea (Kershaw and Baxter, 1995; Macdonald et al., 2005). These sources feed into the Norwegian Coastal Current and the Norwegian Atlantic Current. One branch of the Norwegian Atlantic Current flows into the Arctic Ocean through the Fram Strait and intrudes into the Eurasian Basin. It subducts under the fresh and cold layer of Polar Surface Water, separated by a strong vertical salinity gradient, the halocline, and follows the continental slope to the east.

The other branch of the Norwegian Atlantic Current, mixed with the Norwegian Coastal Current, enters the Barents and Kara Seas where it encounters large heat loss and ice growth leading to dense water formation. This process carries a considerable fraction of these waters to the bottom of the shelf. From here the water cascades down the slope of the shelf into the Eurasian Basin where it joins the Fram Strait branch water (Schauer et al., 1997). At intermediate depths (200 m to 800 m) this Atlantic Water Layer circulates on anticlockwise loops along the steep topography (Rudels et al., 1994). The timescale for the passage of the intermediate water masses through the Eurasian Basin or the Canadian Basin back to Fram Strait is of the order of two to three decades (Smethie et al., 2000; Karcher and Oberhuber, 2002). For the deeper water masses, which are also fed by dense water from the Barents Sea shelf, it is of the order of centuries (Schlosser et al., 1999). The temporal scale of these advection pathways determines the time at which soluble radionuclides which once entered the Atlantic Water Layer of the Arctic, will flow out to southern latitudes through Fram Strait (Smith et al., 1999).

That fraction of Barents Sea water which remains less dense stays in the upper part of the water column and contributes to the Polar Surface Layer and the halocline. Here it mixes with runoff originating from rivers and ice melt water (Rudels et al., 1999). The rivers constitute a second source for dissolved and particle-bound radionuclides that enter the Arctic marine environment (JRNC, 1993). A large fraction of the particle-bound radionuclides is deposited in the river mouths or close to the estuaries (JRNC, 1993; Nies et al., 1998), while dissolved radionuclides and those bound to fine suspended material are carried out onto the shelves.

The water masses which leave the shelves form the Transpolar Drift, a current system which redirects runoff and other surface water to Fram Strait and into the East Greenland Current. The Transpolar Drift extends from the Laptev or East Siberian Sea along the Lomonosov or Alpha Mendeleev Ridges, respectively, to Greenland. The exact location of the Transpolar Drift depends on the atmospheric circulation in the Arctic (Polyakov and Johnson, 2000). The Transpolar Drift separates Atlantic-derived water on the Eurasian Basin side from water on the Canadian Basin side. The latter is influenced by Pacific water which entered from Bering Strait and which circulates with the large clockwise rotating Beaufort Gyre at the surface. This gyre slowly releases water to the Canadian Archipelago and to the US/Canadian flank of the Transpolar Drift. Both, position and intensity of the Transpolar Drift and the Beaufort Gyre are variable in time and affect the distribution of Atlantic- relative to Pacific-derived water (Karcher et al., 2005; Newton et al., 2008) and the residence times for surface water in the Arctic.

Contamination with radionuclides also occurs by fallout from the atmosphere, either as dry deposition or as precipitation. For this source of contamination the distribution of ice-covered versus open ocean areas in the Arctic determines its further fate. While for contaminants deposited on open ocean areas the surface circulation acts as the dominant pathway, those deposited on ice are transported with the ice drift, possibly over large distances and even out of the Arctic Ocean proper. Sea ice is also known to incorporate suspended matter during freezing, a process which constitutes a potential pathway for particle-bound radionuclides (Pfirman et al., 1995, 1997b; Dethleff et al., 2000).

The largest production of sea ice occurs on the shelves, from which the ice is exported towards the central Arctic Ocean. The motion of the sea ice is similar to the surface water circulation, with the Transpolar Drift and the Beaufort Gyre the most prominent large-scale features (Pfirman et al., 1997a). The sea ice leaves the Arctic Ocean through Fram Strait into the Nordic Seas and through the Canadian Archipelago to the Labrador Sea. While salt release during ice freezing destabilizes the water column, the release of freshwater during ice melt stabilizes the upper water layers. The melt of contaminated sea ice thus forms a source for radioactivity in ocean surface water (Pfirman et al., 1995). The most important areas for ice melt in the present day climate are the Nordic Seas and the Barents Sea.

## 6.4.3. Consequences for Arctic marine radioactivity

### 6.4.3.1. Ocean transport

Owing to stronger winds (Furevik et al., 2002) and reduced ice cover in the 21st century, together with intensified dense water production (Bryan et al., 2006, Bitz et al., 2006) an increase in the ventilation intensity of mid-depth water masses in the Arctic Ocean is anticipated. This would lead to a reduction of the age of the water at mid-depth in the Arctic. In the Nordic Seas, intensification of ventilation is projected to reach to 600 m depth. Mid-depth water from both basins, the Arctic Ocean and the Nordic Seas, contributes significantly to the overflows. Therefore, reduced ventilation ages can also be expected south of the overflow sills in the deep water which is fed by the overflows (Bryan et al., 2006). As a consequence of such changes an increased downward transport of dissolved radionuclides or those attached to suspended matter can be expected. This holds, for example, along the Atlantic water pathway in the Nordic Seas, the

Barents Sea and the Eurasian Basin of the Arctic Ocean. A relative increase in the radioactive inventory at mid-depth relative to the surface layer would result. Because the timescale for the circulation of water at mid-depth in the Arctic Ocean is considerably longer than for the Polar Surface Layer (Schlosser et al., 1995), an increased residence time for the radionuclides circulating at mid-depth as compared to the surface would follow. Therefore, the Atlantic Water Layer of the Arctic Ocean is likely to intensify its role as a temporal buffer for radionuclides in comparison to the present. In contrast to the Arctic and the eastern Nordic Seas, for two other important dense-water production areas, the Irminger Sea and the Labrador Sea, a reduction in winter-averaged mixed layer depth and reduced deep-water formation rates are anticipated (Bryan et al., 2006). Here, a reduction in downward mixing of radionuclides originating from the surface must be expected.

Without doubt, the anticipated changes in atmospheric forcing, ice cover and hydrography will affect the patterns of exchanges between the ocean basins, as well as interior circulation regimes. This will have important consequences for the inter-basin exchanges of dissolved radionuclides. However, few investigations based on IPCC-type model experiments have been analyzed with respect to the horizontal exchanges between the basins in the Arctic and sub-Arctic. A deduction of horizontal exchanges from the intensification of the northernmost branch of the AMOC (which is projected for the 21st century) is not possible because the AMOC only describes the vertical circulation of the water. ECHAM5/MPI-OM ensemble simulations (Koenigk et al., 2007; Wu et al., 2008) projected a 20% to 30% increase in net volume inflow to the Barents Seas and net volume outflow through Fram Strait for the second half of the 21st century. There is also a slight increase in the net volume outflow from the Arctic Ocean to the Davis Strait and Labrador Sea of the order of 10%. Other model results based on $CO_2$ increase experiments with the Bergen Climate Model (Bethke et al., 2006) confirm the increase in Barents Sea throughflow in a warming climate and also indicate that inflow and outflow through Fram Strait may increase.

