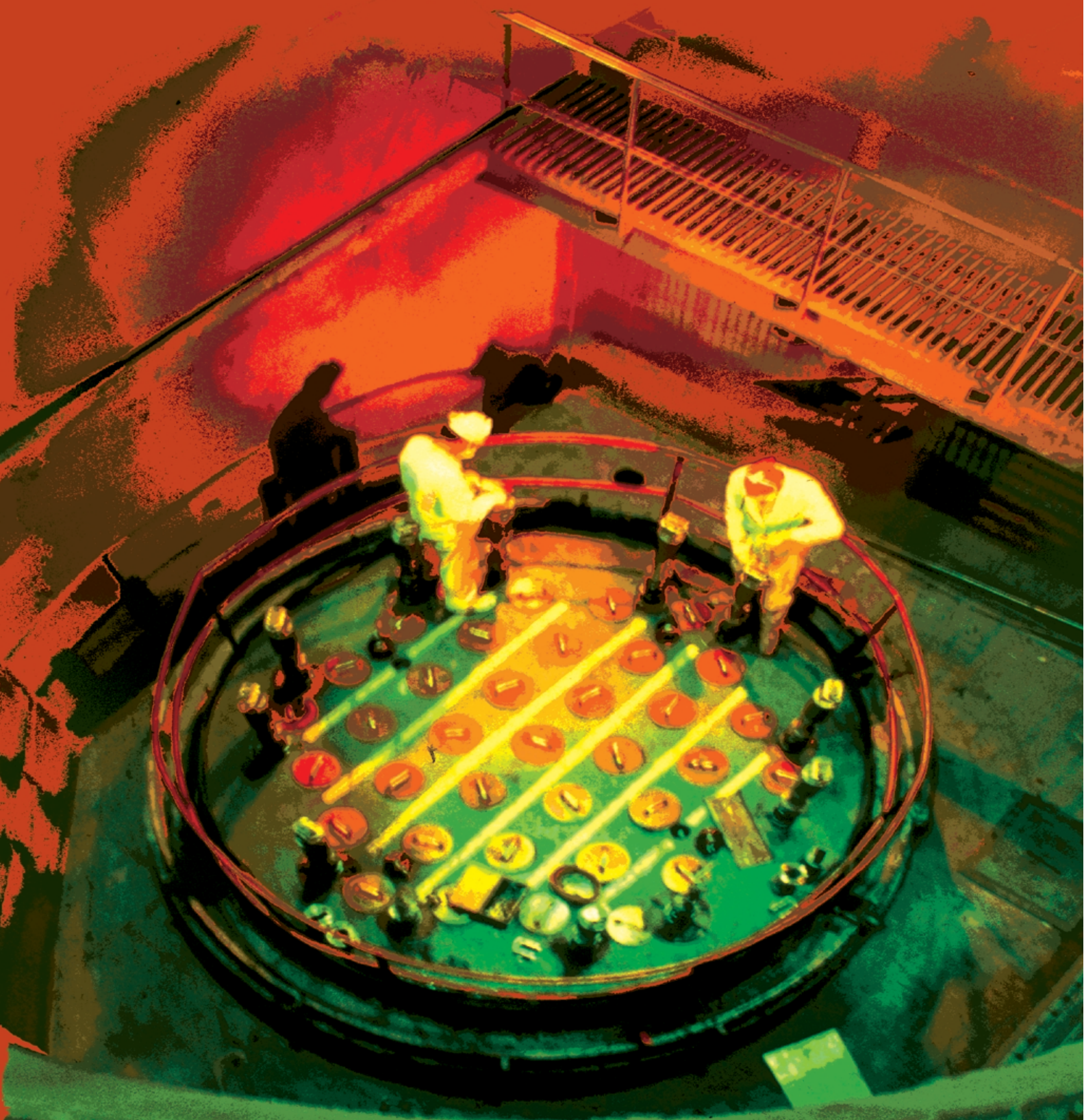
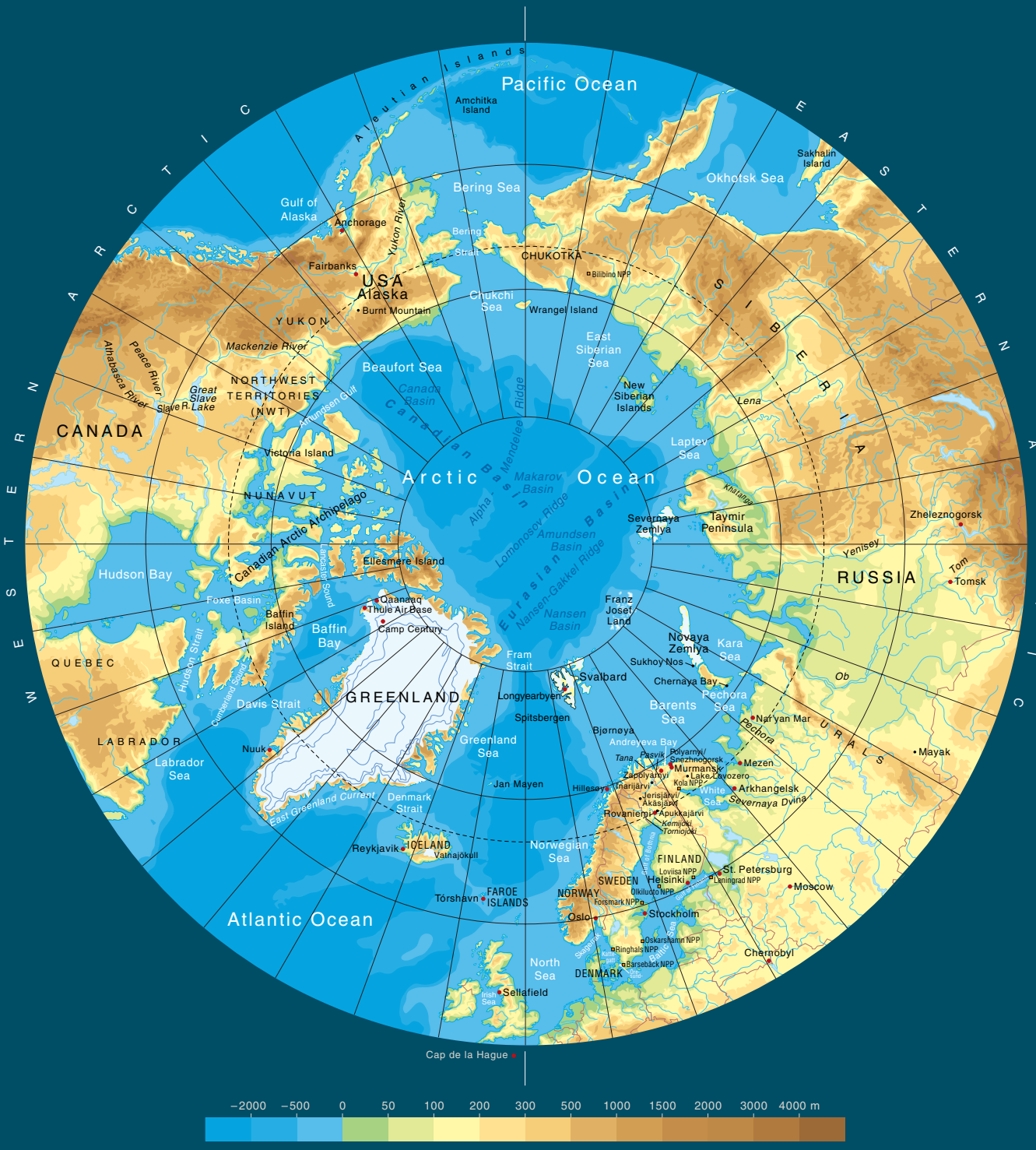
AMAP Assessment 2002:
Radioactivity in the Arctic



Arctic Monitoring and Assessment Programme (AMAP)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Sakhalin Island
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Alaska
Burnt Mountain
Chukchi Sea
Wrangel Island
YUKON
Mackenzie River
Athabasca River
Peace River
Great Slave Lake
Slave R.
NORTHWEST TERRITORIES (NWT)
Beaufort Sea
Canada Basin
Amundsen Gulf
Canadian Basin
Mendeleev Ridge
East Siberian Sea
New Siberian Islands
Lena
SIBERIA
CANADA
Victoria Island
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Makarov Basin
Alpha Ridge
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Khatanga
NUNAVUT
Canadian Arctic Archipelago
Lomonosov Ridge
Amundsen Basin
Severnaya Zemlya
Taymir Peninsula
Zhelezngorsk
Yenisey
Nansen-Gakkel Ridge
Eurasian Basin
Hudson Bay
Ellesmere Island
Nansen Basin
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Tomsk
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Foxe Basin
Nares Strait
Qaanaaq
Thule Air Base
Baffin Island
Camp Century
Baffin Bay
Franz Josef Land
Novaya Zemlya
Kara Sea
QUEBEC
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Cumberland Sound
Fram Strait
Sukhoy Nos
Svalbard
Chernaya Bay
Ob
Davis Strait
GREENLAND
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Barents Sea
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Murmansk
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Jan Mayen
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Kola NPP
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Arkhangelsk
White Sea
Jerisjärvi/
Akäsjärvi
Severnaya Dvina
East Greenland Current
Denmark Strait
Rovaniemi
Apukkajärvi
Kemijoki
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Radioactivity in the Arctic

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Contents

Preface

This assessment report details the results of the 2002 AMAP assessment of Radioactivity in the Arctic. It builds upon the previous AMAP radioactivity assessment that was presented in 'AMAP Assessment Report: Arctic Pollution Issues' that was published in 1998.*

The Arctic Monitoring and Assessment Programme (AMAP) is a group working under the Arctic Council.

The Arctic Council Ministers have requested AMAP to:

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

*This report is one of five detailed assessment reports that provide the accessible scientific basis and validation for the statements and recommendations made in the second AMAP State of the Arctic Environment report, 'Arctic Pollution 2002'** that was delivered to Arctic Council Ministers at their meeting in Inari, Finland in October 2002. It includes extensive background data and references to the scientific literature, and details the sources for figures reproduced in the 'Arctic Pollution 2002' report. Whereas the 'Arctic Pollution 2002' 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 2002 Ministerial Report', which also covers other priority issues (Persistent Organic Pollutants, Heavy Metals, Human Health, and Climate Change Effects on Contaminant Pathways), is reproduced in this report on pages vii to xi.

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 the AMAP Radioactivity Assessment under AMAP phase II 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 other bilateral and multilateral projects that have delivered data for use in this assessment, including the European Union and the Joint Norwegian–Russian Group on Environmental Cooperation.

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

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Lars-Otto Reiersen
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Oslo, June 2004

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

** AMAP, 2002. Arctic Pollution 2002: Persistent Organic Pollutants, Heavy Metals, Radioactivity, Human Health, Changing Pathways. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway. xii+112 pp.

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Executive Summary to the *Arctic Pollution 2002* Ministerial Report

The Arctic Monitoring and Assessment Programme (AMAP) was established in 1991 to monitor identified pollution risks and their impacts on Arctic ecosystems. In 1997 the first AMAP report, *Arctic Pollution Issues: A State of the Arctic Environment Report** was published.

The assessment showed that the Arctic is closely connected to the rest of the world, receiving contaminants from sources far outside the Arctic region. The report was welcomed by the Arctic Council Ministers, who 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 reported by AMAP.

The AMAP information greatly assisted the negotation of the protocols on persistent organic pollutants (POPs) and heavy metals to the United Nations Economic Commission for Europe's Convention on Long-range Transboundary Air Pollution (LRTAP Convention). They also played an important role in establishing the need for a global agreement on POPs, which was concluded in 2001 as the Stockholm Convention. Persistence, long-range transport, and bioaccumulation are screening criteria under both the POPs protocol and the Stockholm Convention, to be applied to proposals to add substances to the agreements. Information from AMAP will be useful in this context in showing whether persistent substances are accumulating in the Arctic and are therefore candidates for control, and also in assessing the effectiveness of the agreements.

The Arctic Council also decided to take cooperative actions to reduce pollution of the Arctic. As a direct follow up of the AMAP reports, the Arctic Council Action Plan to Eliminate Pollution of the Arctic (ACAP) was created to address sources identified through AMAP. ACAP was approved in 2000 and several projects have begun. The AMAP information was also used in establishing priorities for the Arctic Regional Programme of Action to Prevent Pollution from Landbased Sources (RPA), developed by the working group on Protection of the Arctic Marine Environment (PAME), and adopted by the Arctic Council in 1998.

After the first assessment, AMAP was asked to continue its activities and provide an updated assessment on persistent organic pollutants (POPs), heavy metals, radioactivity, human health, and pathways in 2002. Five scientific reports and a plain-language report have been prepared. This Executive Summary provides the main conclusions and recommendations of the 2002 AMAP assessments.

International Agreements and Actions

As described above, the LRTAP Convention protocols and the Stockholm Convention are essential instruments for reducing contamination in the Arctic. However, they cannot have any effect until they are ratified and implemented.

It is therefore recommended that:

- **The UN ECE LRTAP Protocols on Heavy Metals and POPs be ratified and implemented.**
- **The Stockholm Convention on POPs be ratified and implemented.**

Specific recommendations for monitoring activities in support of these agreements are included in subsequent sections.

Persistent Organic Pollutants

The POPs assessment addresses several chemicals of concern, including both substances that have been studied for some time and chemicals that have only recently been found in the environment.

The 1997 AMAP assessment concluded that levels of POPs in the Arctic environment are generally lower than in more temperate regions. However, several biological and physical processes concentrate POPs in some species and at some locations, producing some high levels in the Arctic.

The present AMAP assessment has found that the conclusions and recommendations of the first assessment remain valid. In addition:

It has clearly been established that:

Certain Arctic species, particularly those at the upper end of the marine food chain as well as birds of prey, carry high levels of POPs. Marine mammals, such as polar bear, Arctic fox, long-finned pilot whale, killer whale, harbor porpoise, minke whale, narwhal, beluga, harp seal and northern fur seal, some marine birds including great skua, great black-backed gull and glaucous gull, and birds of prey such as peregrine falcon, tend to carry the highest body burdens.

Most of the total quantity of POPs found in the Arctic environment is derived from distant sources. The POPs are transported to the Arctic by regional and global physical processes, and are then subjected to biological mechanisms that lead to the high levels found in certain species. Several potential source regions have now been identified within and outside of the Arctic. A better understanding of local re-distribution mechanisms has also emphasized the important potential role of local processes and sources in determining observed geographical variability.

There is evidence that:

Adverse effects have been observed in some of the most highly exposed or sensitive species in some areas of the Arctic. Several studies have now been completed on a number of Arctic species, reporting the types of effects that have been associated in non-Arctic species with chronic exposure to POPs, of which there are several examples. Reduced immunological response in polar bears and northern fur seals has led to increased susceptibility

* AMAP, 1997. Arctic Pollution Issues: A State of the Arctic Environment Report. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway, xii+188 pp. *and*
AMAP, 1998. AMAP Assessment Report: Arctic Pollution Issues. Arctic Monitoring and Assessment Programme (AMAP), Oslo, Norway, xii+859 pp.

to infection. Immunological, behavioral, and reproductive effects as well as reduced adult survival has been found in glaucous gulls. Peregrine falcons have suffered from eggshell thinning and reproductive effects. Reproductive effects in dogwhelks are associated with exposure to tributyltin.

It is therefore recommended that:

- **AMAP be asked to further enhance studies aimed at detecting effects in Arctic species relating to exposure to high levels of POPs and to integrate this information with an understanding of general population effects and health.** Without this understanding, it will not be possible to assess whether proposed and existing controls can be expected to afford the necessary protection (e.g., under the LRTAP and Stockholm agreements).

There is evidence that:

The levels of some POPs are decreasing in most species and media in the Arctic, but the rates vary in extent, location and media or species being studied. The decreases can be related to reduced release to the environment. For example, declines in alpha-HCH in air closely follow decreases in global usage, but declines in marine biota are much slower due to a huge reservoir of the substance in the global oceans.

For other POPs, declines are minimal and some levels are actually increasing, despite low current emissions. This illustrates the long period that may pass between the introduction of controls and the resulting decrease in levels in biota, as has been observed for PCBs, toxaphene, and beta-HCH.

It is therefore recommended that:

- **AMAP be asked to continue trend monitoring of POPs in key indicator media and biota.** This will enable assessment of whether the measures taken in the LRTAP Protocol and the Stockholm Convention are being effective in driving down POPs levels in the Arctic.

There is evidence that:

POPs substances other than those included in the LRTAP Protocol and Stockholm Convention may be at or approaching levels in the Arctic that could justify regional and global action. For example, levels of the brominated flame retardants such as polybrominated diphenyl ethers (PBDEs), polychlorinated naphthalenes (PCNs), and some current-use pesticides such as endosulfan have been monitored in Arctic air and biota. PBDEs are increasing in the Canadian Arctic.