A careful interpretation may therefore find support for an intensification of the horizontal Arctic Ocean circulation loop linking the inflow through the Barents Sea and the outflow through the Canadian Archipelago. Such intensification of the Barents Sea branch would have consequences for radionuclide transport with the Atlantic Water and the Norwegian Coastal Current, such as from release from the European reprocessing facilities or the Baltic Sea outflow due to fallout from the Chernobyl accident (Smith et al., 1999). The intensified Barents Sea branch circulation loop also suggests that transport of radionuclides through the Arctic Ocean could occur faster. However, the increased intensity of this loop could be offset by changes in circulation patterns inside the Arctic. The anticipated increase in atmospheric circulation dominated by anti-clockwise motion is likely to lead to a smaller Beaufort Gyre and a shift of the Transpolar Drift from the Lomonosov Ridge far into the Makarov and Canadian Basin. Such a response of the Arctic Ocean to anti-clockwise atmospheric circulation has been observed in the past (Johnson and Polyakov, 2001; Polyakov and Johnson, 2000). It would lead to an elongation of the pathway of radionuclides travelling in the Polar Surface Layer and the halo-

cline and to a larger fraction of radionuclides from Atlantic marine sources in the Canadian Basin surface water.

The enhanced volume inflow to the Barents Sea anticipated for the 21st century increases the exchange between the shelves and the deep basins. In addition, this exchange is expected to be further enhanced by intensified wind-induced upwelling and downwelling at the shelf break, partly due to stronger winds and partly due to a reduced period of ice cover in the season (Carmack and Chapman, 2003; ACIA, 2005). These processes are relevant for the transport of radionuclides entering with Atlantic Water from the south and those which enter the shelf directly, stemming for example from runoff or the dump sites in the Kara Sea (JRNC, 1993; IAEA, 1999). The intensified vertical mixing due to increased storm activity and more open water may also lead to an increased resuspension of sediments, with consequences for the remobilization of radionuclides with sediments or from sediments into the liquid phase (Schiedek et al., 2007). Important areas in this context are the dump sites in Novaya Zemlya bays (Harms and Povinec, 1999) and former nuclear test sites like Chernaya Bay (Smith et al., 2000). Mobilization of sediment-bound radionuclides also depends on the ambient salinity (Oughton et al., 1997). However, it is not possible to anticipate details on the future bottom salinities near contaminated sediments of the Siberian shelves, because competition is anticipated between increased salty Atlantic Water inflow and increased runoff and less salt input by freezing.

### 6.4.3.2. Transport by sea ice

Incorporation of particle-bound radionuclides into freezing sea ice constitutes a shift between transport media. This process has been shown to be important, for example on Eurasian shelves (Dethleff et al., 2000) close to the Kara Sea estuaries of the Ob and Yenisei and near the Kara Sea dumpsites (JRNC, 1993). In the 20th century sea ice formed on the Siberian shelves was likely to survive several seasons to be exported through to the Greenland Sea via Fram Strait (Pfirman et al., 1997b; Harms et al., 2000; Pavlov et al., 2004). This seems to be unlikely for the next couple of decades. The anticipated decrease in summer ice cover in the central Arctic will strongly decrease the chance for sediment in sea ice to survive the summer following its incorporation. Only sea ice which is formed close to Fram Strait can be expected to leave the Arctic Ocean before melting. For sediments incorporated into freezing ice on the shelves it may even not be possible to leave the shelf. In the Kara Sea for example, ice transport from the inner Kara Sea into the deep basins may need more than a year (Nies et al., 1999; Harms et al., 2000). The sediment would be released in summer being still on the shelf. However, resuspension and repeated incorporation into sea ice in the next winter may be possible. A 'multi-hopping' scenario could establish in which contaminated sediment is redistributed on the shelf over a period of several freeze/melt seasons. This anticipated redistribution would make the transport pathway even more complex than at present. On reaching the deep Arctic basin this cycle would stop because the sediment may sink to depth horizons where surface incorporation is no longer possible.

### 6.4.3.3. River water

Whether the anticipated increase in river runoff from the drainage basins to the Arctic Ocean will lead to an increased input of potentially contaminated sediments from Siberian rivers is still unclear. There are increasing chances for riverine radionuclides to reach farther east, for example, into the Makarov and Canadian Basins, with increasing cyclonicity of the atmospheric and oceanic circulation, similar to the 1990s when high Arctic Oscillation (AO) conditions led to increased flow of river water to the East Siberian Shelves and to those basins beyond the Lomonosov Ridge (Dickson, 1999; Johnson and Polyakov, 2001; Schlosser et al., 2002; Karcher at al., 2007).

### 6.4.3.4. Precipitation

Arctic climate projections for the 21st century suggest an increase in precipitation of the order of 30% with the largest relative increase over the central Arctic Ocean (IPCC, 2007). Such weather conditions in the Arctic and Nordic Seas would favor the deposition of radionuclides, as was the case for $^{137}$Cs from the Chernobyl accident in the late 1980s (Macdonald et al., 2005). However, as a consequence of the decreased summer ice cover, deposition of radionuclides on the ice/ocean surface by dry or wet deposition will have a decreasing chance to hit the ice, especially in summer. As for contaminated sediment incorporated into sea ice, radionuclides from fallout travelling with sea ice will have a smaller chance of surviving the melt season. As a consequence a larger input of radionuclides into the Arctic Ocean surface layer, either directly or via ice melt, as compared to the 20th century situation can be expected.

Despite agreement on several basic aspects of climate change, there remains a considerable spread in the IPCC class model that results in many regional features. Because a number of processes relevant for the marine transport of radionuclides, such as convection, depend on subtle balances, uncertainty about the consequences of climate change remains high. Thus, conclusions must remain provisional. Several anticipated changes that exhibit some robustness, when comparing different model experiments are discussed, however.

Most importantly, a reduction in ice cover, especially in summer, will lead to less significance for sea ice as a transport medium for radionuclides from atmospheric fallout and from incorporated sediment. This is likely to lead to a reduced export and a larger fraction of the radionuclides remaining within the Arctic Ocean.

For radionuclides imported with water from the Nordic Seas, the anticipated increase in vertical mixing and convection in the Arctic Ocean could lead to an increased fraction of radionuclides in the mid-depth water. This would increase residence time in the Arctic, compared to surface transport. As a consequence of increased anti-clockwise motion in the Arctic, favoring the Atlantic Water pathway to reach further into the Canadian Basin, a larger fraction of radionuclides transported with the surface water from the European side might enter this part of the Arctic.

This overview has been based on known sources of radionuclides. New sources may arise, however, in part due to the changes in physical conditions in the Arctic. For example, the possible increase in ship transport due to a longer navigation season or the opening of northern sea routes (Mokhov et al., 2007), which may involve transport of reprocessing waste, or the use of ships with nuclear propulsion systems.

## 6.5. Uptake of radioactivity

### 6.5.1. Freshwater environment

The physicochemical characteristics of freshwater systems such as lakes, streams and rivers vary extensively within the Arctic region. Freshwater ecosystems in the Low Arctic can be quite productive, remaining ice free for much of the year; a situation starkly contrasted by the ice-bound lakes and rivers of High Arctic areas, where ice cover may attain thicknesses of up to 3 m for long periods. Ice cover and the spring thaw dominate the ecology of these freshwater systems, with productivity constrained primarily by the low temperatures and lack of light and nutrient (and some contaminant) inputs originating primarily from contributions during the spring snow melt. The formation of ice can result in the underlying water being enriched in contaminant species by partial exclusion of solutes or the export of contaminants to distant locations by ice rafting. Prior to the spring melt, the snow pack accumulates such contaminants as may be deposited on it over the course of the previous winter or as may be introduced via processes such as upward migration of contaminants from the soil into the snow pack due to freezing. In the spring, the rise in temperatures can cause accumulation of these contaminants in the melt water, with the initial 20% to 30% of the melt water containing as much as 60% of the contaminant load of the snow pack. The nature of soils in the Arctic and the presence of permafrost results in the majority of this spring melt water entering the freshwater and marine ecosystems. In contrast to temperate regions, surface waters in Arctic areas are more vulnerable to radioactive contamination, as the proportion of water in a freshwater Arctic body that is derived from groundwater is substantially less than in a temperate water body. Rivers and streams in the High Arctic are usually short and fed by snow or glacial melt water. They may provide habitats for some vegetation, insect larvae and fish, although strong, periodic flow can reduce the number of species present. Low Arctic rivers and streams may provide more stable environments with greater species diversity as in Fennoscandia. Rivers constitute an important vector for radionuclide contamination in the Arctic terrestrial environment. Industrial facilities are often located near rivers or within their catchments and rivers have historically been used as repositories for a variety of wastes. The long courses run by many rivers means that contamination originating from non-Arctic regions can easily reach the Arctic; the situation concerning the Yenisey and Ob Rivers in Russia being a particular example. River ice can incorporate bottom sediments and associated contaminants, and transport these contaminants long distances or remobilize the contamination back into the water column for uptake by flora and fauna. This movement of sediment is associated with the role of lakes as contaminant sinks via the accumulation of sediment and particles.

Terrestrial aquatic food chains, which are typically a little more complex than non-aquatic terrestrial chains, are founded on free floating and benthic algae that are able to photosynthesize, depending on light conditions, between April and the end of the Arctic summer. These in turn are preyed upon by zooplankton, the number and diversity of which are largely a function of location and temperature. Insect larvae and crustaceans may play roles in the food chain but the final trophic level within the aquatic system is usually occupied by fish. Fish types in Arctic freshwaters vary from region to region but may include Arctic char (*Salvelinus alpinus*), brown trout (*Salmo trutta*), northern pike (*Esox lucius*) and grayling (*Thymallus* spp.). Species feeding on freshwater fish include mink (*Mustela vison*), otters (*Lutra lutra*) and a variety of raptors. Freshwater Arctic systems typically accumulate radioactive contaminants from direct deposition of global fallout and from run-off containing previously deposited fallout and natural radionuclides leached from surrounding catchments. A regionally important third source is waterborne discharges from nuclear facilities, such as Mayak PA in the Urals, where major discharges of $^{90}$Sr occurred in the 1950s. $^{90}$Sr is one of the most mobile radionuclides entering freshwater systems, because unlike many radionuclides, including $^{137}$Cs, it is not significantly retained by soils and sediments. The particle reactivity of $^{137}$Cs accounts for its faster rate of elimination from freshwater systems compared to $^{90}$Sr. Where drinking water is mostly derived from ice and snow, for example in Greenland, a reservoir of fallout radionuclide contamination can develop, contaminating water supplies for longer than water supplies derived solely from surface water.

The accumulation of radionuclides in fish from freshwater systems depends on many factors. The uptake of $^{137}$Cs for example, is largely determined by nutrient levels, the overall size of the catchment area of the watercourse or body and the total water volume. Radionuclide activity levels in fish dwelling in highly biologically productive lakes tend to be lower than levels found in fish that reside in lakes whose productivity is lower. The activity levels in fish tend to reflect their position in the food chain and their dietary habits. After the Chernobyl accident, the highest activity levels were first observed in planktiverous fish, with activity levels in predatory species such as pike rising after a longer period. Finnish studies conducted after the Chernobyl accident indicate that the maximum transfer of radionuclides to freshwater fish occurs within three years for most typical species (AMAP, 1997). Activity levels in freshwater fish can be similar to those in sheep and wild animals, but are typically lower than those found in reindeer meat and some mushroom species. Dietary studies conducted by AMAP (1997) indicated that freshwater fish from the Russian Arctic (1960 to 1994), contained activity levels of $^{137}$Cs that were about 10-fold greater than for saltwater fish, decreasing from 40 Bq/kg in the 1960s to 15 Bq/kg before Chernobyl and rising post-Chernobyl to 20 Bq/kg. A similar pattern of reduction and subsequent rise post-Chernobyl is observed for radionuclides in Finnish rivers.

No studies have been conducted to date on the potential impact of climate or environmental change on Arctic freshwater radioecology although a number of aspects are of potential consequence. The potential effects of elevated levels of UV radiation on Arctic freshwater bodies with respect to metallic and radioactive contaminants are likely to be highly dependent on the transparency and DOC level of the water. Arctic freshwater bodies may be expected to be affected from increased UV radiation, especially water bodies with high transparency and low DOC levels. It may be anticipated that most species and populations are locally adapted to present-day irradiance levels; however, the vulnerability of plankton in both Arctic and similar alpine waters to light-induced stress and their low repair capacities thereafter has been demonstrated by Luecke and O'Brien (1983), Hebert and Emery (1990) and Hessen et al. (1990). Organisms in shallow ponds, typical of Arctic ecosystems, without depth refugia could, in particular, be susceptible to UV radiation. Combinations of *in situ* and laboratory studies have been performed with phytoplankton in sub-Arctic alpine areas (Hessen et al., 1995; Van Donk and Hessen, 1996), and particularly with zooplankton in the Canadian Arctic (Hebert and Emery, 1990) and Norwegian sub-Arctic and High Arctic at Svalbard (Hessen et al., 1990). The results of this work serve to demonstrate the high susceptibility of flagellum status, phosphorus uptake, growth rate, and cell wall morphology to UV-B radiation. Van Donk and Hessen (1995) reported that UV-radiation exposure induces cell wall changes that reduce digestibility of phytoplankton for zooplankton. Changes in UV-B radiation levels are likely to have comparatively small effects on biodiversity compared with climate warming. Interactions between UV radiation and radioactive or metallic contaminants with respect to uptake in biota or speciation/behavior have not been elucidated but could merit attention.

Other climatic factors that may have potential effects on freshwater radioecology in the Arctic include temperature and factors related to water quality. Oxygen ($O_2$) solubility in water has an inverse relationship with water temperature, decreasing as water temperatures rise. When oxygen concentrations drop below 2 mg $O_2$/L to 3 mg $O_2$/L, hypoxic conditions are present (Doudoroff and Warren, 2000). Given the thermally-regulated nature of fish metabolic rates, increases in environmental temperature will result in increased oxygen demand, while at the same time the amount of available $O_2$ will be reduced. Thus, fishes exposed to elevated water temperatures can face an 'oxygen squeeze' where the decreased supply of oxygen cannot meet the increased demand. Increased incidence of hypoxia and anoxia in freshwater systems is a likely result of climate change due to the decreased dissolved oxygen concentrations and increased biological oxygen demand that are associated with increasing temperatures (Ficke et al., 2005). Increases in temperature may enhance eutrophic conditions by stimulating explosive macrophyte growth. An overabundance of macrophytes can reduce the amount of fish habitat. This was the case in two Estonian lakes where increases in macrophyte density resulting from eutrophication reduced the amount of northern pike habitat (Kangur et al., 2002). Besides, thermal stratification is a major driving force in determining algal assemblages. Longer periods of stratification create favorable conditions for blue-green algae (George et al., 1990; de Souza et al., 1998; Jones and Poplawski, 1998) which are inedible to most species of zooplankton fed on by planktivorous fish (George et al., 1990).