It is therefore recommended that:

- **AMAP be asked to maintain a capacity to detect current-use POPs in the Arctic.** This will help ensure that Arctic States have an early opportunity to respond to a trend indicating Arctic accumulation, thus allowing a proactive approach to minimize the contamination rather than having to respond to a more serious situation later.

Heavy Metals

The heavy metals assessment focuses on mercury, lead, and cadmium.

It has clearly been established that:

In the Arctic, mercury is removed from the atmosphere and deposits on snow in a form that can become bioavailable. Enhanced deposition occurs in the Arctic. This recently discovered process is linked to polar sunrise, and is unique to high latitude areas. The resulting enhanced deposition may mean that the Arctic plays a previously unrecognized role as an important sink in the global mercury cycle.

There is evidence that:

Some of the deposited mercury is released to the environment at snowmelt, becoming bioavailable at the onset of animal and plant reproduction and rapid growth. Although poorly understood, this process may be the chief mechanism for transferring atmospheric mercury to Arctic food webs.

It is therefore recommended that:

- **The Arctic Council encourage expanded and accelerated research on critical aspects of the mercury cycle and budget in the Arctic.** Such research should include long-range transport, mercury deposition mechanisms, processes leading to biological exposure and effects, and the influence of climate variability and change on these processes.

There is evidence that:

Despite substantial mercury emission reductions in North America and Western Europe during the 1980s, global mercury emissions may, in fact, be increasing. Mercury emissions from waste incineration are likely underestimated. The burning of coal in small-scale power plants and residential heaters, principally in Asia, are major potential sources of current mercury emissions. These emissions are likely to increase significantly due to economic and population growth in this region.

It is therefore recommended that:

- **The Arctic Council promote efforts at global, regional, and national levels to quantify all sources of mercury and report results in a consistent and regular manner to improve emission inventories.** Particular efforts should focus on measuring contributions made by the burning of coal for residential heating and small-scale power plants as well as by waste incineration.

There is strong evidence that:

There is a trend of increasing mercury levels in marine birds and mammals in the Canadian Arctic, and some indications of increases in West Greenland. The effects of these levels are not well understood. However, there are also examples of stable or decreasing levels in other regions, perhaps indicating the importance of local or regional processes.

It is therefore recommended that:

- **AMAP be asked to continue temporal trend monitoring and the assessment of effects of mercury in key indicator media and biota.** This will enable assessment of whether the measures taken in the LRTAP Protocol are being effective in driving down mercury levels in the Arctic.

There is evidence that:
Current mercury exposures pose a health risk to some people and animals in the Arctic. These risks include subtle neurobehavioral effects.

It is therefore recommended that:
- **In view of the fact that reducing exposure to mercury can only be addressed by regional and global action to reduce worldwide emissions, and acknowledging the assessment for global action undertaken by UNEP and its resulting proposals, the Arctic Council take appropriate steps to ensure that Arctic concerns are adequately addressed and to promote the development of regional and global actions.**

It has clearly been established that:
Dramatic reduction in the deposition of atmospheric lead has occurred in Arctic regions where the use of leaded gasoline is banned. Arctic-wide elimination of leaded gasoline use will reduce lead exposure in other regions of the Arctic. Although levels in wildlife and fish have not measurably declined, likely reflecting continued uptake from the large reservoir of lead deposited in soils and sediments, lead levels in the environment are expected to diminish over time if current trends continue.

It is therefore recommended that:
- **The Arctic Council support continued efforts to eliminate the use of leaded gasoline in all Arctic regions.**

It has clearly been established that:
Certain regions of the Arctic contain elevated lead levels in the environment because of past or current use of lead shot by hunters. Even though lead shot is banned in Alaska, for example, lead blood levels in endangered US populations of Steller's eiders are above known avian toxicity thresholds for lead poisoning, which may be responsible for observed reduced breeding success. In Greenland, lead shot appears to be a significant source of human dietary exposure to lead.

It is therefore recommended that:
- **The Arctic Council encourage a complete ban on the use of lead shot in the Arctic, and that enforcement be improved.**

There is evidence that:
Cadmium levels in some seabirds is high enough to cause kidney damage. Monitoring data on cadmium in the abiotic and biotic environment to date provide no conclusive evidence of trends or effects. However, cadmium accumulates in birds and mammals and not enough is known about possible effects.

It is therefore recommended that:
- **The monitoring of cadmium in the Arctic be continued to support human exposure estimates.**

There is evidence that:
Levels of platinum, palladium, and rhodium have increased rapidly in Greenland snow and ice since the 1970s. These elements are used in automobile catalytic converters to reduce hydrocarbon pollution. The toxicity and bioaccumulation potential of these elements are largely unknown, which prevents assessment of their potential impact in the Arctic.

It is therefore recommended that:
- **AMAP be asked to consider the need to monitor trends of platinum, palladium, and rhodium in the Arctic.**

Radioactivity

The radioactivity assessment addresses man-made radionuclides and radiation exposures deriving from human activities.

It has clearly been established that:
In general, levels of anthropogenic radionuclides in the Arctic environment are declining. Most of the radioactive contamination in the Arctic land environment is from the fallout from nuclear weapons testing during the period 1945 to 1980. In some areas, the fallout from the Chernobyl accident in 1986 is a major source. For the Arctic marine environment, a major source of radionuclides is the releases from European reprocessing plants at Sellafield and Cap de la Hague.

However, releases from the reprocessing plants have resulted in increases in levels of some radionuclides in the European Arctic seas during recent years, in particular technetium-99 and iodine-129. The present doses to the population are low but the present levels of technetium in some marine foodstuffs marketed in Europe are above the EU intervention levels for food to infants and are close to the intervention level for adults.

The technetium information adds further weight to the recommendation made by AMAP to the Arctic Council in Barrow in 2000 that:
- **'The Arctic Council encourage the United Kingdom to reduce the releases from Sellafield to the marine environment of technetium, by implementing available technology.'**

There is evidence that:
Radionuclides in sediments are now a source of plutonium and cesium-137 to the Arctic. Earlier releases such as those from Sellafield that have deposited in sediments in the Irish Sea, especially cesium-137 and plutonium, have been observed to remobilize so that these deposits are now acting as sources to the Arctic. Thus, even if operational releases of these radionuclides from reprocessing plants are reduced, releases from environmental sources such as contaminated sediment in the Irish Sea and the Baltic Sea will be observed in the Arctic.

It is therefore recommended that:
- **The Arctic Council support a more detailed study on the remobilization of radionuclides from sediment and its potential effect on the Arctic.**

It is apparent that:
There is continuing uncertainty about the amount of radionuclides present at a number of sources and potential sources in the Arctic. Access to information about civilian and military sources continues to be a problem.

It is therefore recommended that:
- **The Arctic Council promote more openness of restricted information from any sources.**

It has clearly been established that:
Compared with other areas of the world, the Arctic contains large areas of high vulnerability to radionuclides. This is due to the characteristics of vegetation, animals, human diets, and land- and resource-use practices. On land in the AMAP area, there is considerable variation in vulnerability due to differences in these characteristics. In contrast, vulnerability associated with releases of radionuclides to the marine environment is relatively uniform and similar to that for other areas of the world. Maps of vulnerable areas, when combined with deposition maps, can be useful in an accident situation. The information on vulnerability is of importance for emergency planning.

It is therefore recommended that:
- **AMAP be asked to clarify the vulnerability and impact of radioactivity on the Arctic environment and its consequences for emergency preparedness planning.**

It is apparent that:
When performing risk reducing actions, close links to assessment programs are important and interventions should be prioritized in relation to the extent and magnitude of threats posed by nuclear activities, especially in respect to accidents. Interventions themselves can also have negative effects for humans and the environment, and careful judgments have to be made together with environmental impact assessments prior to carrying out a project. It is the view of AMAP that this has not always been done in interventions adopted to date.

It is therefore recommended that:
- **Risk and impact assessment programmes be performed prior to implementation of action to reduce risk.**
- **Risk and impact assessments, including accident scenarios, be performed with regard to the transport of nuclear waste and fuel within the Arctic and nearby areas and with regard to planned storage and reprocessing within the Arctic and nearby areas.**

It is apparent that:
The protection of the environment from the effects of radiation deserves specific attention. The current system of radiological protection is entirely based on the protection of human health. This approach can fail to address environmental damage in areas such as the Arctic that have low human population densities. Recently, an international consensus has emerged that the rapid development of a system and a framework for the protection of the environment needs further effort. The International Union of Radioecology (IUR), with support from AMAP, was one of the first international organizations to promote and present such a system and framework.

It is therefore recommended that:
- **AMAP be asked to take an active part in the continued efforts to address environmental protection, with special responsibility for the Arctic.** This should include the task of adding the need for protection of the environment into monitoring strategies and assessment tools.

It is noted that:
Since the previous AMAP assessment, nuclear safety programmes have been implemented in Russia at some nuclear power plants and other nuclear installations relevant to the Arctic.

It is therefore recommended that:
- **The Arctic Council continue its cooperation with Russia to improve the safety and safeguarding of nuclear installations and waste sites.**

Human Health

The human health assessment considered health risks associated with exposure to contaminants in relation to other lifestyle factors determining health. This assessment has extended geographical coverage and confirmed the conclusions and recommendations from the first assessment.

It has clearly been established that:
The highest Arctic exposures to several POPs and mercury are faced by Inuit populations in Greenland and Canada. These exposures are linked mainly to consumption of marine species as part of traditional diets. Temporal trends of human exposures to POPs have so far not been observed. Exposure to mercury has increased in many Arctic regions while exposure to lead has declined.

It is therefore recommended that:
- **The monitoring of human exposure to mercury, relevant POPs, including dioxins and dioxin-like compounds and other chemicals of concern, be continued in order to help estimate risk, further elaborate geographical trends, and begin to establish time trends of exposure.**

There is evidence that:
Subtle health effects are occurring in certain areas of the Arctic due to exposure to contaminants in traditional food, particularly for mercury and PCBs. The evidence suggests that the greatest concern is for fetal and neonatal development. In the Arctic, human intake of substances with dioxin-like effects is a matter of concern, confirmed by recent results from Greenland. Increasing human exposure to current-use chemicals has been documented, for example for brominated flame retardants. Others such as polychlorinated naphthalenes (PCN) are expected to be found in human tissues. Some of these compounds are expected to add to the total dioxin activity in humans. The AMAP human health monitoring program includes a number of measures of effects, ranging from biomarkers of effects at the molecular level to epidemiological outcomes.

It is therefore recommended that:
- **The human health effects program developed by AMAP be more extensively applied in order to provide a better base for human risk assessment especially concerning pre- and neonatal exposures.**

It has clearly been established that:
In the Arctic, diet is the main source of exposure to most contaminants. Dietary intake of mercury and PCBs ex-

ceeds established national guidelines in a number of communities in some areas of the Arctic, and there is evidence of neurobehavioral effects in children in some areas. In addition, life-style factors have been found to influence the body burden of some contaminants, for example cadmium exposure from smoking. In the Arctic region, a local public health intervention has successfully achieved a reduction of exposure to mercury by providing advice on the mercury content of available traditional foods. The physiological and nutritional benefits of traditional food support the need to base dietary recommendations on risk-benefit analyses. The health benefits of breast-feeding emphasize the importance of local programs that inform mothers how adjustments within their traditional diet can reduce contaminant levels in their milk without compromising the nutritional value of their diet.

It is therefore recommended that:

- **In locations where exposures are high, carefully considered and balanced dietary advice that takes risk and benefits into account be developed for children and men and women of reproductive age.** This advice should be developed by national and regional public health authorities in close consultation with affected communities.
- **Studies of the nutrient and contaminant content of traditional food items be promoted in order to assess their benefits and to estimate exposures as a basis for public health interventions.**
- **Breast-feeding continue to be recognized as a practice that benefits both mother and child.** Nonetheless, if contaminant levels increase or more information indicates increased risk, the potential need for restrictions should continue to be evaluated.

It is noted that:

From the Arctic human health perspective, it is of utmost importance that considerations for global actions against POPs and mercury take into account the concerns for Arctic human health. The Stockholm Convention and the LRTAP protocols should be properly monitored in the Arctic to determine whether their implementation is effective in protecting human health.

It is therefore recommended that:

- **AMAP participate in the global monitoring of human exposure to be established under the Stockholm Convention on POPs.**
- **The Arctic Council monitor proposals for global action on mercury being undertaken by UNEP, and contribute as necessary to ensure that Arctic concerns related to human health are adequately addressed.**

Changing pathways

The assessment of changing pathways provides an introduction to the types of changes on contaminants pathways to, within, and from the Arctic that might be expected as a result of global climate change and variability.

There is evidence that:

The routes and mechanisms by which POPs, heavy metals, and radionuclides are delivered to the Arctic are strongly influenced by climate variability and global climate change. These pathways are complex, interactive systems involving a number of factors, such as temperature, precipitation, winds, ocean currents, and snow and ice cover. Pathways within food webs and the effects on biota may also be modified by changes to climate. Studies using global change scenarios have indicated the potential for substantial changes in atmospheric and oceanographic pathways that carry contaminants to, within, and from the Arctic. These effects mean that climate-related variability in recent decades may be responsible at least in part for some of the trends observed in contaminant levels.

It is therefore recommended that:

- **AMAP be asked to further investigate how climate change and variability may influence the ways in which POPs, heavy metals, and radionuclides move with respect to the Arctic environment and accumulate in and affect biota.** This will enable Arctic States to better undertake strategic planning when considering the potential effectiveness of present and possible future national, regional, and global actions concerning contaminants.

Chapter 1

Introduction

This report concerns issues relating to the presence and effects of radioactive contaminants in the Arctic. It follows on from Chapter 8 of the first AMAP assessment report (AMAP, 1997, 1998; available at *http://www.amap.no*). The purpose of this second assessment is to provide an update in cases where new information has become available that either warrants revised assessment or relates to operations and sources that were not previously considered. Such information has been provided by Arctic countries as a result of further research or as a consequence of AMAP monitoring activities.

The first AMAP assessment was based on information available up to the beginning of 1997. It contained an introduction to the topics of radioactivity and radiological protection and explanations of transport processes and exposure pathways for radionuclides. The assessment then dealt sequentially with: past and present radioactive contamination of the Arctic; individual doses to man estimated from environmental measurements; source-related assessments of past and present releases; source-related assessments of potential releases; and spatial analysis of the vulnerability of Arctic ecosystems. The main part of the assessment concerned: the presence and distribution of radionuclides in the Arctic and their sources; doses to humans resulting from the presence of these radionuclides and their sources at both individual and collective (population) levels; the radiological threats posed by known sources of radionuclides in the Arctic, especially in relation to potential accidents in the civilian and military sectors; and the vulnerability of the Arctic in the sense of the degrees of human exposure consequent to accidental releases in the Arctic compared to other regions of the world. The overall conclusions of the previous AMAP assessment of radioactivity in the Arctic, supported by more detailed explanations were that:

'... the greatest threats [from radioactivity] to human health and the environment posed by human and industrial activities in the Arctic are associated with the potential for accidents in the civilian and military nuclear sectors. Of most concern are the consequences of potential accidents in nuclear power plant reactors, during the handling and storage of nuclear weapons, in the decommissioning of nuclear submarines and in the disposal of spent nuclear fuel from vessels. In the Arctic, terrestrial pathways of human exposure to radioactive contamination are far more important than marine pathways. The vulnerability of Arctic populations, especially indigenous peoples, to radiocaesium deposition is much greater than for temperate populations due to the importance of terrestrial, semi-natural exposure pathways.'

The current system of radiological protection is based solely on the protection of human health. The first AMAP assessment concluded that the feasibility of assessing radiation effects on flora and fauna was limited. During AMAP Phase II, however, the development of a framework which includes the protection of flora and fauna in the basis for radiological protection has been given increased priority. This is warranted owing to the low human population density of the Arctic and the need to ensure that flora and fauna are adequately protected from the effects of radionuclides in the environment.

The general recommendations of the first AMAP assessment advocated:

- rigorous adherence to international guidance on radiological protection;
- more authoritative and comprehensive evaluations of the risk posed by accidents in the nuclear power industry;
- increased attention to nuclear safety in nuclear fleet operations; and
- improved estimates of habits and diets of Arctic residents and radionuclide transfer rates in the environment leading to human exposure thereby improving the basis for assessing radiation exposure and risk and deciding on the need for intervention.

These general recommendations were augmented by more specific recommendations concerning the storage of spent nuclear fuel and radioactive waste, monitoring, and limitations in the availability of information and scientific understanding.

The topics covered in this assessment are basically similar to those of the previous report but with emphasis given to new information and the results of investigations instigated as a direct result of the first AMAP assessment. Particular attention is given to progress in the development of a radiological protection system that includes provisions to guard against adverse effects of radionuclides on biota. Additional topics new to this assessment are the loss of the nuclear-powered submarine *Kursk* off Murmansk in August 2000, and the potential for increased transport of spent nuclear fuel and mixed oxide fuel. These topics are augmented by re-assessments of specific activities or sources based on information that has become available since the beginning of 1997. Only those issues that pertain directly to conditions in the Arctic or that pose threats to the Arctic environment are addressed in this assessment.