An increase in fish metabolism due to warmer temperatures also facilitates a faster depuration of toxicants. For example, MacLeod and Pessah (1973) reported that rainbow trout (*Oncorhynchus mykiss*) placed in Hg-contaminated

water and subsequently moved to clean water reduced their body burdens of Hg at higher temperatures: at 20 °C the reduction in Hg concentrations in fish tissue became apparent after 10 days, as opposed to 20 to 30 days for fish held at 5 °C and 10 °C (MacLeod and Pessah, 1973). However, despite their increased ability to metabolize pollutants at warmer temperatures, fishes may still experience increased negative effects at higher temperatures, but these effects may be toxicant-specific. Köck et al. (1996) suggested that inessential metals such as Cd and Pb are difficult for fish to depurate because no specific metabolic pathway exists to process them. Therefore, fish accumulate heavy metals more quickly at higher temperatures. Köck et al. (1996) documented this effect with Arctic char. Fish exposed to Cd and Pb were unable to completely metabolize the metals, resulting in positive correlations between metal body burdens and water temperature, as well as between metal concentrations and the age of the fish.

Within the context of the situations described, freshwater Arctic radioecology would seem to be potentially vulnerable for a number of reasons. Oxygen concentrations in freshwater bodies are important in determining the redox condition of a number of radionuclides and therefore their solubilities etc. Development of oxygen-depleted conditions could be postulated to be of significance in the functioning of sediments in Arctic freshwater bodies as radionuclide sinks and the availability of radionuclides within the water column.

## 6.5.2. Marine environment

The bioavailability of radionuclides (and metallic species) in the marine environment is likely to be subject to the effects of climate change and there is enough evidence, both Arctic-specific and from other regions, to support this idea, although there are insufficient data to actually prove or disprove it. Radionuclide-specific data are scarce and so evidence relating to metals has been included in this section. Before examining direct evidence for climatic vulnerabilities on the biological uptake of radionuclides, it is worth assessing whether Arctic biota tend to display higher levels of accumulation or uptake than biota in more temperate zones. Most recent estimates of radionuclide concentration factors in polar marine invertebrates are comparable to concentration factors in temperate-zone animals. The recent EPIC assessment focused on the Arctic, where extreme physical conditions (temperature, seasonality in light intensity, ice cover) may significantly alter radionuclide transfer to biota (Kryshev and Sazykina, 1986; Sazykina, 1995, 1998). Site-specific radionuclide concentration factors for Arctic marine biota were collated for European Arctic sea areas including the Norwegian, Barents, White, Kara, and Greenland Seas. Concentration factors were collated for Arctic fish, seabirds, marine mammals, zoobenthos, and macroalgae for $^{90}$Sr, $^{137}$Cs, $^{239+240}$Pu and $^{99}$Tc, based on a number of literature reviews for data obtained between 1961 and 1999. For some radionuclide-organism combinations, data for neighboring sea areas (the North Sea and North Atlantic) were used due to the paucity of Arctic-specific data (Beresford et al., 2001).

Estimated concentration factors for $^{137}$Cs uptake by fish (cod, *Gadus brosme*), marine mammals (whales and seals), and macroalgae (*Fucus vesiculosus*) displayed an obvious

time dependence reflecting the slow response of organisms to ambient seawater concentrations. The process of $^{137}$Cs accumulation by Arctic marine biota was not in equilibrium over the long observational time periods considered in this study (Beresford et al., 2001), illustrating the limitations of the concentration factor approach in such assessments. Several tentative conclusions relating to differences between uptake values for Arctic environments and world-average values could be drawn however:

- Concentration factors for $^{90}$Sr in macroalgae, benthos and fish from the Arctic seas appear to be higher than world average values, although for macroalgae this may reflect the types of seaweed studied: a mix of red, green and brown species versus brown species only. For the benthos, two different groups were compared: benthic invertebrates such as annelids and echinoderms, and pelagic zooplankton. Thus, conclusions concerning benthos can only be considered tentative.
- Concentration factors for $^{137}$Cs in invertebrates from the Arctic seas are somewhat higher than average values from generalized world data; however, they are similar to world-averaged values for fish and macroalgae. However, as for $^{90}$Sr, a direct comparison of data for invertebrates and zooplankton may be misleading. For seabirds, the limited extent of available data renders any conclusion concerning similarities or differences in datasets uncertain.
- Concentration factors for $^{239+240}$Pu and $^{99}$Tc in fish, marine mammals, and macroalgae from the Arctic seas show great variability and some concentration factors were higher than the world-average data.

As well as studies comparing radionuclide uptake data for Arctic and non Arctic areas, studies have also focused on the role of temperature and salinity; both accepted climate-vulnerable variables. Although some of the studies reported below concern non-Arctic species in non-Arctic environments, they have been included to illustrate general relationships.

Despite the high level of contaminant inputs to northern marine areas, relatively little attention has been paid in the radioecological (or relevant trace element) literature to the potential effects of low temperatures on the bioaccumulation of radionuclides and other trace elements in northern marine food chains. Because the metabolic effect of low temperatures in ectothermic marine biota could feasibly influence uptake and mobilization of contaminants, the results of studies conducted in temperate zones may not always be applicable to the cold-water marine habitats (Hutchins et al., 1996b; Boisson et al., 1997). Bioconcentration factors for a range of contaminants for Arctic and temperate organisms suggest that significant differences do exist (Fisher et al., 1999). The potential for trophic transfer and bioaccumulation of radionuclides in cold-water ecosystems has been little studied, even though it is clear that the lower metabolic rates of ectotherms and, to some extent, enhanced lipid deposition at colder temperatures could affect contaminant accumulation and retention (Hutchins et al, 1998). Body temperature is a primary factor influencing the metabolic turnover rates of biota (Hemmingsen, 1960; Peters, 1983, 1996; Wen and Peters, 1994; Gillooly et al., 2001). Temperature relationships may also govern the equilibration half-times of at least some

contaminants, in particular methylmercury and radiocesium (Ugedal et al., 1992; Rowan and Rasmussen, 1995; Trudel and Rasmussen, 1997; Forseth et al., 1998). This provides some support for the concept that material turnover is largely controlled by the general metabolic turnover, even in the case of non-essential contaminants and apparently passive uptake (Meili, 2002).

The bioaccumulation of $^{241}$Am, $^{57}$Co and $^{137}$Cs was studied in sea stars (*Asterias forbesi*) by Hutchins et al. (1996a) at two different temperatures; 2 °C and 12 °C. The lower temperature appeared to greatly increase the retention of food-ingested radionuclides; the biological half-life of $^{241}$Am was 31 days at 12 °C, but virtually infinite at 2 °C. Retention of food-ingested $^{57}$Co also increased at 2 °C ($T_{b1/2}$ = 41 days) while $^{137}$Cs was not accumulated from ingested food. Low temperature significantly depressed the net influx rates of $^{137}$Cs from water but did not appear to affect net uptake of either $^{241}$Am or $^{57}$Co. Temperature had little apparent effect on the retention of any of the three isotopes absorbed from the dissolved phase. The results of these studies appear to suggest that radionuclides taken up through trophic transfer may be retained far more efficiently in high-latitude marine biota than by fauna from warmer ecosystems (Hutchins et al., 1996a).