Chapter 2

Sources of Radionuclides

2.1. Introduction

The previous AMAP assessment (AMAP, 1998) described the actual and potential anthropogenic sources of radionuclides and associated radiological effects relevant to the Arctic environment. Actual sources are those from which there are continuing releases of radionuclides to the open environment (i.e., to areas outside normal regulatory control). These include emissions from nuclear fuel reprocessing plants in Western Europe, routine emissions from nuclear power plants in the Arctic (both civilian and military), and fallout from the atmosphere as a legacy of atmospheric nuclear weapons tests. Potential sources are contained sources of radionuclides that are managed in a manner that has the goal of preventing radiologically significant releases to the environment. These include civilian and military nuclear reactors, nuclear waste storage facilities, and authorized accumulations of radionuclides in controlled areas such as the Mayak storage ponds. Such containment can fail, leading to additional releases of radionuclides to the open environment and possible increased exposure of humans and other organisms.

This chapter includes new information on sources of radioactivity covered in the previous AMAP assessment as well as on some additional sources not addressed in the previous assessment. An overview of the actual, potential, and removed sources in the Arctic discussed in this chapter is given in Figure 2·1; Table 2·1 shows the changes of name of the Russian reprocessing plants.

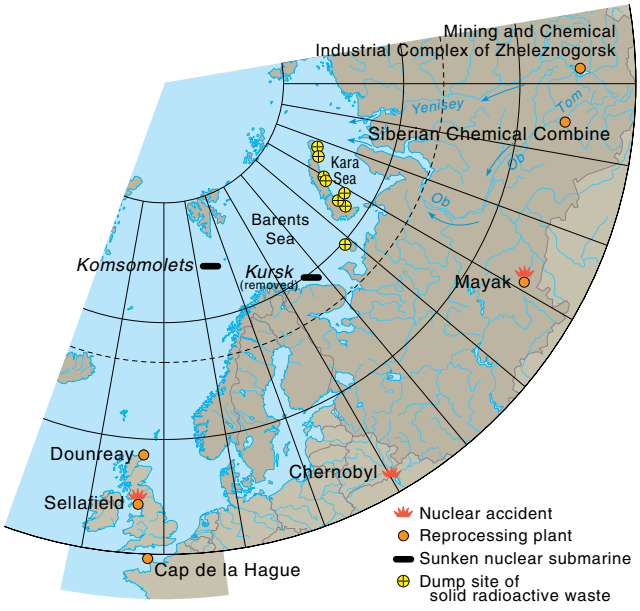

Figure 2·1. Sites discussed in this chapter.

Table 2·1. Changes of name of Russian reprocessing plants.

Present name	Name used in the former Soviet Union
Mayak	Chelyabinsk-65
Siberian Chemical Combine (SCC)	Tomsk-7
The Mining and Chemical Industrial Complex of Zheleznogorsk (KMCIC)	Krasnoyarsk-26

2.2. Major reprocessing sources

Nuclear fuel reprocessing is conducted to recover uranium (U) and plutonium (Pu) from spent nuclear fuel for re-use. Most spent nuclear fuel from reactors is retained on-site in interim storage pending decisions on ultimate disposal or retrievable storage. Worldwide, only about 5 to 10% of spent nuclear fuel is reprocessed. During reprocessing, radionuclides are freed from their contained state as the fuel is brought into solution. In this liquid state, the potential for release in waste discharges is greater than at other stages of the fuel cycle. Routine releases have mostly been in liquid effluents discharged to the sea and emissions to the atmosphere. The main commercial reprocessing plants are in France, Japan, and the U.K. (UNSCEAR, 2000).

The locations of the western European reprocessing plants, at Sellafield (U.K.), Cap de la Hague (France), and Dounreay (U.K.), are shown in Figure 2·1. Routine liquid effluent releases from the Dounreay reprocessing plant have now ceased. Liquid radioactive wastes from operations at Sellafield and La Hague are discharged via pipelines directly into the Irish Sea and the English Channel, respectively. Radionuclides, especially those that are soluble or conservative (i.e., those that are less particle-reactive than others), from these sources are transported northward by regional ocean currents. Sellafield (formerly Windscale) has been the main contributor to these activity releases. Maximum discharges of ^{137}Cs and the actinides ^{239,240}Pu and ^{241}Am from Sellafield occurred during the mid- to late-1970s (Gray *et al.*, 1995). The introduction of the Site Ion-Exchange Effluent Plant (SIXEP) in 1985 was followed by a dramatic reduction in discharges of ^{90}Sr, ^{134}Cs, and ^{137}Cs (Gray *et al.*, 1995). Increased throughput and processing of residues led to increased discharges of Pu and americium (Am) during the early- to mid-1970s. Discharges then decreased following the operation of a flocculation precipitation facility from the mid-1970s, the cessation of discharges of concentrates to the sea, and the commissioning of the salt evaporator in 1985 (Gray *et al.*, 1995).

The first AMAP assessment (AMAP, 1998) addressed the importance of western European nuclear fuel reprocessing plants as radionuclide sources, especially with respect to radiocesium. This assessment considers radionuclides in these discharges that were not discussed in detail in the earlier assessment, namely ^{99}Tc and ^{129}I, and the remobilization of Pu and cesium (Cs) from Irish Sea sediments that constitute a secondary source.

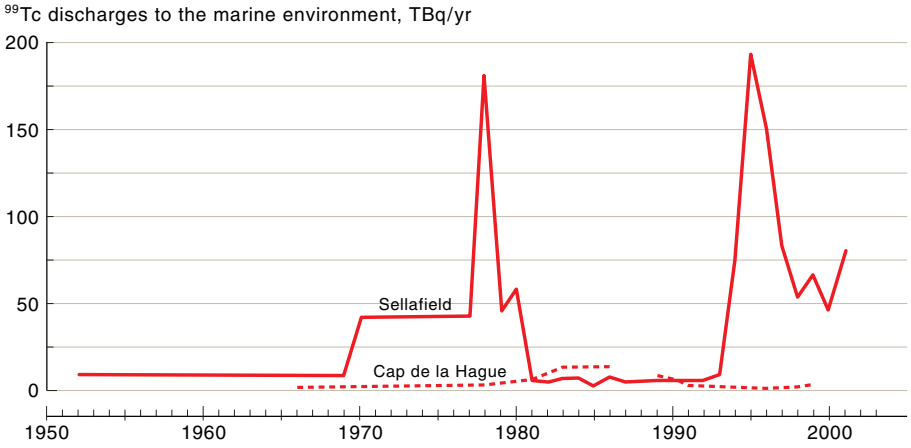

Figure 2·2. Discharges of ^{99}Tc from Sellafield and Cap de la Hague (compiled from CEC, 1990; Gray *et al.*, 1995; Mayall, 2002).

2.2.1. Technetium-99

Technetium-99 is a long-lived fission product. Liquid discharges from the Sellafield reprocessing plant on the northwest coast of England are the major source of ^{99}Tc to Arctic marine ecosystems. Discharges of ^{99}Tc were historically high but declined in the late-1970s. Throughout the 1980s and early 1990s (1981-1993) discharges were relatively low at 1.9 to 6.6 TBq/yr. During this period the ^{99}Tc, which is present in medium active concentrate streams, was held on-site. In 1994, the Enhanced Actinide Removal Plant (EARP) became operational and began to treat the backlog of stored wastes (Gray *et al.*, 1995). This resulted in a considerable increase in the discharge of ^{99}Tc, from about 5 TBq/yr in 1993 to 72-190 TBq/yr post-1994, as illustrated in Figure 2·2. The total quantity of ^{99}Tc discharged from Sellafield is greater than that released to the stratosphere between 1945 and 1980, approximately 140 TBq, and present in global fallout (Rioseco, 1987).

The increased discharges from 1994 onwards together with the long half-life and conservative properties of this nuclide have resulted in ^{99}Tc being detectable over long distances. Increased activity levels in biota and seawater were observed in Norwegian coastal environments by 1997 (Kolstad and Lind, 2002), (see Section 3.3.1).

2.2.2. Iodine-129

Iodine-129 is both a naturally occurring and man-made radioisotope of iodine. The pre-nuclear era ratio of ^{129}I : ^{127}I was estimated at ~10^{-12} for the ocean. Releases from nuclear weapons tests prior to the 1980s increased this to ~10^{-10} (Fehn *et al.*, 1986; Hou *et al.*, 2000a; Raisbeck and Yiou, 1999; Schink *et al.*, 1995). However, the largest releases of ^{129}I have been from the two main European reprocessing plants, especially in recent years (Figure 2·3). By 1999, 2113 kg and 865 kg of ^{129}I had been discharged to the marine environment from Cap de la Hague and Sellafield, respectively (BNFL, 2000; Gray *et al.*, 1995; GRNC, 1999; Hou *et al.*, 2000b; Raisbeck and Yiou, 1999). This is an order of magnitude greater than the total ^{129}I inventory from nuclear weapons test releases and three orders of magnitude greater than the release from the Chernobyl accident (Raisbeck and Yiou, 1999). Discharges of ^{129}I from European reprocessing plants continue.

The radionuclides discharged to the English Channel from Cap de la Hague and to the Irish Sea from Sellafield are transported to the North Sea by the Atlantic Current and then to the Norwegian coast and the Arctic by the Norwegian Coastal Current.

2.2.3. Plutonium and ^{137}Cs remobilization from sediments

The Sellafield reprocessing plant has discharged about 700 TBq of plutonium-alpha (i.e., α-particle emitting Pu isotopes) since it was commissioned in 1952. The major proportion has been deposited in sediments of the Irish Sea (Kershaw *et al.*, 1999) and adjacent sea areas. When the discharges were reduced during the 1980s, it became apparent that Pu was being remobilized from the sediments because concentrations of dissolved Pu species in the water column did not reflect changes in the discharges (Hunt and Kershaw, 1990). The amount of dissolved Pu leaving the Irish Sea via the North Channel has been estimated at 0.6 to 1.2 TBq/yr (Cook *et al.*,

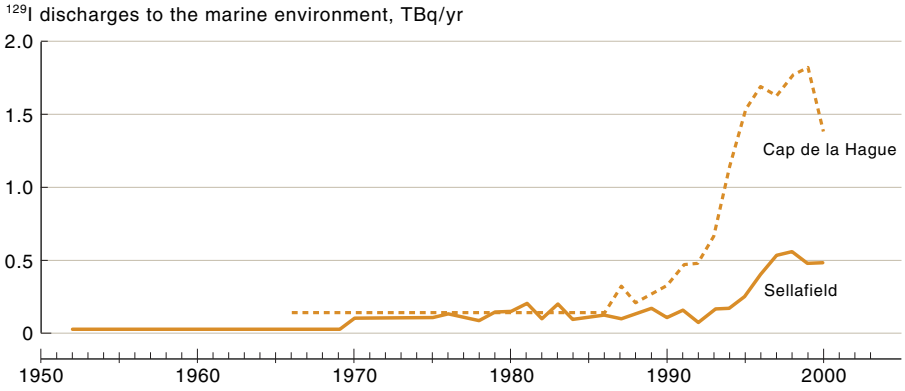

Figure 2·3. Marine discharge of ^{129}I from Sellafield and Cap de la Hague (compiled from BNFL, 2000; Gray *et al.*, 1995; GRNC, 1999; Hou *et al.*, 2000b; Raisbeck and Yiou, 1999).

1997; Leonard *et al.*, 1999). This pathway became a recognized source of Pu to Arctic waters in the mid-1990s (Grøttheim, 2000; Herrmann *et al.*, 1998; Kershaw *et al.*, 1995) (see section 3.3.4). Isotope ratios in the Sellafield discharge have varied throughout its period of operation, but values around 0.2 seem indicative of $^{238}Pu : ^{239,240}Pu$ activity ratios as well as of $^{240}Pu : ^{239}Pu$ atom ratios in the remobilized Sellafield-derived Pu (Kershaw *et al.*, 1995).

Most of the historic ^{137}Cs discharges from Sellafield (~40 PBq) remained in solution and were transported out of the Irish Sea, principally to Nordic waters (AMAP, 1998). Complexed cesium in seawater has a weak affinity for sediment particles and so only a very small proportion was deposited in the sediments of the Irish Sea and downstream sea areas such as the North Sea and Skagerrak. When the ^{137}Cs discharges were reduced in the mid-1980s, monitoring showed that concentrations in the water leaving the Irish Sea originated mainly from remobilized activity previously deposited in the sediments (Hunt and Kershaw, 1990). Cook *et al.* (1997) estimated an annual loss of 86 TBq of ^{137}Cs from Irish Sea sediments based on sampling in 1992. By comparing inventories of ^{137}Cs in Irish Sea sediments undertaken in 1988 and 1995, Poole *et al.* (1997) concluded that 350 to 573 TBq had been remobilized from the sediments during that period. The remobilization of ^{137}Cs from Irish Sea sediments has thus been a more significant source to Arctic waters than the 1990s discharges.

2.3. Dumping

In March 1993, a report was prepared by the governmental commission on radioactive waste disposal in seas adjacent to the territory of the Russian Federation (Yablokov, 1993). This is generally known as the 'White Book' or the 'Yablokov Report'. During its preparation, some Russian experts claimed that the upper estimated inventory for the reactors dumped in the Arctic seas were not justified. A major source of uncertainty at that time in the calculation of fission products, activation products, and transuranium elements (actinides), was the lack of information about the mass, enrichment, and burn-up of the spent nuclear fuel.

The first AMAP assessment outlined the results of the International Arctic Seas Assessment Project (IASAP). As part of IASAP, two Russian institutes (the Kurchatov Institute and the Institute of Physical and Power Engineering) undertook studies to improve estimates of the total inventory in the nuclear reactors (with and without spent nuclear fuel) dumped in the seas off the northwest regions of the Russian Federation (IAEA, 1997). Efforts to improve these estimates continued in Russia after IASAP was completed. In January 1998, a workshop associated with an International Science and Technology Center project (Lavkovsky, 1998) discussed the results of these studies and concluded that they did not significantly modify the outcome of IASAP (IAEA, 1998a). The results are summarized in the rest of this section (Sivintsev and Kiknadze, 1998).

The more recent calculations of radionuclide activities in the reactor compartments of the icebreaker *Lenin* (OK-150) and nuclear submarines dumped in the Kara Sea compared to the corresponding estimates in the 'White Book' are given in Table 2·2.

The comparison in Table 2·2 shows that the 'White Book' underestimated the inventory in the reactor compartment of the *Lenin* icebreaker. This is because earlier assessments included only the most long-lived radionuclides, namely ^{90}Sr, ^{137}Cs, ^{239}Pu, and ^{241}Am. The activity due to short-lived radionuclides, such as ^{144}Ce, ^{144}Pr, ^{147}Pr, ^{106}Ru, and ^{106}Rh, as well as relatively long-lived activation products from structural materials such as ^{14}C, ^{60}Co, ^{59}Ni, and ^{63}Ni, was not included. When these radionuclides were included, the estimated activity present increased by a factor of 3.5 (Table 2·2; Figure 2·4).

Radionuclide inventories for six submarine reactors dumped in coastal bays and near to the east coast of Novaya Zemlya between 1965 and 1981 and containing spent nuclear fuel were re-estimated between 1993 and 1998 by Sivintsev and Kiknadze (1998). Information on the quantity, enrichment, and burn-up of nuclear fuel in the submarine reactors was collated and evaluated by

Table 2·2. Radionuclide activities (TBq) in the reactor compartments of the icebreaker *Lenin* and submarines dumped in the Kara Sea.

	Number of reactors	Estimated activities at time of dumping		Estimated activity at time of dumping decayed to 2000 (Sivintsev and Kiknadze, 1998)
		'White Book' (Yablokov, 1993)	Sivintsev and Kiknadze (1998)	
Reactor compartment and screening assembly from nuclear icebreaker *Lenin* OK-150	1	3700	19500	1840
Nuclear submarines with spent nuclear fuel				
Reactor from nuclear submarine No. 285, starboard*	1	29600	11600	561
Reactor compartment from nuclear submarine No. 901	2	14800	2950	625
Reactor from nuclear submarine No. 421	1	29600	1050	253
Nuclear submarine No. 601	2	7400	1340	567
Nuclear submarines without spent nuclear fuel				
Reactor compartment from nuclear submarine No. 254	2	ca. 1850	93	8
Reactor compartment from nuclear submarine No. 285, portside*	1	ca. 1850	48	5
Reactor compartment from nuclear submarine No. 260	2		44	4
Reactors from nuclear submarine No. 538	2		6	4
Total		88800	36600	3870

* the fuel had been removed from the portside reactor when the reactor compartment was dumped.