Baines et al. (2005) undertook studies on the potential effects of temperature and geographic origin on the ability of Arctic and temperate blue mussels (*Mytilus edulis*) to take up and retain a range of metals from ingested food (filtered algal food). The results indicated that geographic origin appeared to have little effect on either the percentage of the metal assimilated from food (the assimilation efficiency, AE) or the rate at which the ingested metal was subsequently excreted (measured by the efflux constant, $k_e$). In contrast, experimental temperature appeared to be much more significant, causing the AE for Ag, Am and Zn to be 122% to 945% higher, and the $k_e$ for Cd and Co to be 50% to 80% lower at 2 °C than at 12 °C. The effect of temperature on the trophic accumulation factor (TAF = AE/$k_e$) was even more pronounced and systematic with the behavior of metals characterized by higher TAFs at 2 °C than at 12 °C; the effects being greatest for the non-essential metals Ag and Am (6- to 7-fold), least for the elements Co, Se and Zn (2- to 3-fold) and intermediate for Cd (4-fold). The geographic origin of the test subjects affected the TAF for Cd only, with temperate mussels displaying slightly higher potential for biomagnification. The study appears to indicate that, with the exception of Cd, mussels from temperate and Arctic zones bioaccumulate metals from food in a similar fashion, and that temperature has a much greater effect on bioaccumulation than the origin of the biota. Warmer water can increase growth rates and stimulate ecosystem production. For example, aquatic invertebrates, at the base of the food web may mature more rapidly, to a smaller adult size, and reproduce more frequently (Arnell et al., 1995). While such a notion could lead to the expectation that warmer Arctic temperatures may lead to more invertebrate food being available for fish, warmer water temperatures will also increase the rate of microbial activity and thus the rate of decomposition of organic material, which may ultimately result in less food being available for invertebrates and thus fish (Meyer and Edwards, 1990).

The effects of salinity and temperature on the accumulation of $^{137}$Cs by an estuarine clam under laboratory conditions were studied by Wolfe and Coburn (1970). They reported that concentration factors for $^{137}$Cs in whole soft parts of the clam *Rangia cuneata* decreased with increasing temperature and suggested that this may be due to the chemical similarity of $^{137}$Cs and potassium.

Using the three representative nuclear waste components present in dumped Russian waste ($^{241}$Am, $^{57}$Co, $^{137}$Cs) Hutchins et al. (1998) examined the effects of temperature on radionuclide assimilation and retention by a common member of Arctic benthic community – the bivalve *Macoma balthica*. *Macoma balthica* is widely used as a bioindicator species in temperate ecosystems as well Arctic ecosystems (Luoma et al., 1985; Cain and Luoma, 1990), to determine the kinetics of processes that control uptake from food and water as well as kinetic constants of loss. Uptake and loss kinetics of isotopes for both soft tissues and shell are important, because many predators such as demersal fish and some marine mammals ingest both parts of this soft-shelled bivalve. This represents a potential source of radionuclide exposure to indigenous human populations in the Arctic.

The study by Hutchins et al. (1998) employed relatively short exposures as well as relatively short depuration periods. $^{137}$Cs was obtained from water and $^{241}$Am and $^{57}$Co were obtained from both water and food. For all three radionuclides, body distributions were correlated with source, with most radioactivity obtained from water found in the shell and most radioactivity obtained from food in the soft tissues. The dissection results (Hutchins et al., 1998) supported earlier work suggesting that isotope concentrations found mostly in the visceral mass can be taken as evidence that the primary exposure route is through food, while radioactivity obtained from water sources should be located almost entirely on the shell (Bjerrgaard et al., 1985; Fisher and Teyssie, 1986). Such information is less useful for elements such as Cd which are significantly accumulated in soft tissues from water but not on shells (Wang et al., 1996; Fisher et al., 1996).

Arctic temperatures appear to have only minor effects on the individual processes that govern the kinetics. The results obtained by Hutchins et al. (1998) indicate that the only substantial effect of Arctic temperatures was that colder temperatures reduced the assimilation efficiency, and therefore the uptake of $^{241}$Am, by *Macoma balthica* from diatom food. That is, the assimilation efficiency of ingested $^{241}$Am was significantly higher at 12 °C than at 2 °C; $^{137}$Cs was not accumulated in soft tissue from water during short exposures and was rapidly lost from shell with no thermal dependence; no effects of temperature on $^{57}$Co assimilation or retention from food were observed. The results show that generally, Arctic conditions have relatively limited effects on biological processes influencing bioaccumulation of radionuclides, and that bivalve concentration factors may not be appreciably different in polar and temperate waters.

All other thermal effects on efflux rates and distribution in the different pools of ingested radioisotopes, even when statistically significant were very small.

The common marine mussel *Mytilus edulis* is widely used in monitoring metal pollution in coastal and estuarine waters (Goldberg et al., 1983; De Kock and Kramer, 1994). Therefore, understanding the possible roles of temperature and other environmental factors on metal uptake and accumulation in this species is critical to correctly relate tissue concentra-

tions to those in the surrounding environment. Temperature affects both metal chemistry in seawater (Byrne et al., 1988) and physiology of mussels (Dame, 1996). Temperature affects metal chemistry by changing chemical speciation, pH, solubility, reaction rates and physical kinetics (Byrne et al., 1988; Blust et al., 1994). Theoretical calculations of chemical speciation in seawater indicate that changes in temperature and pH have most effect on strongly hydrolyzed and carbonate complexes and less effect on chlorides and free metal ions (Byrne et al., 1988). Although it is widely known that uptake is largely controlled by the free metal ion, some studies have also shown a significant contribution from other species, particularly metals bound to weak complexes (Campbell, 1995; Hudson, 1998; Lorenzo et al., 2005). Chemical speciation indicates that increase in temperature generally results in increase in the concentrations and activities of bioavailable metal forms, and therefore, enhances uptake.

The uptake and accumulation of Cu, Co, Cd and Pb in *Mytilus edulis* were studied at different temperatures (6 °C to 26 °C). Results from exposure of isolated gills showed a positive relationship between temperature and metal uptake. However, in whole organism experiments, only the accumulations of non-essential metals (Cd, Pb) showed a similar trend while Co and Cu, two essential metals, were independent of and inversely related to temperature, respectively. With the exception of Cu, elimination process appeared to be independent of temperature. The study also showed that neither changes in scope for growth (SFG) of mussels nor chemical speciation could fully account for the observed temperature effects. Overall, these results suggest that fundamentally (i.e., at epithelial membranes), temperature effects on uptake are largely due to changes in solution chemistry and physical kinetics, which favors higher uptake at high temperature. But at the whole organism level, complex physiological responses appear to mask the relationship, particularly for biologically essential metals like Cu (Mubiana and Blust, 2007).

Baines et al. (2005) compared the uptake potential of a mussel species for a variety of radionuclides (Am, Co, Cd, Ag) with respect to both geographic origin (Arctic vs. temperate) and physico-chemical parameters. The research results indicated that the geographic origin of the species had little role in either the assimilation or excretion of radionuclide species, but that temperature played a much larger and more significant role, not only on the two parameters mentioned but also on the trophic accumulation factor.

The net uptake of Zn, Cd, Pb and Cu by the common mussel *Mytilus edulis* exposed to different conditions was investigated with a view to using this species as an indicator of contamination of the marine environment by these metals (Phillips, 1976). The variables studied were season, position of the mussel in the water column, water salinity, water temperature, and the simultaneous presence of all four metals. Each of the five variables affected the net uptake of some or all of the metals studied under some conditions. Seasonal variation in concentrations of Zn, Cd and Cu was found in samples collected at three separate locations.

Near to freshwater inputs of trace metals, the concentrations of Zn, Cd and Pb in mussels were found to vary according to the depth at which the mussels were collected; in summer when freshwater run-off is less, this effect was absent. Low salinities did not affect the net uptake of Zn by mussels,

but increased the net uptake of Cd and decreased that of Pb. Low temperatures had no effect on the net uptake of Zn or Pb; the net uptake of Cd was unaffected by low temperatures at high salinities but was decreased by low temperatures at low salinities. The presence of the other metals had no effect on the individual net uptake of Zn, Cd or Pb. In contrast to the other metals, the net uptake of Cu by the mussels was extremely erratic, and was affected by salinity and temperature changes and by the presence of the other metals and changes in their relative concentrations. The effects of other metals on the net uptake of Cu cannot be easily eliminated or allowed for; it is, therefore, suggested that the mussel should not be used as an indicator of Cu in the marine environment (Phillips, 1976).

Loss of ten radionuclides (radioisotopes of Pu, Am, Np, Eu, Ce, Ag, Tc, Zn, Co and Mn) by a Baltic mussel (*Mytilus edulis*) population was studied near the salinity minimum in the Bothnian Sea (northern Baltic Sea) by labeling mussels in the laboratory and then allowing them to depurate during a 300-day period in the field at two locations: one with normal temperatures, the other with temperatures 8 °C to 10 °C above normal. During winter, the clearest effect of heating was accelerated loss of Ag. Also, Zn was apparently lost more rapidly in warm water, whereas none of the remaining nuclides showed loss rates significantly different from zero at either temperature. At normal temperatures during spring and summer all analyzed elements were lost faster than in the heated winter experiment despite similar average temperature conditions. Loss rates were apparently faster than in full-salinity waters. Plutonium to Am ratios decreased during depuration. The Pu fraction lost after the 300-day period was estimated to be twice the corresponding fraction of Am. Europium to Am ratios remained unchanged for all seasons and temperatures, whereas Ce to Am ratios decreased to half during the initial loss phase, after which they remained unaltered. It was concluded that Eu behaved as an ideal analogue to Am (Dahlgaard, 1986).

Several authors have indicated that temperatures above 20 °C are unfavorable for *Mytilus edulis* (Widdows 1978; Incze et al., 1980; Almada-Villela et al., 1982). This could explain the slower loss rate for warm mussels during spring and summer. Comparison of cold winter values and warm winter values reveals that below 20 °C elevated temperature actually accelerates losses of Ag and Zn, whereas no significant effect is seen for the other elements. Cold summer and spring experiments and the warm winter experiments effectively cover the same temperature range. The present results document a distinctly higher loss rate during cold summer conditions than during warm winter conditions; this cannot be due to a temperature effect as such.

The situation regarding radiological risks in the Arctic was assessed by AMAP in 2002 (AMAP, 2004a). This section provides an overview of how those risks have changed since that assessment, taking account of changes in existing sources and the new sources and issues discussed in previous chapters. Some of the changes are the result of direct action to reduce risks, while others are incidental. It is also the case that the range of sources in the current assessment is wider than that addressed previously, with consideration now given to technically-enhanced naturally-occurring radioactive material (TENORM) as well as to artificial radionuclides.

*Chapter 7*

# Arctic Ecosystem Vulnerability, Human Exposure and Resource Impacts

## 7.1. Management of threats, risks and harm

Before discussing the management of radiological threats, risks and harm, it is useful first to distinguish between radiological threats and the risks that arise from them, and the types of harm that they might cause.

A threat presents a capacity for harm, whereas a risk takes into account how likely that harm is to occur. Sometimes the term 'hazard' is used instead of 'threat'. The point is that radioactive material (or other hazardous materials) may present a threat to human health and/or the environment, but through appropriate management, the chances of that threat actually creating harm can be reduced, for example, to levels that may be judged acceptable or tolerable.

Such management may include steps to reduce the threat itself, by reducing the source term or by removing it altogether; but it may also include a variety of technical measures to reduce the chances of harm arising, for example, by reducing discharges from the source, by reducing the chance of accidents, or by having in place an effective emergency response program so that the harm arising from any releases may be mitigated. Environmental monitoring plays a part, providing an understanding of what is happening as a result of past releases, and providing a test bed for hypotheses about the consequences of future releases. It also makes it possible to check retrospectively that past assumptions about the effects of proposed future releases were correct.

For major remediation projects which have implications for national strategies, the role of strategic planning is very important, as illustrated by the completion of the Phase 2 Strategic Master Plan (SMP) to integrate all Rosatom programs and plans with those of other Russian agencies involved in decommissioning activities, including bilateral and multilateral international agreements (see section 2). The projects organized under the SMP are designed to reduce threats and are carried out within a risk management framework, which allows priorities to be identified and for specific industrial work to be carried out safely and in accordance with the Russian legal framework, taking into account international recommendations and taking advantage of experience of good practice in other countries. The long-term objectives for legacy site management are recognized as only being achievable over a period measured in decades rather than years.

As progress is made towards achieving the planned objectives, the overall SMP serves as an example, and, for north-west Russia, as a reference point for ongoing operational planning. The SMP includes: the incorporation of additional technical information as it becomes available; identification of the necessary legal and regulatory framework; identification of accessible financial resources; and provides the benchmark for new key decision making.

Environmental monitoring clearly provides important input for the iterative process above, and as such should be planned so as to take account of the time frame of the decommissioning projects, now recognized to be at least ten years (see section 2).

Furthermore, as the SMP acknowledges, the program relies on an effective legal and regulatory framework. This implies action to be taken by regulators to ensure that: 1) relevant norms and standards exist, which take account of the vulnerability of the Arctic ecosystem, for application to the sites and facilities of interest; 2) decommissioning projects and the corresponding risk management measures are implemented in an appropriate way; 3) licence conditions are complied with; and 4) environmental monitoring is in place to demonstrate that the planned and implemented measures are working as intended.

The process of identifying the main threats and addressing them in a regulatory context, according to priorities for risk reduction, is illustrated with respect to the rehabilitation of the Sites of Temporary Storage (STS) at Andreeva and Gremkiha; see Ilyin et al. (2005) and Sneve et al. (2007a, 2008). Similar consideration has been given to the decommissioning of RTGs by Sneve and Reka (2007). This work has contributed to the setting of site-specific protection measures, monitoring requirements, enhanced emergency preparedness and response, and the regulatory basis for management for slightly radioactively contaminated industrial waste.

Another aspect related to the control of radioactive material is security. The Finnish and Russian Customs Authorities together with the Nuclear and Radiation Safety Authorities of both countries provide a training program for customs officials, to provide them with security-relevant information on technology, competencies and practices in controlling legal shipments and in detecting illicit trafficking at harbors, land and railway border crossings. This Finnish-Russian co-operation is aimed at maintaining and further developing effective national control of nuclear materials. The focus is on support to regulatory work, training of customs officials and border guards, developing verification measurement tools, and sharing knowledge and experiences regarding the national system and its operations. The spent fuel measurement equipment (SFAT) intended for regulatory inspections at the Kola nuclear power plant, was constructed at STUK (Radiation and Nuclear Safety Authority, Finland), and its functionality was tested at the Loviisa nuclear power plant. The system was demonstrated at the Kola nuclear power plant in 2008.