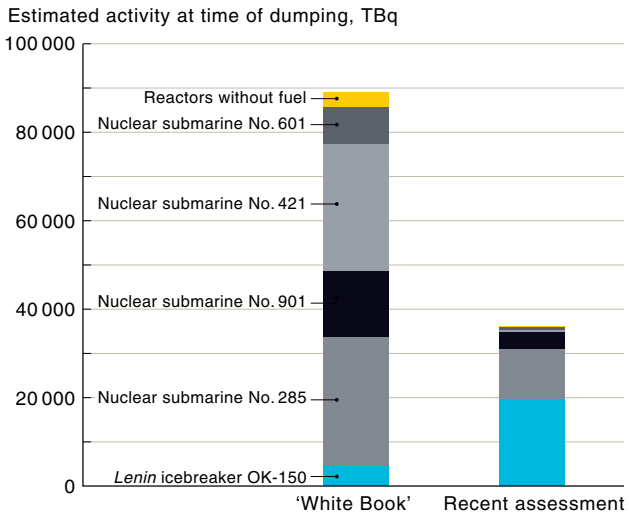

Figure 2·4. Estimates of the activity in dumped nuclear waste in the Kara Sea. The data are derived from the 'White Book' (Yablokov, 1993) and calculations by Sivintsev and Kiknadze (1998) and reflect activity at the time of dumping.

the Kurchatov Institute and the Institute of Physical and Power Engineering and has enabled improved estimates regarding the composition and activity of radionuclides in these potential sources of radioactive contamination. Calculations by Sivintsev and Kiknadze (1998) show that the total inventory in the submarine reactors with spent nuclear fuel at the moment of dumping was 4.8 times lower than that stated in the 'White Book' (Yablokov, 1993). These authors also estimated that the total inventory in all reactors dumped near Novaya Zemlya did not exceed 36 PBq (970 kCi). Thus, the value of 89 PBq (2.4 MCi) given for this activity in the 'White Book' was an overestimate by about a factor of 2.5 compared to the more recent estimate. The total inventory in the dumped material was estimated to have decayed to 3.9 PBq by 2000, of which 1.8 PBq was due to the dumped reactor from the *Lenin*.

2.4. Radioactive particles from the Thule nuclear weapons accident

On 21 January, 1968, a B-52 aircraft from the U.S. Strategic Air Command crashed onto the sea ice of Bylot Sound 11 km west of the Thule Air Base in Greenland. The aircraft disintegrated on impact and an explosion and fire ensued. The four nuclear weapons onboard were destroyed and fissionable material (Pu and U) was dispersed. The Pu was present in an insoluble oxide form and was mainly associated with particles of an average size of 2 µm (U.S. Air Force, 1970). During the months following the accident, a clean-up program was conducted during which most of the debris and contaminated ice were removed from the area. However, the impact created a hole in the ice and it is likely that some of the Pu initially fell through the ice. Furthermore, contamination could not be removed from the vicinity of the hole because the fractured sea ice quickly re-froze. Thus, relatively high contamination levels from the central impact area could have remained embedded in deep ice layers and therefore may not have been recovered in the clean-up process. The following summer, during ice melt, the ice sheet drifted in a northerly direction in Bylot Sound, probably causing the observed contamination of sediments north of the impact point. The total amount of Pu dispersed in the accident has been estimated at 6 kg, of which 3.5 ±0.7 kg were found on and in the sea ice, recovered and shipped to the United States (AMAP, 1998; U.S. Air Force, 1970). An estimated residue of ~1 TBq (~0.4 kg) remained in the ice after clean-up. Thus, up to about 3 kg of Pu may have entered the environment. The actual input may have been lower than this however, owing to an unknown amount of Pu associated with the aircraft debris removed from the site (AMAP, 1998). After the accident, the amount of Pu in the marine sediments of Bylot Sound was estimated based on samples taken in 1970, 1974, 1979, 1984, 1991, and 1997 (Aarkrog, 1971, 1977; Aarkrog *et al.*, 1984, 1987, 1994; Dahlgaard *et al.*, 2001; Eriksson *et al.*, 1999). These estimates centered on a range of 1 to 1.6 TBq or approximately 0.5 kg.

The nonhomogeneous nature of the Pu contamination in marine sediments (Figure 2·5) has been noted for many years, but was previously assumed not to significantly influence the inventory estimates. The work upon which this assumption was made was based solely on radiochemistry and alpha-spectrometry, and the presence of Pu associated with 'hot particles' may have been underestimated partly owing to the incomplete dissolution of these particles during sample analysis.

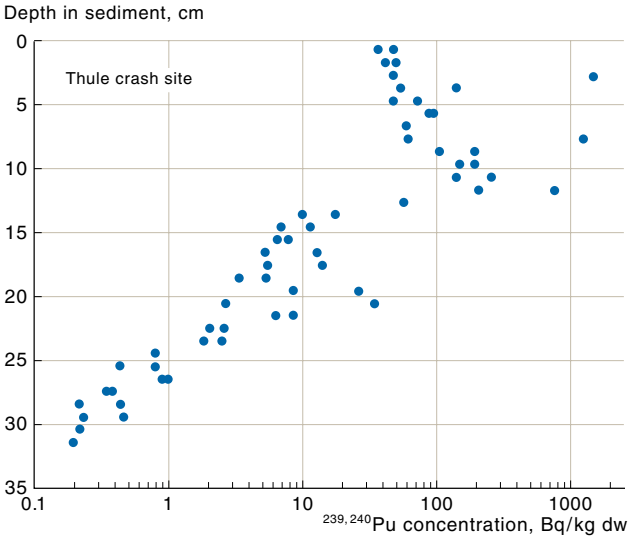

Figure 2·5. Concentrations of ^{239,240}Pu measured by radiochemistry and alpha-spectrometry in 1 g aliquots from a sediment core sampled in 1997 at location 25 (76°30.42'N, 69°13.88'W; water depth 237 m), 2.3 km southeast of the point of aircraft impact (Eriksson, 2002). Note the non-homogeneous results from repeated analysis indicating the possible presence of 'hot particles'.

An improved method to determine the total inventory of the heterogeneously distributed contamination of marine sediments was developed by Eriksson (2002). This is based on a gamma spectrometric screening of the ^{241}Am concentration in 450 one-gram aliquots from six sediment cores. Based on radiochemical determination of the Pu concentration in 20 of these subsamples, the ^{241}Am values were recalculated to provide estimates of ^{239,240}Pu concentrations. A Monte Carlo simulation was then used to generate the probable distribution of the activity and, based on that, a total inventory was estimated by integrating a double expo-

nential function. The resulting estimate of Pu in sediments at Bylot Sound, based on a limited number of sediment cores, was 10 TBq ± 50% (3.5 kg Pu ± 50%). This is consistent with the estimate of lost Pu, ~3 kg, provided by the U.S. Air Force (1970) but is subject to a large uncertainty (± 50%).

2.5. Nuclear powered vessels

2.5.1. Russian Northern fleet

Several nuclear-powered vessel accidents have occurred worldwide, including the loss of entire nuclear-powered submarines at sea, with and without nuclear weapons onboard. Few of these accidents are relevant to the Arctic. The first AMAP assessment addressed the issue of accidents involving nuclear-powered vessels as sources of radioactivity in the Arctic. That assessment also considered the status of submarine decommissioning from the Russian Northern Fleet and associated radioactive waste management issues. The present assessment gives specific attention to the loss of the Russian fleet submarine *Kursk* in the Barents Sea in 2000.

The decommissioning of nuclear-powered submarines of the Russian Northern Fleet is continuing. At the beginning of 2002, 94 decommissioned nuclear-powered submarines (52 in northwest Russia and 42 in the Russian Far East) were being stored afloat and contained spent fuel in their nuclear reactors. Approximately 110 submarines have been taken out of operation in the Northern Fleet. It is anticipated that 18 to 20 submarines can be dismantled per year (CEG, 2003).

The primary location for the storage of spent nuclear fuel (about 80% of the total inventory) from the Russian Northern Fleet is at Andreyeva Bay on the Kola Peninsula. There has been some leakage of radionuclides from this storage site and soil levels of the order of 10 MBq/kg of ^{137}Cs and 1 MBq/kg of ^{90}Sr have been observed in the area (Akhunov *et al.*, 2001). Further work is warranted to assess the total radionuclide inventory in the area. The handling and storage of spent nuclear fuel is of major importance in terms of threats to the Arctic environment. Long-term disposal options are still being considered and there are several radioactive waste management problems yet to be resolved.

2.5.2. The loss of the *Kursk* in the Barents Sea

On August 12, 2000, a Russian submarine, the *Kursk*, sank in international waters east of Rybatschi Peninsula in the Barents Sea. The submarine, a Russian Oscar Class II attack submarine, sank to a depth of 116 m at 69°36.99'N, 37°34.50'E, about 190 km from Murmansk. The *Kursk* was 154 m long and weighed 14700 t. The submarine was commissioned in 1995 and was powered by two pressurized-water reactors. Each reactor had an output of 190 MW(th), or less than 10% of that of a typical nuclear power plant. The reactors were shut down during the accident and the submarine was not carrying nuclear weapons.

Two joint Russian–Norwegian expeditions to the *Kursk* were mounted in 2000; the first undertaken with the *Seaway Eagle* from 17 to 22 August 2000, and the second with the MSV *Regalia* from 20 October to 7 November 2000. During both expeditions, seawater and sediments in the close vicinity of the *Kursk* were sampled extensively and water was sampled from inside the *Kursk* to determine whether there had been any leakage from the reactors. During the *Seaway Eagle* expedition divers collected a sample of escaping air from the submarine as the rescue hatch was opened. An air sampling device was used on both expeditions to measure airborne radioactivity over the main deck of the ship. Dose rate measurements and sample analyses were performed on-site in mobile laboratories onboard the survey ships. Replicates of all samples were taken ashore for more accurate analyses.

There was no indication of radionuclide leakage from the submarine; activity concentrations in the vicinity were normal. The concentration of ^{137}Cs in sediments ranged from 0.7 to 1.2 Bq/kg, which is similar to concentrations in recent samples from other areas of the Barents Sea (AMAP, 1998; Grøttheim, 2000). Iodine-131, ^{134}Cs, and ^{60}Co were not detected in any of the samples. Activity concentrations of γ-emitting radionuclides in the seawater samples were not elevated. The results of all analyses were below detection limits of 0.5 Bq/L for ^{131}I, ^{137}Cs, ^{134}Cs, and ^{60}Co.

Radioactive contamination of the marine environment in the area where the *Kursk* sank was also assessed during several Russian cruises. A radioecological study in September 2000 undertaken from the research vessel *Dalnie Zelentsy* showed the activity concentration of ^{137}Cs in seawater in this region to be no higher than background at 2 to 3 Bq/m^3 (Amundsen *et al.*, 2002a; Izrael *et al.*, 2000).

In 2001, a company was contracted to raise, transport, and moor the *Kursk* in a floating dock. To support these operations, a special expedition was set up to undertake environmental monitoring in the vicinity of the submarine during all phases of the recovery operation. An environmental monitoring and assessment group was created and charged with three tasks: 1) to measure the current state of environmental contamination; 2) to predict possible changes in the radiation situation during the salvage operation; and 3) to provide relevant information promptly.

Joint Russian–Norwegian environmental monitoring in relation to the recovery of the *Kursk* was carried out from 25 September to 10 October 2001 onboard the research vessel *Semen Dezhnev* of the Russian Northern Fleet. This expedition involved four stages:

- a survey of background activities in the Barents Sea (remote from the submarine) prior to the recovery operation;
- a survey in the vicinity of the *Kursk* prior to recovery;
- a survey in the vicinity of the *Kursk* during recovery; and
- a survey at the site from which the *Kursk* was recovered, soon after its removal from the area.

Summaries of the onboard gamma spectrometric measurements are reported in Table 2·3. No artificial γ-emitters (^{134}Cs, ^{60}Co, and others) indicative of radionuclide leakage from the submarine were detected. After the *Kursk* was removed, surface and near-bottom seawater samples were analyzed in land-based laboratories for a wider range of radionuclides, including ^{137}Cs, ^{90}Sr, and ^{239,240}Pu.

Table 2·3. Summary of onboard measurements for ^{137}Cs collected during the joint Russian–Norwegian expedition to monitor radioactive contamination during the raising of the *Kursk* (Amundsen *et al.*, 2002b; Anon, 2001).

	Filtered near-bottom seawater, Bq/m^3	Surface layer bottom sediments, Bq/kg ww	Fish, Bq/kg ww
Background area of the Barents Sea, 25-27 Aug. 2001	2.5-2.7	1.7-3.1	<5
The *Kursk* accident area before lifting, 28 Sept. to 7 Oct. 2001	2-3	<3	<5
The *Kursk* accident area during lifting, 8 Oct. 2001	2.5-3	<2	–
The *Kursk* accident area after lifting, 9-10 Oct. 2001	2.5-3	<3	<3

The ^{137}Cs concentrations were about 3 Bq/m^3 in seawater and <2 to 4 Bq/kg ww in the surface layer of bottom sediments and in fish. Such values are typical of those found in the Barents Sea in recent years. Thus, no increase in ^{137}Cs was observed in the marine environment during the raising of the *Kursk*. Laboratory radiochemical analyses (Anon, 2001) indicate that ^{90}Sr concentrations in surface and near-bottom water samples obtained after the *Kursk* was raised were 5.1 ± 0.6 Bq/m^3 and 1.6 ± 0.2 Bq/m^3 respectively. For ^{239,240}Pu these concentrations were 13.4 ± 2.4 mBq/m^3 and 3.8 ± 1.1 mBq/m^3 respectively. The ^{90}Sr and ^{239,240}Pu seawater concentrations were also typical of recent values for the Barents Sea. Thus, no impact of either the *Kursk* accident or the recovery operation on the radioecological situation was detected (Amundsen *et al.*, 2002b; Grøttheim, 2000; Roshydromet, 2001).

2.6. Baltic outflow as a source of ^{137}Cs

Outflow from the Baltic Sea incorporated in the Norwegian Coastal Current acts as a transport pathway for radioactivity to the Arctic. The Chernobyl accident contaminated the Baltic with 4 to 5 PBq of ^{137}Cs in 1986. The inventory in the seawater was reduced by 50% within the first few years (see AMAP, 1998, Table 8·41) despite additional influxes through runoff. The amount of Chernobyl-related ^{137}Cs deposited on Finland entering the Baltic Sea through Finnish rivers amounted to only 65 TBq, leaving >97% on the land (Saxen and Ilus, 2001).

A time series of ^{137}Cs data from the Danish Straits revealed that contamination from the Chernobyl accident in 1986 is still the dominant source of ^{137}Cs in the Baltic Sea and Danish Straits. The Danish Straits are a transition zone between the brackish surface water of the Baltic Sea (salinity 7) and the high salinity water of the North Sea (salinity 33 to 34). Inflowing North Sea water is gradually entrained in the outflowing surface water from the Baltic. Therefore, surface water salinity gradually increases from the Baltic to the North Sea. Linear regressions of ^{137}Cs concentration versus salinity can be used to estimate the time trend of ^{137}Cs in the different water masses. A mean residence time for the Baltic Sea of 29 yr equates to a 'half-life' for the water volume of 20 yr (Dahlgaard, 2002). The net outflow from the Baltic was thus calculated at 39 TBq for 2000. This value showed an exponential decrease between 1991 and 2000 with an effective half-life of 13.4 yr. When the physical half-life of ^{137}Cs (30.17 yr) is subtracted, the decay-corrected cesium net outflow can be described by an exponential decrease with a half-life of 24 yr. This is equivalent to the estimated 20-yr half-life for Baltic Sea water mentioned above (Figures 2·6 and 2·7).

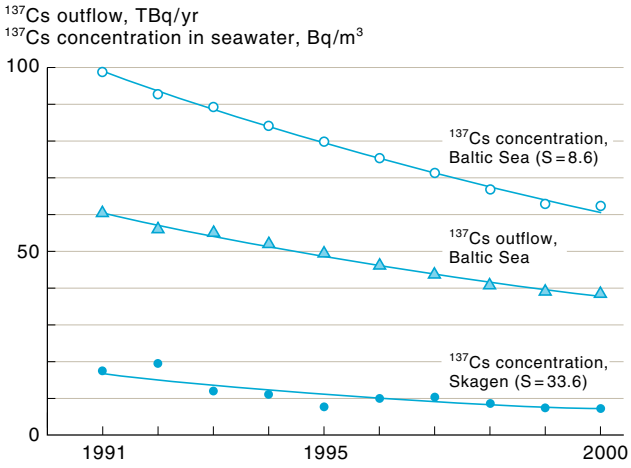

Figure 2·6. Net ^{137}Cs outflow from the Baltic Sea between 1991 and 2000 and concentrations in water masses of salinity 33.6 and 8.6 (Dahlgaard, 2002). The trend lines are an exponential fit.