This type of regulatory co-operation supports the development of common solutions to common problems as well as more effective and efficient supervision of specific projects. In the longer term, it also supports the general development of an enhanced safety culture. While it is vital for each organization to understand and address its own responsibilities, it is also necessary to have a confident communication process with operators, technical support organizations and others.

Progress in such interaction is illustrated by international workshops (Sneve and Kiselev, 2008), which provide a forum for operators and regulators to share experience, for example on developments at sev RAO facilities in the Arctic (Shandala et al., 2008a).

It is necessary to have a common understanding of the word harm. For the purposes of this assessment, harm is taken to include: detrimental impacts on human health, detrimental impacts on the environment, and detrimental impacts on economic resources.

For risk management to be effective, it is necessary to understand the spatial and temporal distribution of these impacts and how likely they are to occur. This means understanding how radioactive contamination is distributed, and in particular: how concentrated it is in particular locations, how this will change in the future, which environmental media are concerned, and how humans and other biota interact with that contamination and those media.

Recent progress in this area is provided in the proceedings of the International Conference on Radioecology and Environmental Radioactivity in Bergen, June 2008 (NRPA, 2008). Development of such scientific understanding and its application within risk assessment is vital to being able to determine priority issues and is itself also a part of risk management. Broader scientific developments in the fundamental understanding of how ionizing radiation interacts with, and affects, living things are also important. All such developments can contribute to plans for risk management.

There are many social and economic factors relevant to evaluation of these detriments; it is not only a matter of radiological protection. Thus, a further complication is that action to reduce one risk may increase another, and a holistic view of the problem is needed to achieve an optimal solution. Strong action to minimize one impact is likely to exacerbate another unless a holistic approach is taken to the problem. Progress in the application of optimization techniques, dealing with a range of relevant attributes, also plays a part in future risk reduction efforts. This is illustrated with respect to rehabilitation of Sites of Temporary Storage (Bylkin et al., 2007).

While a holistic approach is important, practically speaking, large projects must be broken down into manageable components. Accordingly, strategic planning is necessary to allow the synthesis of all the management issues and to support an appropriate risk balancing process. This is also a component of risk reduction and was illustrated by IBRAE (2007).

Protection of the Arctic environment is clearly a multinational issue. Sharing of experience and ideas is bound to produce benefits, both in development of wider cooperation and in how to address issues at the local level. Scientific and technical exchange visits promote information exchange and are another mechanism with potential to lead to risk reduction (NRPA, 2007b).

This demonstrates that the reduction of threats and risks, and the minimization of harm, is complex, relying on technical, socio-economic and scientific inputs. AMAP's role is to supply scientific input concerning environmental monitoring and risk assessment. It is notable that the Bergen conference (NRPA, 2008) included topical sessions on emergency preparedness and rehabilitation (EPR), TENORM , radioecology, risk assessment, the Arctic, speciation radioecology, radiation in society, radioactive waste, and environmental protection.

Noting the above, the following sections are intended to: 1) reprise the nature of the risk profile and recommendations from the previous AMAP AMAP assessment of radioactivity in the Arctic (2004); 2) review international developments in radiation protection standards, in so far as they impact upon the Arctic situation; 3) review the steps taken to reduce risks since the previous AMAP assessment; 4) review changes in the sources and trends in risks; 5) consider the progress relative to the previous AMAP recommendations and 6) ask what are the remaining threats and risks and what are the priorities? As well as, what lessons can be learnt for future planning of AMAP activities in support of risk reduction?

## 7.2. Risk profile and recommendations in 2002

One of the main sources of radionuclides identified by AMAP in its previous assessment (AMAP, 2004a) was releases from reprocessing plants, including those at Mayak (Russian Federation), Sellafied (UK), Dounreay (UK) and Cap de la Hague (France). This concerns continuing discharges, but also the potential for the remobilization of activity already released to the environment that is currently bound up in sediments and other environmental media and has not yet reached the Arctic. Reprocessing plants, along with reactors and spent fuel stores in the Arctic, were regarded as the major potential sources of radioactive contamination. Contamination from reprocessing plants has been widely dispersed throughout the environment and is relatively dilute by the time it reaches the Arctic; it is thus impractical to mitigate any potential impacts from the long-lived radionuclides involved. The previous AMAP assessment gave considerable attention to the then ongoing increase in releases of $^{99}$Tc from Sellafield. Sources and potential sources of radioactivity within the Arctic present similar threats to discharges from reprocessing plants, but also present acute threats in terms of the potential for high local concentrations of radioactivity. Thus the major concerns for potential high environmental contamination relate to nuclear power plants and other large sources within the Arctic.

The previous AMAP assessment concluded that the particular vulnerability of the Arctic to radioactivity related primarily to the scope for bio-accumulation of radionuclides within components of the ecosystems with which humans interact. It was subsequently recommended that AMAP extends its consideration of these issues with respect to environmental protection. The situation was recognized as being different for the marine and terrestrial environments. Contamination of the terrestrial environment following accidental releases to the atmosphere is likely to be patchy compared to marine releases. It was recommended that AMAP be asked to clarify the vulnerability and impact of radioactivity on the Arctic, and to consider the implications for emergency preparedness planning. It is clear that local sources of radioactivity, capable of producing acute but high local concentrations, may be more appropriate for and amenable to mitigation after the event, whereas distant sources can only be ameliorated by reducing the source itself.

The previous AMAP assessment noted that substantial efforts were underway to reduce risks associated with nuclear reactors and radioactive waste handling. Further effort was recommended, so as to lead to a net improvement in nuclear safety. It was suggested that decisions on the optimized application of resources to achieve this objective needed to be founded upon appropriate risk management and risk analysis. This implies having quantifiable measures of harm and an ability to assess them, for example, through the application of risk assessments, taking account of the likelihood of events leading to release to the environment. A prospective assessment of an amount of radioactivity in the environment is not in itself a measure of impact, unless that concentration represents, or can be converted into, a measure of harm – to humans, to the environment and/or to resources. Accordingly, AMAP (2004a) recommended that risk assessments be used to evaluate risks before new activities are implemented. Also, that such assessments should address normal operations and accident scenarios, and should include estimates of uncertainties.

Continued cooperation between authorities was said to be required on the development of initiatives concerning health and safety, and emergency preparedness. Of particular interest were health and safety risks immediately before, during and after a risk-reducing initiative. This recognizes that to achieve a long term safety objective, action may be needed now which could increase risks in the short term, if not appropriately managed. It was noted that, at that time, such cooperation had not been prioritized, although it is not expensive and can contribute significantly to the development of large multi-lateral internationally funded projects. Accordingly, AMAP suggested that a further strengthening of the Russian authorities responsible for nuclear protection would increase their ability to effectively implement improved management practices.

## 7.3. International developments in protection standards and their implementation

Since 2002, the ICRP has issued several important recommendations (ICRP, 2003, 2006, 2007a,b). These have implication for risks and risk management, both for human health and environmental protection.