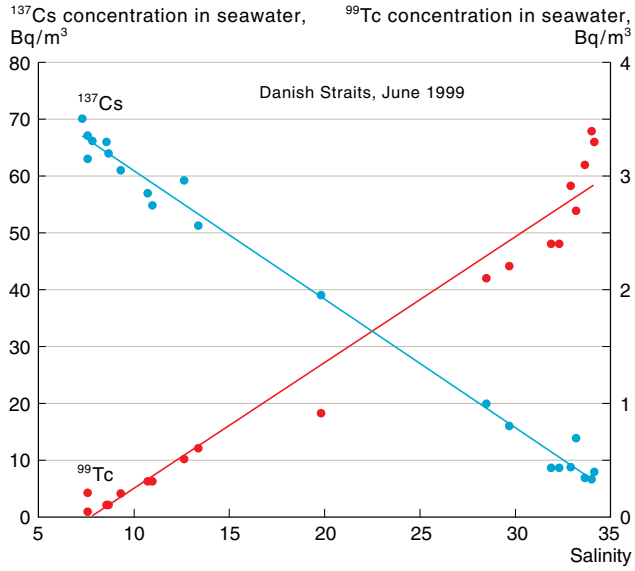

Figure 2·7. Concentrations of ^{137}Cs and ^{99}Tc in the Danish Straits in June 1999 as a function of salinity (Dahlgaard, 2002).

2.7. Radioisotope thermoelectric generators

Radioisotope thermoelectric generators (RTGs) have been used over the past few decades as local sources of electricity in Russia, the United States, and other countries. These work on the principle of radioactive decay (thermal) energy being converted into electric energy using thermoelectric converters. RTGs are used in inaccessible northern areas with harsh climatic conditions to supply power to navigation systems and beacons to improve maritime safety.

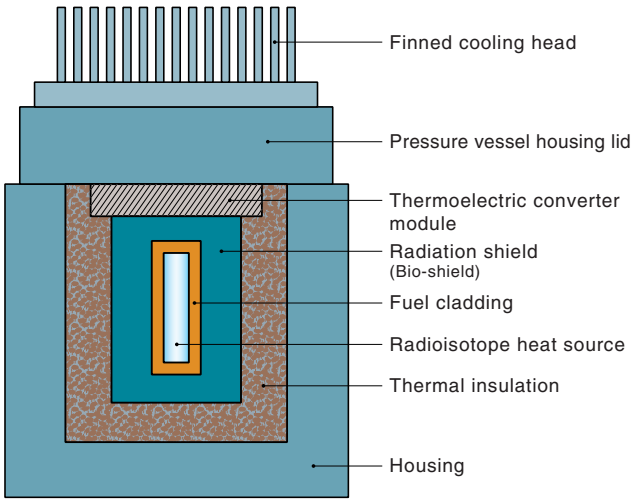

Figure 2·8. Schematic illustration of a typical RTG (Lamp, 1994).

One of the main components of an RTG is a radioisotopic heat source (RHS). Unlike radioactive radiation sources, an RHS contains much more radioactive material (isotope fuel) and operates at a higher temperature. It is therefore critical to maintain the integrity and tightness of RHS cladding during the service life of an RTG and in cases of possible emergency. This is achieved by creating a leak-tight, multi-envelope ampoule made of heat- and corrosion-resistant materials and by using heat-proof, and radiation- and chemically-stable isotope fuel that is compatible at high temperatures with the structural materials used in RHS cladding (Fadeev *et al.*, 1980). The fuel used in an RHS is primarily ^{90}Sr titanate ($SrTiO_3$ in Russia and the United States, or Sr_2TiO_4 in the United States only), which is a solid ceramic material. This has been selected for its strength, fire-resistance, and low water solubility.

The primary risk associated with RTGs is radiation exposure of humans, animals, plants, and the environment from the ^{90}Sr source (U.S. Congress, 1994). The absorbed dose rate from physical proximity to an operating RTG is <0.1 mGy/hr at a distance of 1 m from the RTG surface. Of greater concern is the resulting exposure if the inner shield of the RTG is breached allowing the release of ^{90}Sr into the environment. Natural disasters and most accidents associated with human activities present little risk of such ^{90}Sr releases to the environment. Greater risk is associated with individuals intentionally damaging or dismantling an RTG and not recognizing the hazards. In the event that radioisotope material is released, the dispersal area would not normally be large and clean-up activities are not difficult. Residual ^{90}Sr material in the environment will remain in an inert form (strontium titanate ceramic) with minimal uptake by plants and minimal incorporation into the food chain. Figure 2·8 shows a typical RTG construction.

2.7.1. RTGs in Russia

In Russia, RTGs are used in automatic meteorological stations in polar and uninhabited northern areas so that meteorological data can be continuously transmitted by radio. A network of RTGs, as stand-alone power supply sources, has also been established for navigation purposes. Increasing levels of maritime traffic have resulted in the creation of new high-latitude sea routes and an increase in the length of the navigation period in remote and uninhabited areas. Some of the designs of RTGs used in Arctic Russia are shown in Figure 2·9. The characteristics of RTGs used in Russia, including ^{90}Sr activities, are given in Table 2·4.

In Russia, there have been two cases (in 1987 and 1997) in which RTGs were lost due to emergency dumping at sea during their transport by helicopter using external suspension devices (Noie *et al.*, 1997). Both incidents occurred in the vicinity of Sakhalin Island. The two RTGs have not yet been found. However, monitoring in the area in which the RTGs were lost, including that conducted during a joint Russian–Japanese–Korean cruise (Noie *et al.*, 1997), did not detect increased levels of ^{90}Sr. In the Arctic region of Russia, there have been no losses of RTGs during transport, although on at least one occasion, RTGs on the Kola Peninsula have been sabotaged. In 2001, three RTGs were destroyed by people intending to steal parts. The open sources were located and removed and no radioactive contamination was detected (Fylkesmannen i Finnmark, 2001). Under

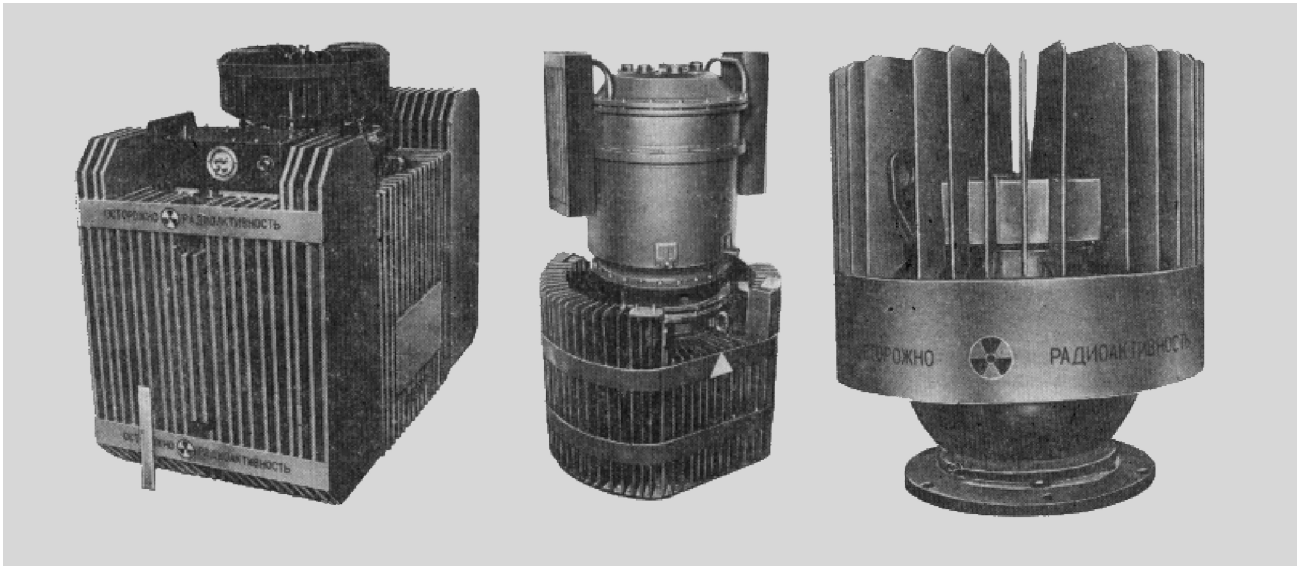

Figure 2·9. Examples of different types of RTG used in Arctic Russia (Kodyukov *et al.*, 1980). From left to right: 'Reut-1', 'Reut-2', 'Beta-M'.

Table 2·4. Basic characteristics of RTGs using strontium titanate (Fadeev *et al.*, 1980). Each row reflects data for different RTGs deployed in the Russian Arctic.

Range of operating temperature	Thermal capacity, W	^{90}Sr activity, PBq (Ci)
up to 450°C	148	0.81 (22000)
»	255	1.48 (40000)
»	200	1.15 (31000)
»	60	0.33 (9000)
»	650	3.7 (100000)
up to 600°C	260-290	1.48-1.67 (40000-45000)
»	240	11.1 (300000)
»	550	3.15 (85000)
»	200-260	1.11-1.48 (30000-40000)

Russian–Norwegian bilateral cooperation on environmental protection, RTGs are currently being replaced by photovoltaic systems (Fylkesmannen i Finnmark, 2001).

2.7.2. RTGs in the U.S.A.

The U.S. Air Force operates a seismic observatory on Burnt Mountain in Alaska (67°25'N, 144°36'W) for verifying compliance with nuclear test ban treaties. The data collection and communications equipment at the station is powered by ten RTGs; seven of model Sentinel 25E, one of model Sentinel 25F, one of model Sentinel 25A, and one of model Sentinel 100F (U.S. Congress, 1994). Some characteristics of the Sentinel RTGs are shown in Figure 2·10 and Table 2·5.

Figure 2·10. An RTG instrument housing at Burnt Mountain, Alaska (Photo by Stan Read, 1992).

In August and September 1992, a tundra fire encroached on the Burnt Mountain site. It damaged some data cables but did not disturb monitoring, communications, and power equipment. The fire raised concern among nearby inhabitants regarding the safety of radioactive material as the power source at the station.

Owing to the continuing high level of public concern, in 1999 the U.S. Air Force conducted an environmental assessment to evaluate the comparative safety of radioactive and non-radioactive sources of power (U.S. Air Force, 1999). It examined the safety of the RTGs at Burnt Mountain and assessed the viability and risks of two alternative power sources – thermoelectric and photovoltaic generators. As a result, a decision was made to replace the RTGs with a centralized hybrid power generation system. Planning has started for the removal of the RTGs and the installation of replacement power systems.

Currently, the U.S. Department of Energy and the U.S. Air Force are cooperating on the conduct of an environmental assessment to evaluate procedures for the removal and disposal of RTGs (Huizenga, 2001).

2.8. Russian reprocessing plants

2.8.1. The Siberian Chemical Combine

The Siberian Chemical Combine (SCC) complex is located 20 km from Tomsk near Seversk. This plant began operating in 1953 and comprises facilities posing potential threats to the population and the environment (Kryshev and Riazantsev, 2000). These include:

- uranium-graphite reactors used for power generation and the production of Pu;
- plant for the production of enriched uranium hexafluoride using centrifuges;
- plant for producing uranium mixed oxide fuel and uranium hexafluoride;
- radiochemical plant at which irradiated material is reprocessed to separate and purify U and Pu salts; and
- chemical-metallurgic plant for nuclear materials production.

Table 2·5. Some characteristics of the RTGs at Burnt Mountain, Alaska (U.S. Congress, 1994).

	Sentinel 25A	Sentinel 25E	Sentinel 25F	Sentinel 100F
Activity of initial charge, PBq (Ci)	3.48 (94000)	3.89-4.03 (105000-109000)	4.0 (108000)	12.2 (329000)
Year of initial charge	1968	1969-1971	1970	1972
Exposure rate at housing surface, mGy/hr	0.55	0.65	0.75	1.25

The activities of the SCC related to the production of Pu, U, and transuranic elements result in the generation of large quantities of liquid, solid, and gas-aerosol radioactive waste. The storage and disposal facilities for radioactive material, including underground disposal sites for liquid radioactive waste, are thus important components of the SCC. The total activity of liquid radioactive waste disposed of in deep underground strata by pumping through a system of injection pipes is estimated to be 1.5×10^{19} Bq, while the amount in open storage is about 4.6×10^{18} Bq (Kryshev and Riazantsev, 2000). There is some potential for contamination of the Arctic by radionuclides in SCC discharges through their transport to the Kara Sea by the Tom and Ob rivers. The SCC liquid discharges move from the sedimentation reservoir via the waste channel to the Romashka River, and then to the River Tom in the area of Chernilshchikovo. The wastewater contains ^{24}Na, ^{32}P, ^{46}Sc, ^{51}Cr, ^{59}Fe, ^{60}Co, ^{65}Zn, ^{76}As, ^{137}Cs, ^{152}Eu, ^{239}Np, ^{239}Pu, and other radionuclides. Historically, the major contribution to radioactivity in SCC wastewater was derived from the operation of reactors using direct flow (single pass) cooling systems. These were decommissioned between 1989 and 1992 and the release of radionuclides to the open hydrological network is now much lower. However, discharges of contaminated water in previous years have led to significant accumulation of radionuclides in bottom sediments and biota as well as in the Tom and Romashka River floodplains.

As an example, Table 2·6 presents annual discharges of selected radionuclides in SCC wastewater to aquatic systems in recent years. Other radionuclides subject to discharge monitoring (^{46}Sc, ^{51}Cr, ^{59}Fe, ^{60}Co, ^{65}Zn, ^{58}Co, ^{54}Mn, ^{90}Sr, ^{131}I, ^{144}Ce, and tritium) were not detected in 1998 and 1999, either at the site of wastewater discharge to the Tom River or at monitoring stations downstream.

River water and bottom sediments in the area around the SCC have been monitored continuously. As an illustration of the results, Table 2·7 and Figure 2·11 present recent data on the concentrations of some artificial and naturally-occurring radionuclides (Roshydromet, 2001).

The extent of radioactive contamination in water bodies in the near zone of the SCC has improved since the single pass coolant reactors were taken out of operation. Figure 2·12 shows temporal changes in gamma dose rates above the water surface near the right bank of the Tom River as a function of increasing distance from its confluence with the Romashka River. The years selected correspond to changes in plant operations: 1989 – prior to the first single pass reactor shutdown; 1990 – after the first single pass reactor shutdown; 1991 – following the second single pass reactor shutdown; and 1992 – following the third single pass reactor shutdown. The dose rates in 1992 were about an order of magnitude lower than in 1988 when all reactors were operational. At distances of >10 km downstream of the confluence of the two rivers, the observed dose rate was close to 0.1 Gy/hr as early as 1990. This is similar to background.

The distance between the estuary of the Ob River and the Tom inlet is about 2665 km. In 1977, studies were conducted on the influence of SCC discharges on radioactive contamination of the Ob River as far as the estuary (Vakulovsky, 1993). Analyses of water and bottom sediment samples detected ^{51}Cr, ^{58}Co, ^{60}Co, ^{54}Mn, ^{65}Zn, and ^{46}Sc. Most of the ^{51}Cr occurred in the dissolved fraction whereas 70 to 95% of the ^{58}Co, ^{60}Co, ^{54}Mn, ^{65}Zn, and ^{46}Sc occurred in the suspended fraction. The particle reactivity of most isotopes in the river has resulted in concentrations in water remaining reasonably low.

The relationship between the flux of radioactive material transported in the suspended fraction and distance is virtually exponential with the flux being reduced by a factor of two in 250 to 300 km. Manganese-54 exceeded the detection limit of 3.7 Bq/m^3 (1×10^{-13} Ci/L) 1500 km upstream of the Ob Estuary. Chromium-51, ^{58}Co, and ^{60}Co were detectable 2000-2200 km, and ^{46}Sc up to 2500 km upstream of the estuary.

Table 2·6. Examples of the levels of activity (TBq/yr (Ci/yr)) in wastewater discharged to the aquatic environment from the Siberian Chemical Combine (Roshydromet, 2000, 2001).

	Authorized discharge	Actual discharge 1998	Actual discharge 1999
^{24}Na	777 (21000)	208 (5626)	196 (5285)
^{32}P	40.7 (1100)	25 (689)	27 (729)
^{239}Np	14.8 (400)	7.7 (207)	11.4 (307)

Table 2·7. Activity concentrations (Bq/kg dw) for radionuclides in bottom sediments in the vicinity of the Siberian Chemical Combine in 1999 (Roshydromet, 2001).

		Tom River			
	Romashka River	500 m downstream of confluence with the Romashka River	1000 m downstream of confluence with the Romashka River	1500 m downstream of confluence with the Romashka River	Chernilshchiki Channel
^{46}Sc	67.5	86.9	130	39.2	29.8
^{137}Cs	63.6	83.2	458	76.6	85.6
^{134}Cs	6.9	11.8	8.0	7.9	4.1
^{54}Mn	16.3	14.5	22.8	27.4	18.2
^{51}Cr	477	510	483	243.3	146
^{60}Co	260	337	388	152.4	116
^{152}Eu	42.7	62.8	237	26	17.5
^{154}Eu	7.8	18	45.1	9.2	7.8
^{59}Fe	25.4	23.2	24.6	10.9	8.8
^{65}Zn	284	360	337	144	99
^{40}K	386	457	510	430	433
^{226}Ra	10.6	12.7	27.5	18.3	18.2
^{232}Th	15.6	13.6	37.5	22	20.7

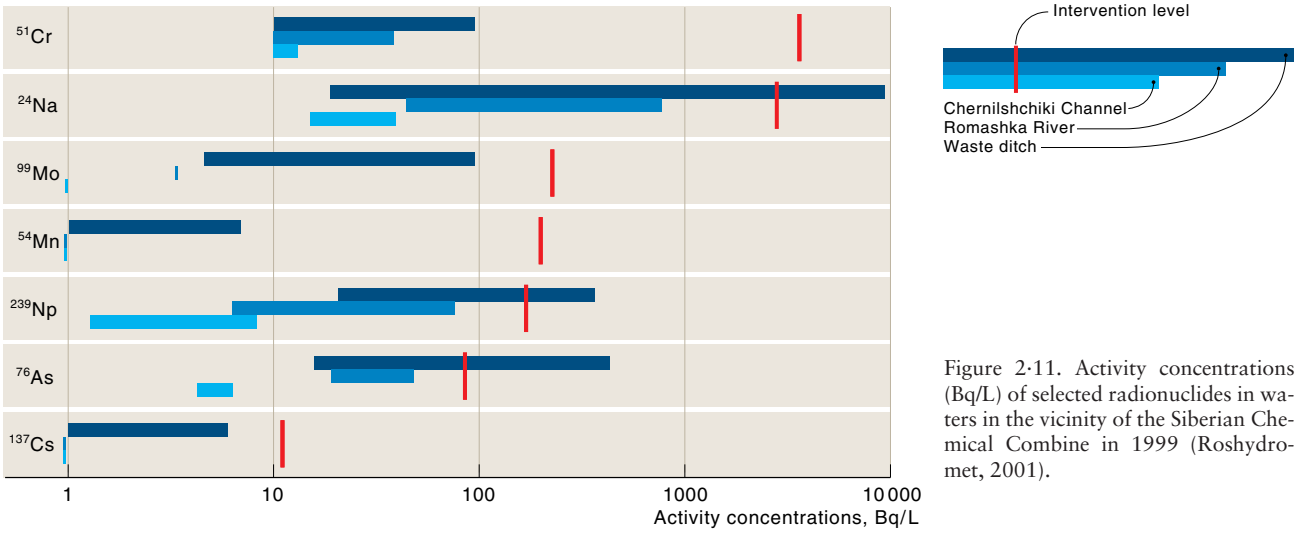

Figure 2·11. Activity concentrations (Bq/L) of selected radionuclides in waters in the vicinity of the Siberian Chemical Combine in 1999 (Roshydromet, 2001).

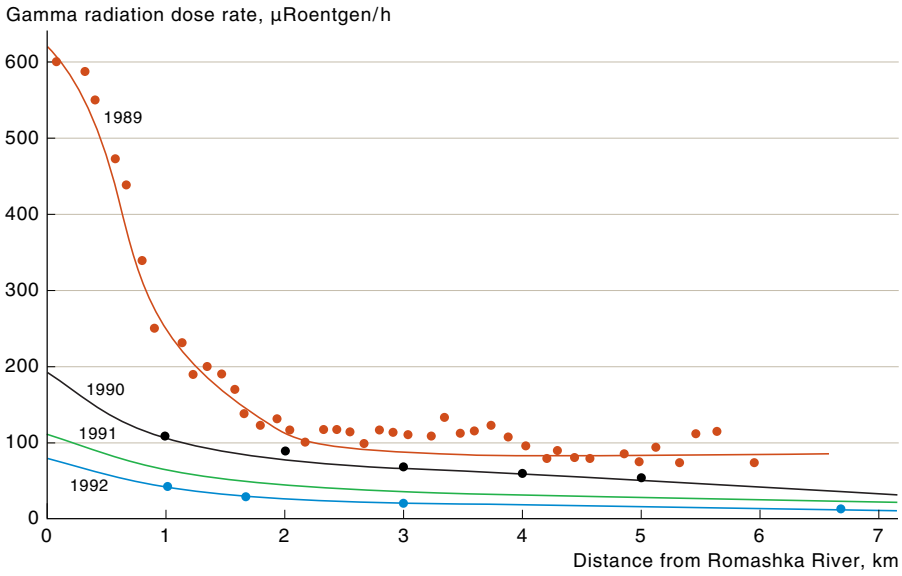

Figure 2·12. Variations in the gamma radiation dose rate above the water near the right bank of the Tom River as a function of distance downstream from the Romashka River confluence (Roshydromet, 1993).

Thus, most of the radionuclides discharged from the SCC are removed from the water column during transport along the Tom/Ob river system and are not detectable in the lower reaches of the Ob or in the Ob Estuary (Kuznetsov, 1995; Vakulovsky, 1993). Studies carried out in 1994 showed that global atmospheric fallout is the predominant source of the long-lived radionuclides, ^{137}Cs and ^{239,240}Pu, in bottom sediments of the Ob delta and estuary (Panteleyev *et al.*, 1995; Sayles *et al.*, 1999).

2.8.2. The Mining and Chemical Industrial Complex of Zheleznogorsk

The Krasnoyarsk Mining and Chemical Industrial Complex (KMCIC) is located 9 km from Zheleznogorsk and covers 360 km^2 on the right bank of the Yenisey River. Currently, the KMCIC includes a reactor facility (one uranium-graphite reactor with a closed cooling circuit), a radiochemical plant processing irradiated U disks, a facility for the storage of spent fuel assemblies, and a radioactive waste processing plant (Kryshev and Riazantsev, 2000).

The first industrial direct-flow (single pass cooling loop) reactor of the KMCIC began operating in August 1958 and the second in 1961. The third reactor with a closed circuit became operational in 1964. The direct flow uranium-graphite reactors, AD and ADE-1, were shut down in 1992 (Kryshev and Riazantsev, 2000).

The reactor and radiochemical production plant constitute the main sources of environmental contamination due to gas-aerosol release and water discharges containing radionuclides. Another potential source of contamination is the site where liquid radioactive waste of varying activity is buried underground (the total activity of accumulated liquid radioactive waste is about 3.7×10^{18} Bq; Kryshev and Riazantsev, 2000). There are also solid radioactive waste storage facilities and disposal sites within the KMCIC. All radioactive waste produced by the KMCIC is buried within its site boundary.

In terms of possible radioactive contamination of the Arctic, consideration must first be given to the water transport of radionuclides originating from the KMCIC to the Kara Sea; namely transport via the Yenisey River. Cooling waters from the operating reactor are retained in a pool to allow physical decay of short half-life radionuclides. The water is then discharged to the Yenisey River along with other reactor releases and discharges from the radiochemical production plant several kilometers downstream. Because only one power reactor is now in use and the pool reduces the concentrations of released

Table 2·8. Discharges (GBq (Ci)) of selected radionuclides in wastewater from the Mining and Chemical Industrial Complex of Zheleznogorsk (Roshydromet, 2001).

	Authorized discharge	Actual discharge 1998	Actual discharge 1999
^{24}Na	185000 (5000)	57600 (1560)	72600 (1960)
^{32}P	22200 (600)	7360 (199)	9840 (266)
^{46}Sc	370 (10)	30 (0.81)	41.4 (1.12)
^{51}Cr	14800 (400)	2580 (69.7)	3170 (85.6)
^{54}Mn	148 (4)	3.8 (0.104)	4.1 (0.11)
^{60}Co	374 (10.1)	50.5 (1.37)	50.9 (1.38)
^{64}Cu	5550 (150)	1120 (30.3)	1490 (40.2)
^{65}Zn	370 (10)	34 (0.92)	31.1 (0.84)
^{76}As	5550 (150)	738 (19.9)	1070 (28.9)
^{90}Sr	81.4 (2.2)	3.6 (0.097)	41.1 (1.11)
^{131}I	555 (15)	17.4 (0.47)	45.9 (1.24)
^{134}Cs	29.6 (0.8)	0.56 (0.015)	0.56 (0.015)
^{137}Cs	114.7 (3.1)	64.4 (1.74)	42.2 (1.14)
^{152}Eu	185 (5)	2.3 (0.062)	2.96 (0.08)
^{239}Np	7400 (200)	4670 (126)	7100 (192)

nuclides, there has been a significant decrease in concentrations in the Yenisey River as well as in aquatic plants and fish, especially for the short-lived isotopes. However, bottom sediments and the river floodplain are contaminated with long-lived radionuclides such as ^{60}Co, ^{137}Cs, and ^{152}Eu as a result of previous releases, largely those from the two direct-flow reactors. Since 1992, discharges of radioactivity in wastewater from the KMCIC have not led to additional contamination of bottom sediments above previous levels (Roshydromet, 1998). Table 2·8 presents data on the discharges of selected radionuclides in wastewater from the KMCIC in 1998 to 1999.

There is continuous monitoring of environmental contamination in the vicinity of the KMCIC. Table 2·9 provides, as an example, data on the concentrations of selected artificial radionuclides in river water in the near zone of the facility in 1999.

Table 2·9. Concentrations (Bq/L) of selected radionuclides in Yenisey River water in the vicinity of the Mining and Chemical Industrial Complex of Zheleznogorsk in 1999 (Roshydromet, 2001).

	Monitoring point 250 m downstream of the outlet	Monitoring point 10 km downstream of the outlet	Intervention level NRB-99 (Minzdrav, 1999)
^{24}Na	10.7	2.2	–
^{32}P	0.73	0.14	58
^{46}Sc	<0.01	<0.001	93
^{51}Cr	0.4	<0.1	3700
^{54}Mn	<0.001	<0.0002	200
^{59}Fe	<0.01	<0.001	77
^{58}Co	<0.01	<0.001	19
^{60}Co	0.005	0.0017	41
^{65}Zn	<0.01	<0.001	36
^{76}As	<0.2	<0.03	87
^{90}Sr	0.0078	0.0069	5
^{95}Zr	<0.001	<0.0002	150
^{95}Nb	<0.001	<0.0002	240
^{103}Ru	<0.001	<0.0002	190
^{106}Ru	<0.002	<0.0002	20
^{131}I	<0.01	<0.002	6.3
^{134}Cs	<0.0003	<0.00005	7.3
^{137}Cs	<0.004	0.0018	11
^{152}Eu	<0.001	<0.0002	99
^{239}Np	<1.3	<0.2	170

The concentrations of artificial radionuclides in the Yenisey River in the vicinity of the KMCIC are currently much lower than intervention levels defined in NRB-99 (Minzdrav, 1999). In addition to continuous monitoring in the vicinity of the KMCIC, monitoring of radioactive contamination is regularly conducted throughout the Yenisey River system (the KMCIC is 2400 km from the Yenisey outlet to the Kara Sea). One study measured river contamination between 1971 and 1993. This included analyses of bottom sediments, which are good indicators of river ecosystem contamination. Table 2·10 presents some results of the bottom sediment survey of August 1973. This study showed the impact of the KMCIC to be detectable to a distance of about 2000 km downstream of the plant and thus into the Arctic. Activity concentrations in bottom sediments at the boundary of the far zone of the Krasnoyarsk contamination trace are a thousand times lower than those in the near zone, but still exceed levels expected from global fallout.

Table 2·10. Activity concentrations (kBq/m^2) in bottom sediments of the Yenisey River in August 1973 (Vakulovsky *et al.*, 1995).

	Distance from discharge source 6 km	250 km	800 km	1930 km
^{46}Sc	330	26	2.4	<2
^{51}Cr	1550	<6	<6	<6
^{54}Mn	220	48	2.3	<0.2
^{58}Co	200	32	3.8	<0.7
^{60}Co	440	65	4.8	<0.3
^{59}Fe	400	15	<3	<3
^{65}Zn	1470	260	23	<0.6
^{134}Cs	140	2	<0.4	<0.4
^{137}Cs	460	90	18	4.4
^{144}Ce	110	34	3.5	<3
^{152}Eu	220	36	2.2	<0.2
^{154}Eu	70	12	<0.4	<0.4
Total	5800	620	60	5

Data on the concentrations of radionuclides in river water in the far zone impacted by the KMCIC for different periods of observation are shown in Table 2·11. Beyond the near zone the contamination of water decreased markedly although trace amounts of artificial radionuclides were detected at considerable distances (>1300 km) from the discharge. Nevertheless, activity concentrations in river water are several orders of magnitude lower than the intervention levels regulated by the radiation safety norms.

Most of the discharged radionuclides (^{46}Sc, ^{54}Mn, ^{58}Co, ^{59}Fe, ^{60}Co, and ^{65}Zn) are transported on suspended particles. Only ^{24}Na and ^{51}Cr are transported mainly in dissolved form. As most of the ^{51}Cr occurs in dissolved form it can be transported over long distances (Table 2·11). These monitoring data, combined with water discharge data for the Yenisey River, can be used to estimate the fluxes of radionuclides from the near and far zones of the KMCIC before the removal from service of the two single pass reactors. Estimated average fluxes for 1985 to 1991 are shown in Table 2·12.

The results suggest that the fluxes of long-lived radionuclides from the far zone were modest. Thus, these discharges are only likely to have had a minor influence on radioactive contamination of Arctic seas.

Table 2·11. Activity concentrations in Yenisey River water (Vakulovsky *et al.*, 1995).

	Intervention level, mBq/L	Distance from discharge source, km	Activity concentration, mBq/L average 1985-1991	1992	1993
^{46}Sc	93000	250	11	<0.7	<0.7
		850	5	<0.7	<0.7
		1360	5	<0.7	<0.7
^{51}Cr	3700000	250	1900	700	<14
		850	1100	150	<14
		1360	700	160	<14
^{54}Mn	200000	250	8	<0.7	<0.7
		850	3	<0.7	<0.7
		1360	3	<0.7	<0.7
^{58}Co	19000	250	26	4	<1.5
		850	12	<1.5	<1.5
		1360	7	10	<1.5
^{60}Co	41000	250	7	<1.5	<1.5
		850	4	8	<1.5
		1360	4	<1.5	<1.5
^{59}Fe	77000	250	42	<3	<3
		850	7	<3	<3
		1360	8	<3	<3
^{65}Zn	65000	250	23	<2	<2
		850	<2	<2	<2
		1360	<2	<2	<2
^{137}Cs	11000	250	6	2	1
		850	5	6	2
		1360	5	2	2

Of the long-lived radionuclides, only ^{137}Cs can reach the Kara Sea in significant quantities. The amount of ^{137}Cs entering the Kara Sea in the Yenisey River outflow is estimated to have averaged between 0.8 and 2.8 TBq/yr over the period 1958 to 1993. This includes ^{137}Cs from both global fallout and discharges from the KMCIC and should be considered an upper limit. Thus, routine discharges from the KMCIC over this period could have resulted in an input of 30 to 100 TBq of ^{137}Cs to the Kara Sea (Vakulovsky *et al.*, 1995). This is less than the ^{137}Cs input from atmospheric fallout (about 1.4 PBq for the Kara Sea) and comparable with the input to the Kara Sea from Sellafield, about 150 TBq (NRPA, 1993).

Table 2·12. Estimated average fluxes (TBq/yr) of radionuclides in the Yenisey River for 1985 to 1991 (Vakulovsky *et al.*, 1995).

	Near zone (15 km from the discharge)	Far zone (1360 km from the discharge)
^{32}P	1800 ±1000	–
^{46}Sc	9 ±4	1.7±1.0
^{51}Cr	2300 ±1200	250 ± 250
^{54}Mn	9 ±5	1.0±0.7
^{58}Co	40 ±20	2.5±1.4
^{60}Co	5 ±1	1.4±0.8
^{59}Fe	20 ±10	2.8±1.0
^{65}Zn	17 ±8	–
^{90}Sr	1.6±0.7	1.5±0.6
^{137}Cs	6 ±3	1.8±1.0

2.9. Nuclear detonations

The total number of nuclear detonations, including the first nuclear test in 1945, has been 2419, with an aggregated yield of 530 Mt of TNT equivalent (UNSCEAR, 2000). The vast majority (1876) of these detonations were underground, but with a comparatively low aggregate yield of 90 Mt. In contrast, the 543 atmospheric detonations contributed 440 Mt or 83% of the aggregate yield (UNSCEAR, 2000). The calculations of yield are based on certain assumptions about partitioning between fission and fusion yields in individual tests. The reliability of these assumptions will determine uncertainties in the estimates of yield. The largest 25 atmospheric nuclear tests, those exceeding 4 Mt, account for nearly 66% of the total explosive yield and about 55% of the estimated total (fission plus fusion) yield.

The largest atmospheric test was an air detonation with a yield of 50 Mt conducted by the former Soviet Union at the Novaya Zemlya test site on 30 October 1961. The largest underground nuclear tests in the Arctic were a 1.5 to 10 Mt test conducted by the former Soviet Union at the Novaya Zemlya test site on 27 October 1973, and a test of approx. 5 Mt conducted by the United States at Amchitka, Alaska, on 6 November 1971. In addition, the former Soviet Union conducted a number of crater detonations where nuclear devices were detonated at shallow depths for non-military purposes.