- Publication 91 (ICRP, 2003) provides a framework for assessing the impact of ionizing radiation on non-human species, and technical development of that framework has been carried out within the EC EPIC and ERICA projects, and its application reviewed within the PROTECT project.
- Publication 101 (ICRP, 2006) provides updated guidance on assessment of doses to critical groups and also provides a broader perspective on the process of optimization. New terminology is introduced, referring now to assessment of the dose to the representative person, as opposed previously to the critical group, for the purpose of radiation protection of the public and the optimization of radiological protection. The document includes significant additional guidance on dealing with uncertainties and the application of probabilistic assessment methods. In addition, it sets out recommendations on assessment of collective doses, and how they should be disaggregated into relevant temporal and spatial components, and interpreted in the light of the individual dose rate at which the collective dose is delivered. These recommendations have separate implications for analysis of local and remote sources of contamination to the Arctic.
- Publication 103 (ICRP, 2007a), the updated basic recommendations, concludes that the risks per unit ionizing radiation dose for humans are marginally lower than previously thought. It also states that it is inappropriate for public health planning purposes, to calculate health effects based on a link between assessments of very small radiation doses received by large numbers of people over very long periods of time and risk per unit dose. However, ICRP were not explicit about the meaning of very small radiation doses in this context.
- Publication 104 (ICRP, 2007b), on the scope of protection control measures, includes new guidance especially in relation to natural radioactivity. It stated that account should be taken not only of the justification and optimization of controlling measures, but also of the different expectations of those affected by those measures. This wording implies a separation of stakeholders' views from a scientific evaluation of the control measures. It is thus not clear if stakeholder involvement is meant to allow non-optimum control measures, or whether it would be better to include stakeholders' views in the determination of the optimum measures. The solution may be to have a clearer distinction between what is claimed as scientific support for a proposition and value judgments made about the implications of the proposition.

Over the same period, the IAEA has consolidated its safety principles into a single document (IAEA, 2006b) addressing all aspects of radiation protection in one place, namely, accidents, existing and routine situations today, and the long term. It also provides the basis for integration of the IAEA's radiation safety standards into a wider safety related program. This provides for the holistic consideration of different kinds of radiation risks, but room remains for the integration of these principles with wider environmental and human health protection principles.

## 7.4. Progress and ongoing threat and risk mitigating activities

The material presented in earlier sections demonstrates material progress with:

- control of planned discharges from remote sources which only have disperse impacts on the Arctic and sources located in the Arctic which could have locally acute and disperse impacts;
- mitigation of accident risks, both the probability and the consequences, taking into account emergency preparedness and response;
- mitigation of existing situations, which include legacy site issues and the continuing effects of past accidents; and

- development of systems for the long-term management of nuclear legacy sites, spent nuclear fuel and radioactive wastes.

Such progress can be measured in terms of: 1) actual reductions in specific threats and risks (see below); 2) enhanced technical coordination of major industrial projects on nuclear legacy management; 3) enhanced coordination of emergency response and preparedness, as well as practical exercises in relevant locations; 4) enhanced security and safeguards; 5) development of updated regulatory requirements and guidance, as well as improved cooperation among regulatory authorities; and 6) enhanced scientific knowledge of the behavior of radionuclides in the Arctic, based on monitoring and analysis.

## 7.5. Trends in threats and risks

### 7.5.1. Threats and risks arising within the Arctic

The number of RTG sources remaining in the Arctic has been reduced substantially. About 50% have been removed or are planned to be removed in the near future, and a future program is under development. In addition, an improved regulatory basis has been developed for supervision of removal, and plans are in place for the safe management of the recovered sources. Thus, the threat presented by the RTG s has been reduced and the likelihood of future accidents and incidents has also been reduced. A more detailed assessment, taking account of the specifics of different RTG types, in different locations, would be necessary for a fuller understanding of the risk reduction.

Further significant progress has been made since 2002 in the reduction of the number of obsolete Russian nuclear powered submarines awaiting defueling and dismantling, with approximately 50 having been completed at a current rate of about ten or more per year. This has been the result of major cooperation between Russia and other countries.

Substantial progress has been made with putting in place the physical and legal infrastructure for the management of submarine spent nuclear fuel and radioactive waste stored at sites in the Arctic, notably at Andreeva Bay and Gremikha (Russian Federation). This includes work at the sites themselves and on transport facilities and at other sites due to receive radioactive materials from the sites. Such work has been necessary before recovery of the fuel and wastes can take place. Similar progress has been made with spent nuclear fuel and radioactive waste associated with nuclear icebreakers and their ancillary facilities, such as the Lepse storage vessel.

The continuation of these activities in the Russian northwest, and a corresponding expectation of a continuing reduction in threats, is supported by the recent development of an updated Strategic Master Plan and a program for its implementation.

Movement of radioactive material in the Arctic is anticipated to increase, as a result of remediation activities and possible new activities, such as new floating nuclear power stations.

### 7.5.2. Threats and risks arising remote from the Arctic

The planned discharges from the European reprocessing plants at Sellafield (UK) and Cap de la Hague (France) have been reduced and this trend is likely to continue through the implementation of the OSPAR Convention. The reprocessing plant at Dounreay (UK) is no longer in operation.

The potential for a large accident resulting in transfer of contamination from reprocessing plants by air or sea to the Arctic remains; however, the volume of high active liquor in store at Sellafield is due to be reduced substantially before 2020.

Activity deposited in bed sediments in the Irish Sea (the reprocessing plant at Sellafield discharges into the Irish Sea) and the Baltic, identified previously as a possible source of contamination of the Arctic, remains a threat. Climate change may have implications for the rate of remobilization of activity from the seabed to the water column.

### 7.5.3. Impacts

The trend in releases and hence in concentrations of activity from the various sources is downwards. Thus, the trend in overall impacts in terms related to total exposures is also downwards. In addition, because of reduced sources, enhanced security, and enhanced safety supervision, the likelihood of impacts from accidents is also downwards. However, significant hazardous operations remain to be undertaken in relation to the management of spent nuclear fuel.

## 7.6. Status of implementation of the AMAP 2002 recommendations

Actions have been taken to reduce discharges from Sellafield to the marine environment. Political pressure resulted in a new purification treatment plant which reduced the discharges of $^{99}$Tc by more than 90%.

A more detailed study of remobilization of activity from marine sediments does not appear to have been carried out.

Implementation of the recommendation on improved access to information on civilian and military sources is difficult to judge. New cooperation with military nuclear safety authorities is a positive step, but security concerns make it more difficult to analyze risks from some sources.

Progress has been made with the development of a basis for protection of the Arctic environment from radioactivity.

Further work might yet be done to clarify the vulnerability and impact of radioactivity on the Arctic environment, especially with respect to TENORM, and the consequences for emergency preparedness.

Substantial progress has been made to reduce risks from nuclear reactors, spent fuel and radioactive waste. This has been largely possible because of continued international cooperation.

Risk assessment has been increasingly used to identify how to address threats and identify priorities. However, there is little evidence of the application of uncertainty analysis in such assessments. To date, priorities have been identified largely on the basis of the physical state and size of the source.

Ideally, each major source would be considered in turn, sub-divided into: 1) risks to human health, at the individual and at societal levels; 2) risks to non-human biota, with emphasis on any areas which are or could be contaminated at levels which exceed derived standards for environmental protection; and 3) risks to resources.

Substantial progress has been made to improve cooperation among authorities, including the participation and strengthening of Russian authorities responsible for nuclear safety and radiation protection, in the areas of worker, public and environmental protection.

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