Recently, there has been new information on local contamination resulting from underground nuclear explosions for non-military purposes carried out by the former Soviet Union. Research has shown that these explosions led to considerable contamination of localized areas. The sites 'Kraton-3' and 'Crystal' in the Republic of Sakha, where such tests were conducted, contain residual radioactive contamination despite earlier clean-up efforts. Concentrations of Pu in lichen in the vicinity of the Kraton-3 and Crystal sites are relatively high. The average plutonium concentration in lichen in the most contaminated area surrounding the Kraton-3 site is 2.1

kBq/kg, which is a factor of 780 greater than the background level created by global fallout. However, such levels of contamination only exist in the immediate vicinity of such sites. For instance, at the Crystal site, ^{239,240}Pu activity concentrations in soils and lichens about 2 km from the crater are three to four orders of magnitude lower than the maximum values. Measurements of the activity concentration of $^{239+240}$Pu in bottom sediments suggest there has been a loss of radionuclides into the Markha River with the potential for subsequent remobilization and transport over larger areas (Gedeonov *et al.*, 1999).

The United States conducted underground nuclear explosions on Amchitka Island between 1965 and 1971. With the exception of radioactive gas leakage during the Long Shot test, radionuclides in Amchitka freshwater and terrestrial environments are essentially of fallout origin (Dasher *et al.*, 2002). This is consistent with hydrogeological knowledge and some limited modelling that indicates leakage from the underground test cavities will eventually occur into the nearshore marine environment (Wheatcraft, 1995), see also Section 3.7.3.

2.10. Operational releases from nuclear power plants

The locations of the nuclear power plants (NPPs) discussed in this section are shown in Figure 2·13.

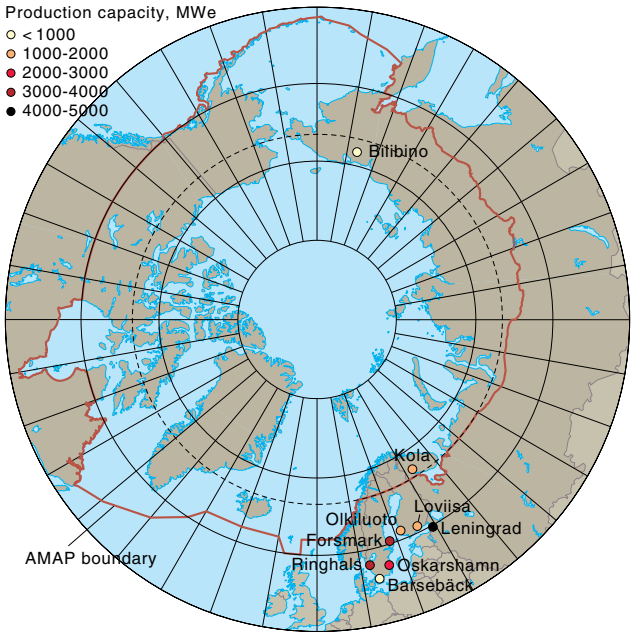

Figure 2·13. Nuclear power plants within or near the AMAP area.

Figure 2·14 shows releases between 1995 and 2000 for NPPs in Sweden, Finland, and northern Russia. Not all types of release are documented for all NPPs.

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Figure 2·14. Atmospheric releases from nuclear plants of a) noble gases, b) ^{131}I (not detected for Bilibino NPP), c) aerosols (no data for Russian NPPs); and discharges to water of d) ^{137}Cs (no data for Leningrad and Bilibino NPPs and Ringhals 2-4), and e) ^{60}Co (no data for Leningrad and Bilibino NPPs). See Annex Tables 2·A1-2·A9.

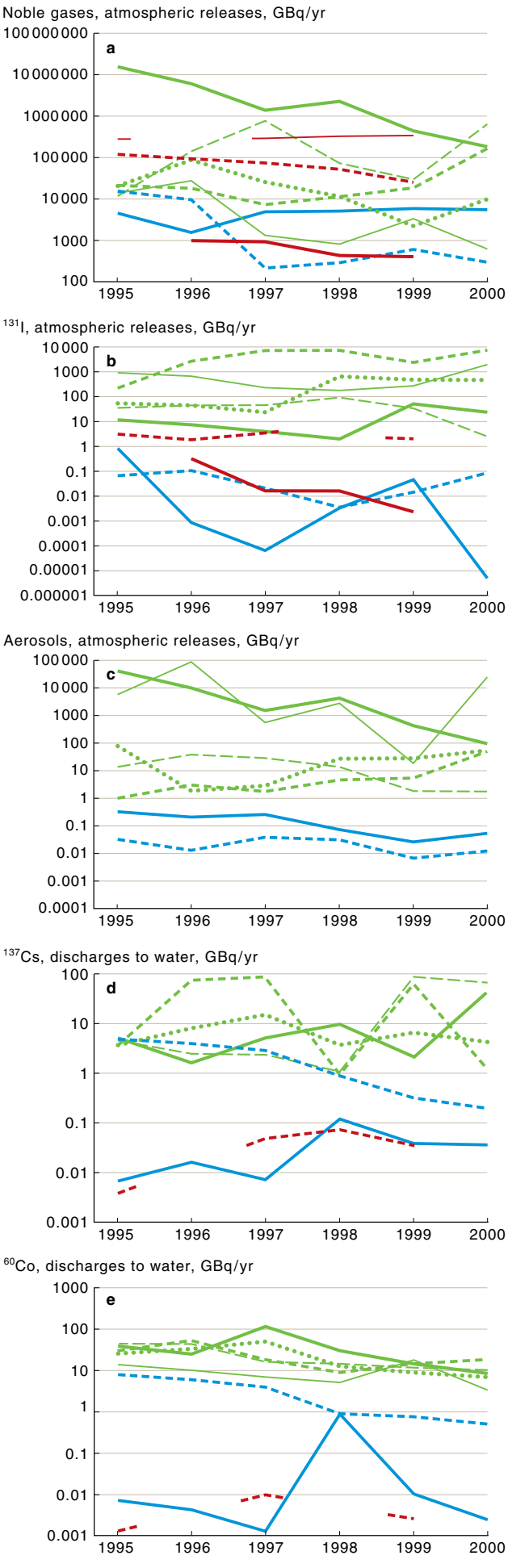

Table 2·13. Details of Finnish nuclear power plants. PWR: pressurized water reactor; BWR: boiling water reactor.

Unit (company)	Reactor type/model	Installed capacity, gross/net, MW(e)	Commercial start-up date
Loviisa 1 (Fortum Power and Heat Oy)	PWR/VVER-440/V213	510/488	1977
Loviisa 2 (Fortum Power and Heat Oy)	PWR/VVER-440/V213	510/488	1981
Olkiluoto 1 (Teollisuuden Voima Oy, TVO)	BWR	870/840	1979
Olkiluoto 2 (Teollisuuden Voima Oy, TVO)	BWR	870/840	1982

2.10.1. **Finnish NPPs**

Finland has two NPPs both situated on the Baltic Sea coast: the Loviisa NPP on the Gulf of Finland and the Olkiluoto NPP on the Gulf of Bothnia. Two reactor units are in operation at both sites (see Table 2·13).

Annex Table A2·2 tabulates the annual radionuclide releases (GBq/yr) from each of the two NPPs for 1995 to 2000 for the six most abundant radionuclides in liquid effluent that also contribute most to individual dose. Corresponding releases in gaseous effluents are presented in Annex Table A2·1.

The principal noble gas release from Loviisa has been ^{41}Ar and, for Olkiluoto, xenon isotopes. Among aerosols, the main releases from Loviisa have been ^{60}Co, ^{110m}Ag, and ^{124}Sb, and from Olkiluoto ^{54}Mn, ^{58}Co, and ^{60}Co. The annual release limits for the two NPPs are shown in Table 2·14.

Table 2·14. Annual release limits (GBq) for the Loviisa and Olkiluoto nuclear power plants (AMAP Data Centre; Tossavainen, 2002).

	Liquid effluents		Gaseous effluents	
	^{3}H	Other nuclides	Noble gases*	Iodines*
Loviisa	150000	890	22000000	220
Olkiluoto	18000	300	18000000	110

* in ^{131}I equivalents

2.10.2. **Russian NPPs**

There are two NPPs in the Russian Arctic; at Kola and Bilibino. Of the Russian NPPs outside the Arctic, the Leningrad NPP is of interest within the context of AMAP assessments. The atmospheric releases of radionuclides from these three plants and liquid effluents from the Kola and Bilibino NPPs in 1994 to 1995 were reviewed during the first AMAP assessment. Annexes A2·3 and A2·4 show atmospheric releases and liquid discharges of radionuclides from the Kola NPP in recent years.

Data on atmospheric releases of radionuclides from the Bilibino NPP in recent years are presented in Annex A2·5. The ^{60}Co discharge from this NPP in 1999 was 0.03% of the permitted discharge and for other man-made radionuclides discharges were below detection limits (Roshydromet, 2001). Data for the Leningrad NPP atmospheric releases are given in Annex Table A2·6.

2.10.3. **Swedish NPPs**

Sweden has four NPPs: two on the east coast (Forsmark and Oskarshamn on the Baltic Sea) and two on the west coast (Ringhals on the Kattegatt and Barsebäck on the Öresund), as specified in Table 2·15.

No specific discharge limits have been formally defined by the Swedish Radiation Protection Authority. Limits are currently imposed through restrictions on doses to members of critical groups (see Chapter 4). Thus, for each nuclear facility and for each nuclide released, site-specific release-to-dose conversion ratios have been calculated. These values pertain to doses to individuals in hypothetical critical groups and take into consideration local dispersion conditions and moderately conservative assumptions on dietary conditions and the contribution of locally-produced foodstuffs.

Measurements of actual releases of ^{129}I were not requested under previous regulations. Likewise, emissions of ^{14}C and ^{3}H were not routinely measured. Emissions of ^{14}C were estimated on the basis of international experience as 0.2 TBq/GWyr for pressurized-water reactors and 0.6 TBq/GWyr for boiling-water reactors. However, new regulations require all nuclides to be measured. Releases are given in Annex Tables A2·7 to A2·9.

Table 2·15. Details of Swedish nuclear power plants. BWR: boiling water reactor; PWR: pressurized-water reactor.

Site	Discharges to	Reactor type	Net electric output, MW(e)	Commercial start up
Barsebäck 1	Öresund	BWR, ABB-Atom	600	1975*
Barsebäck 2	Öresund	BWR, ABB-Atom	600	1977
Forsmark 1	Baltic Sea	BWR, ABB-Atom	968	1980
Forsmark 2	Baltic Sea	BWR, ABB-Atom	969	1981
Forsmark 3	Baltic Sea	BWR, ABB-Atom	1158	1985
Oskarshamn 1	Baltic Sea	BWR, ABB-Atom	445	1972
Oskarshamn 2	Baltic Sea	BWR, ABB-Atom	605	1975
Oskarshamn 3	Baltic Sea	BWR, ABB-Atom	1160	1985
Ringhals 1	Kattegat	BWR, ABB-Atom	830	1976
Ringhals 2	Kattegat	PWR, Westinghouse	870	1975
Ringhals 3	Kattegat	PWR, Westinghouse	915	1981
Ringhals 4	Kattegat	PWR, Westinghouse	915	1983

*ceased to generate electricity 1 December 1999 when the reactor was shut down.

2.10.4. Camp Century, Greenland

A portable pressurized-water reactor designated PM-2A was operated by the U.S. Army at Camp Century in Greenland between 1960 and 1963. Power capacity was 1560 kW(e); authorized power was 10000 kW(th). The plant was installed at an experimental ice camp for the provision of power and heat. The camp and the reactor were embedded in the surface layer of the Greenland ice cap at 77°11'N, 61°08'W. The plant was shut down in July 1963, and dismantled and transported to the United States during April to June 1964. The existence of the base and the reactor was made public and was featured in the National Geographic Magazine in the 1960s.

2.11. Summary

Significant new information on sources of actual and potential radioactive contamination of the Arctic environment has been provided to complement that published at the conclusion of AMAP Phase I. Evidence is presented regarding the transport of Pu derived from the Sellafield reprocessing plant discharges into the AMAP area as a result of the remobilization of Pu isotopes from bottom sediments of the Irish Sea. Previous activities at the nuclear facilities in Siberia (the SCC and the KMCIC) do not appear to have led to any significant radioactive contamination of the Arctic as a result of water transport, due to the effective scavenging of radionuclides into bottom sediments of the upstream reaches of the Ob and Yenisey rivers.

Additional investigations have provided improved estimates of the activity in dumped nuclear reactor compartments containing spent nuclear fuel and other dumped objects in the Kara Sea. These suggest that the previous estimate of the aggregate activity in dumped reactors was overestimated while that for the spent nuclear fuel from the icebreaker *Lenin* was underestimated. A recalculation of the amount of excess Pu in the environment of Bylot Sound, taking more complete account of the incidence of 'hot particles', has yielded a higher value than previous estimates, but with a large uncertainty. The median value obtained from these studies is more comparable with the estimates of unrecovered Pu from the crash of a B-52 aircraft in 1958.

As no information on current storage and handling of nuclear weapons in the Arctic was available, it was not possible to make any assessment of these subjects.

Monitoring of radioactive contamination in the vicinity of the *Kursk* submarine that sank in the Barents Sea in August 2000 has shown that the loss of this submarine and its subsequent recovery have not led to significant releases of radionuclides to the environment.

Annex. Tables

Table A2·1. Annual atmospheric releases (GBq) from the Loviisa and Olkiluoto NPPs in Finland (AMAP Data Centre; Tossavainen, 1998, 1999, 2000, 2001).

	1995	1996	1997	1998	1999	2000
Loviisa						
Noble gases*	4600	1600	4900	5300	6000	5600
^{3}H	190	220	250	190	180	200
^{14}C	140	99	230	340	320	280
Aerosols	0.35	0.22	0.25	0.073	0.027	0.062
Iodines*	0.77	0.0009	0.000072	0.0033	0.045	0.0000057
Olkiluoto						
Noble gases*	17000	9700	210	300	610	300
^{3}H	130	210	300	440	520	460
^{14}C	640	650	670	720	760	750
Aerosols	0.034	0.014	0.045	0.032	0.0065	0.013
Iodines*	0.067	0.11	0.018	0.003	0.015	0.084

*Noble gases expressed as ^{87}Kr equivalents and iodines as ^{131}I equivalents.

Table A2·2. Annual releases (GBq) of the most abundant radionuclides in liquid effluents from the Loviisa and Olkiluoto NPPs in Finland (AMAP Data Centre).

	1995	1996	1997	1998	1999	2000
Loviisa						
^{3}H	12 000	9400	12 000	9300	14 000	11 000
^{60}Co	0.0073	0.0042	0.0013	0.89	0.01	0.0027
^{110m}Ag	0.0092	0.0016	0.00026	0.082	0.014	0.029
^{124}Sb	0.032	0.016	0.00034	0.032	0.042	0.022
^{134}Cs	0.0043	0.016	0.0025	0.039	0.0084	0.0046
^{137}Cs	0.0064	0.016	0.0067	0.12	0.04	0.038
Olkiluoto						
^{3}H	1500	2400	1300	1200	1100	1000
^{54}Mn	3.6	1.4	0.92	0.11	0.09	0.049
^{58}Co	1.6	0.82	0.68	0.12	0.25	0.11
^{60}Co	8.1	5.9	3.7	0.9	0.79	0.5
^{134}Cs	4.4	2.5	1.3	0.28	0.071	0.043
^{137}Cs	4.9	4	2.8	0.88	0.32	0.21

Table A2·3. Annual atmospheric releases (GBq) from the Kola NPP (Energgoatomizdat, 1998; Roshydromet, 2000, 2001; U.S.S.R. Goshydromet, 1991).

	Authorized release	Actual release 1995	1996	1997	1998	1999
Noble gases	2 700 000	129 000	101 000	75 500	52 900	26 400
^{131}I	540	3.2	1.81	3.3	-	2.1
Long-lived radionuclides	810	2.3	0.925	0.2	0.044	0.367

Table A2·4. Annual releases of liquid effluents and wastewater (GBq) from the Kola NPP (Energgoatomizdat, 1998; Roshydromet, 2000, 2001; U.S.S.R. Goshydromet, 1991).

	1995	1996	1997	1998	1999
^{51}Cr	–	–	n.d.	–	0.0278
^{54}Mn	0.00022	–	n.d.	–	0.0013
^{58}Co	0.000074	–	n.d.	–	0.0012
^{60}Co	0.00137	–	0.01	–	0.0028
^{65}Zn	–	–	–	–	440 000
^{131}I	–	–	–	–	1900 000
^{134}Cs	1 220 000	–	92 000 000	23 300 000	780 000
^{137}Cs	0.0037	–	0.046	0.077	0.0374

n.d.: below detection limit.

Table A2·5. Annual atmospheric releases (GBq) from the Bilibino NPP (Energgoatomizdat, 1998; Roshydromet, 2000, 2001; U.S.S.R. Goshydromet, 1991).

	1995	1996	1997	1998	1999
Noble gases	293 000	–	304 000	339 000	349 000
^{131}I	n.d.	–	n.d.	–	n.d.
Long-lived radionuclides	n.d.	–	n.d.	–	n.d.

n.d.: below detection limit.

Table A2·6. Annual gas-aerosol releases (GBq) of radionuclides from the Leningrad NPP (Energgoatomizdat, 1998; Roshydromet, 2000, 2001).

	Authorized release	Actual release 1996	1997	1998	1999
Noble gases	27 000	1000	960	440	410
Long-lived radionuclides	0.810	0.052	0.023	0.016	0.0020
^{131}I	0.540	0.029	0.017	0.017	0.0021
^{90}Sr	0.0027	0.000031	0.000007	0.000016	0.0000067
^{137}Cs	0.027	0.0013	0.00052	0.0007	0.0003
^{60}Co	0.027	0.000074	0.00014	0.00017	0.00020
^{54}Mn	0.027	0.00017	0.00017	0.00021	0.00013
^{51}Cr	0.027	0.011	0.0041	0.0093	0.00052

Table A2·7. Annual atmospheric releases (GBq) from Swedish NPPs (AMAP Data Centre).

		1995	1996	1997	1998	1999	2000
Noble gases	Ringhals 1	15 700 000	6 690 000	1 310 000	2 340 000	463 000	192 000
	Ringhals 2-4	15 300	20 900	1330	811	3220	611
	Barsebäck 1-2	22 100	17 900	7320	10 900	19 300	158 000
	Forsmark 1-3	19 800	87 000	25 600	11 400	2330	9800
	Oskarshamn 1-3	112 000	138 000	794 000	74 000	31 600	670 000
^{131}I	Ringhals 1	12.3	7.46	4.20	2.00	52.0	23.5
	Ringhals 2-4	930	650	202	180	288	1970
	Barsebäck 1-2	208	2680	7930	7420	2200	7900
	Forsmark 1-3	57.9	44.6	23.2	652	488	446
	Oskarshamn 1-3	33.9	45.1	46.1	92.0	22.4	2.31
Particulate/ aerosol	Ringhals 1	44 700	10 600	1704	4450	401	92.9
	Ringhals 2-4	5140	88 000	504	2500	19.1	2290
	Barsebäck 1-2	1.00	3.06	1.60	4.49	5.09	47.0
	Forsmark 1-3	84.4	1.84	2.77	27.7	20.9	51.8
	Oskarshamn 1-3	14.0	40.8	30.5	13.7	1.83	1.89

Table A2·8. Annual discharges (GBq) to water from Swedish NPPs.

		1995	1996	1997	1998	1999	2000
^{3}H	Ringhals 1	832	790	490	550	986	514
	Ringhals 2-4	21000	24600	22500	25300	39600	25900
	Barsebäck 1-2	554	1100	760	490	690	400
	Forsmark 1-3	2340	1990	2000	1530	1420	1400
	Oskarshamn 1-3	1190	1380	1360	1110	1290	1180
Other nuclides	Ringhals 1	69.5	47.9	155	52.4	29.4	11.4
	Ringhals 2-4	81.1	48.2	47.3	40.6	41.8	24.7
	Barsebäck 1-2	57.8	194	58.3	35.7	26.4	24.7
	Forsmark 1-3	60.5	72.4	115	25.5	25.2	18.2
	Oskarshamn 1-3	97.6	130	51.1	82.0	26.9	20.9

Table A2.9. Annual discharges (GBq) to water from Swedish NPPs of the most abundant radionuclides*

	1995	1996	1997	1998	1999	2000
Ringhals 1						
^{51}Cr	4.01	4.60	5.90	3.60	1.97	14.7
^{54}Mn	83.4	2.54	9.90	2.20	2.35	66.4
^{58}Co	4.35	3.94	11.0	3.00	4.25	95.5
^{60}Co	37.9	25.3	110	30.0	14.2	8.60
^{134}Cs	2.75	32.5	42.0	95.0	14.8	304
^{137}Cs	5.63	1.73	5.30	10.0	2.26	42.1
Ringhals 2-4						
^{51}Cr	8.02	5.09	1.92	1.40	4.58	1.88
^{58}Co	39.2	19.2	19.7	20.5	33.4	12.0
^{60}Co	14.1	10.0	6.90	5.31	17.7	3.12
^{110m}Ag	2.73	3.04	2.20	5.36	2.80	1.15
^{124}Sb	1.91	1.61	12.4	4.39	1.59	3.18
^{125}Sb	1.26	2.24	1.69	96.0	94.6	1.45
Barsebäck 1-2						
^{51}Cr	10.8	13.0	25.0	16.0	5.30	45.0
^{54}Mn	5.66	60.0	5.20	2.40	1.50	1.50
^{58}Co	6.16	65.0	5.80	3.70	3.00	1.90
^{60}Co	29.7	52.0	20.0	9.20	15.0	18.0
^{124}Sb	32.9	38.0	49.0	1.30	25.0	20.0
^{137}Cs	3.72	74.0	83.0	1.00	61.0	1.20
Forsmark 1-3						
^{58}Co	7.62	4.12	7.60	94.0	1.20	45.3
^{60}Co	24.7	32.1	49.1	12.0	8.70	6.65
^{65}Zn	7.24	7.77	11.0	1.90	1.53	55.0
^{110m}Ag	1.30	2.71	8.20	1.30	61.9	21.2
^{134}Cs	1.68	7.71	13.0	2.40	2.75	1.73
^{137}Cs	3.72	8.49	15.0	3.72	6.35	4.66
Oskarshamn 1-3						
^{51}Cr	24.5	59.4	21.3	53.2	6.17	4.83
^{58}Co	9.52	8.83	2.51	3.94	2.08	97.1
^{60}Co	42.8	43.4	17.0	15.0	11.5	9.94
^{110m}Ag	12.0	1.42	1.52	1.45	90.7	88.3
^{125}Sb	3.85	3.51	1.19	1.02	1.32	34.6
^{137}Cs	4.92	2.41	2.38	1.12	89.7	67.5

* These releases are included in the summarized values given in the previous tables.

Chapter 3

Radioactive Contamination and Vulnerability of Arctic Ecosystems

3.1. Introduction

Monitoring the levels and trends of man-made radionuclides in Arctic environments is a central part of the AMAP programme. The first AMAP assessment presented several radionuclide time series for the Arctic. Although, some of these have been extended in the present assessment, the main emphasis has been on the provision of new information. Milk is a key foodstuff which was not specifically addressed in the monitoring and trend section of the first assessment. To compensate for this, a number of time trends in ^{137}Cs and ^{90}Sr activity concentrations in milk are described below.

Although well known, the presence of ^{99}Tc and ^{129}I in the Arctic from fallout and earlier Sellafield discharges was not addressed in the first AMAP assessment because they were considered of less radiological significance than other radionuclides. However, increased rates of discharge from European reprocessing plants have raised awareness of these radionuclides, especially technetium, and brought the issue into political focus. Also, new information on remobilization of sedimented radionuclides from earlier discharges has made ^{239,240}Pu activity concentrations more relevant for current monitoring activities.

Available data have improved in terms of their quantity, the range of variables monitored, and the length of time series. This has enabled analysis of vulnerability for Arctic pathways of radiation with respect to three criteria: spatial variation in transfer rates, spatial variation in ecological half-lives, and variation in contamination between species. Radioecological vulnerability, which is also referred to as radioecological sensitivity (Howard, 2000), is considered in terms of the dose to man (or concentration in environmental compartment, if dose to man is not applicable) following a unit radionuclide input to the environment. The estimation of integrated transfer coefficients (Bq/m^3 per kBq/m^2) in the first AMAP assessment is an example of such analysis.

Five site-specific studies are included here, covering Arctic areas that were inadequately dealt with in the first assessment or for which significantly improved data are now available.

3.2. Atmosphere

Measurements of activity concentrations in the atmosphere provide one of the best means of detecting recent releases of radionuclides. Measurements in the surface atmosphere have not detected any significant new releases of artificial radioactivity over the last six years; in the Russian Arctic, for example, low mean annual concentrations in the surface atmospheric layer, of $<0.1\times10^{-5}$ Bq/m^3 of ^{137}Cs and $<0.02\times10^{-5}$ Bq/m^3 of ^{90}Sr, have been measured. Recently reported data for $^{239/240}$Pu deposition in Arctic Finland after the Chernobyl accident are also low at <0.25 Bq/m^2 (Paatero *et al.*, 2002).

The first AMAP assessment contained few air measurement data for Arctic areas in the early-1960s; the period of most intensive atmospheric nuclear weapons tests. Atmospheric ^{137}Cs activity concentrations in Finland are now available for a 40-year period (Figure 3·1). This dataset is based on measurements obtained at six sites and for different time periods. Because there is little spatial variation in precipitation across Finland, the data for the two southerly sites (Nurmijärvi and Seutula) from 1971 until the Chernobyl accident in 1986 are typical of Arctic conditions. The post-1995 data indicate

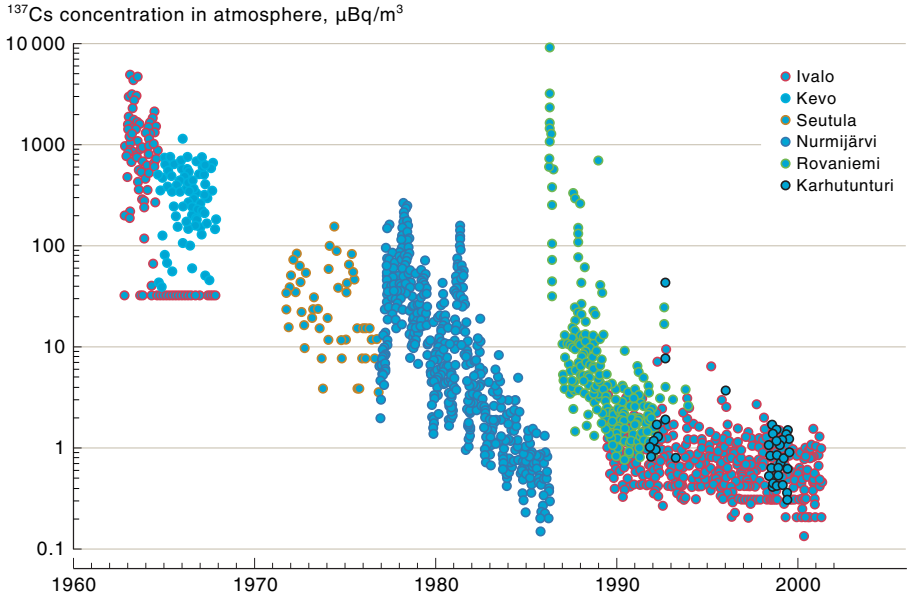

Figure 3·1. Atmospheric activity concentrations of ^{137}Cs at several sites in Finland.

lower levels in winter, due to snow cover, and higher levels when snow is not present, due to resuspension. The newly available data indicate higher values during the early-1960s than previously reported by AMAP for this period. Measurements at Ivalo during the peak period of global fallout are highly variable (Aaltonen *et al.*, 2002a).

Some new measurements from summer 1986 have also been made available since the first assessment (Aaltonen *et al.*, 2002b). After the Chernobyl accident, atmospheric ^{137}Cs activity concentrations in Arctic Finland were comparable to the peak values recorded in the 1960s, but the spatial variability was much greater.

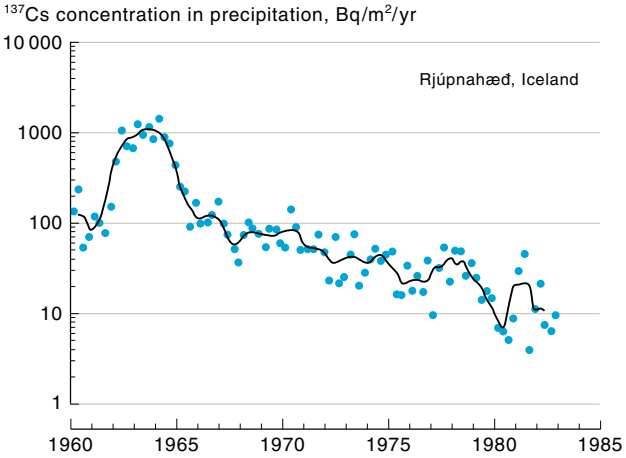

Figure 3·2. Activity concentrations of ^{137}Cs in precipitation at Rjúpnahæð (near Reykjavík). The points are quarterly measurements and the line a four-period moving average.

For Iceland, ^{137}Cs activity concentrations in precipitation at Rjúpnahæð (near Reykjavík) during the global fallout period are shown in Figure 3·2. Although the relevant time period is not shown on this graph, there was no detectable Chernobyl fallout in Iceland.

3.3. Marine environment

3.3.1. Technetium-99

Discharges from Sellafield in the 1970s and global fallout were mainly responsible for the initial occurrence of ^{99}Tc in Arctic seas. The recent increases in discharges from Sellafield are now the major source. Activity concentrations of around 20 to 25 Bq/kg dw were measured in the seaweed *Fucus vesicolosus* between 1994 and 1996 along the northern Norwegian coast (Yiou *et al.*, 2002). Using 1×10^5 as a concentration factor for the uptake, this corresponds to a seawater activity concentration of approximately 0.02 Bq/m^3.

A time series of ^{99}Tc activity concentrations in seawater at Hillesøy, a coastal location in northern Norway, showed an increase from 0.46 Bq/m^3 in summer 1997 to a maximum of 2.0 Bq/m^3 in early 2001. Following that peak, the levels steadily decreased to <1 Bq/m^3. For *Fucus vesiculosus*, activity concentrations increased from around 50 Bq/kg dw in July 1997 to >400 Bq/kg dw in January 2001. Concentrations then appeared to level off during 2001 and decrease throughout 2002, although more observations will be needed to confirm this trend (Figure 3·3). A comparison of ^{99}Tc activity concentrations in *Fucus* and seawater from this site indicates a concentration factor from water to *Fucus* in the

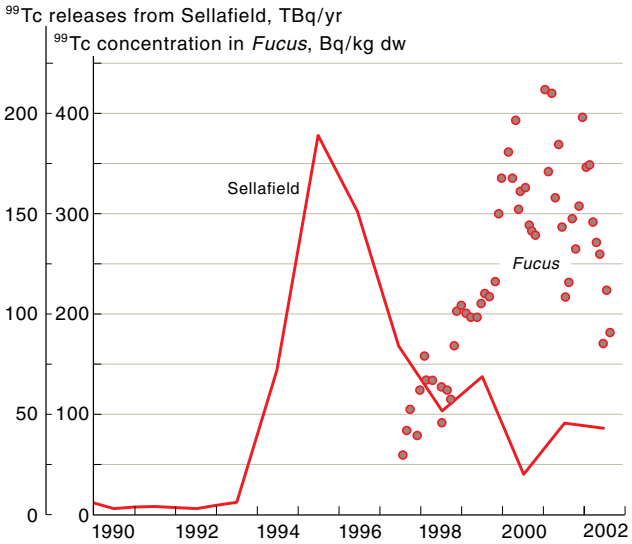

Figure 3·3. Temporal variation in ^{99}Tc activity concentrations in *Fucus* at Hillesøy (northern Norway) and releases from the Sellafield reprocessing plant (Kolstad and Lind, 2002).

range 1×10^5 to 2.6×10^5, as equilibrium between ^{99}Tc in the water and seaweed appears to occur slowly (Kolstad and Lind, 2002).

In 1999, ^{99}Tc activity concentrations observed along the Norwegian coast were associated with the peak releases from Sellafield in 1995, whereas those observed in Greenlandic and Icelandic waters were mainly due to releases from Sellafield and La Hague throughout the 1970s and 1980s. Global fallout levels are at least an order of magnitude lower than discharges from Sellafield (Dahlgaard, 1994; Dahlgaard *et al.*, 1995).

Karcher *et al.* (2003) simulated the ^{99}Tc distribution arising from Sellafield discharges between 1990 and 1999. This was done using a three-dimensional coupled ice-ocean model forced with daily variable atmospheric data (Karcher *et al.*, 2002). The ^{99}Tc was assumed to have been released from January 1990 onwards, with actual Sellafield discharge data input to the model at the North Channel in the northern Irish Sea. Figure 3·4 compares the model results for June 1999 with measured ^{99}Tc activity concentrations in the Northeast Atlantic in 1999. Predicted activity concentrations in the southern Barents Sea are in reasonable agreement with measured values. Values measured around Greenland, Iceland, and the Faroe Islands are mainly influenced by pre-1990 discharges and are not represented by the model.

3.3.2. Iodine-129

Levels of the naturally occurring long-lived iodine isotope ^{129}I have been elevated substantially over the past decades, initially due to atmospheric nuclear weapons tests and subsequently due to emissions from nuclear fuel reprocessing (see Figure 2·3). Atmospheric weapons tests resulted in an increase in the ratio between radioactive ^{129}I and its stable counterpart, ^{127}I, in the surficial compartments (i.e., surface soils, surface seawater, and freshwaters) of the Northern Hemisphere from 10^{-12} to 10^{-10}.

The iodine isotope ratio (and other radionuclide ratios such as ^{129}I:^{137}Cs and ^{129}I:^{99}Tc) can be used to trace the movement of water masses from the Norwegian Coastal Current into the Arctic basin and the At-

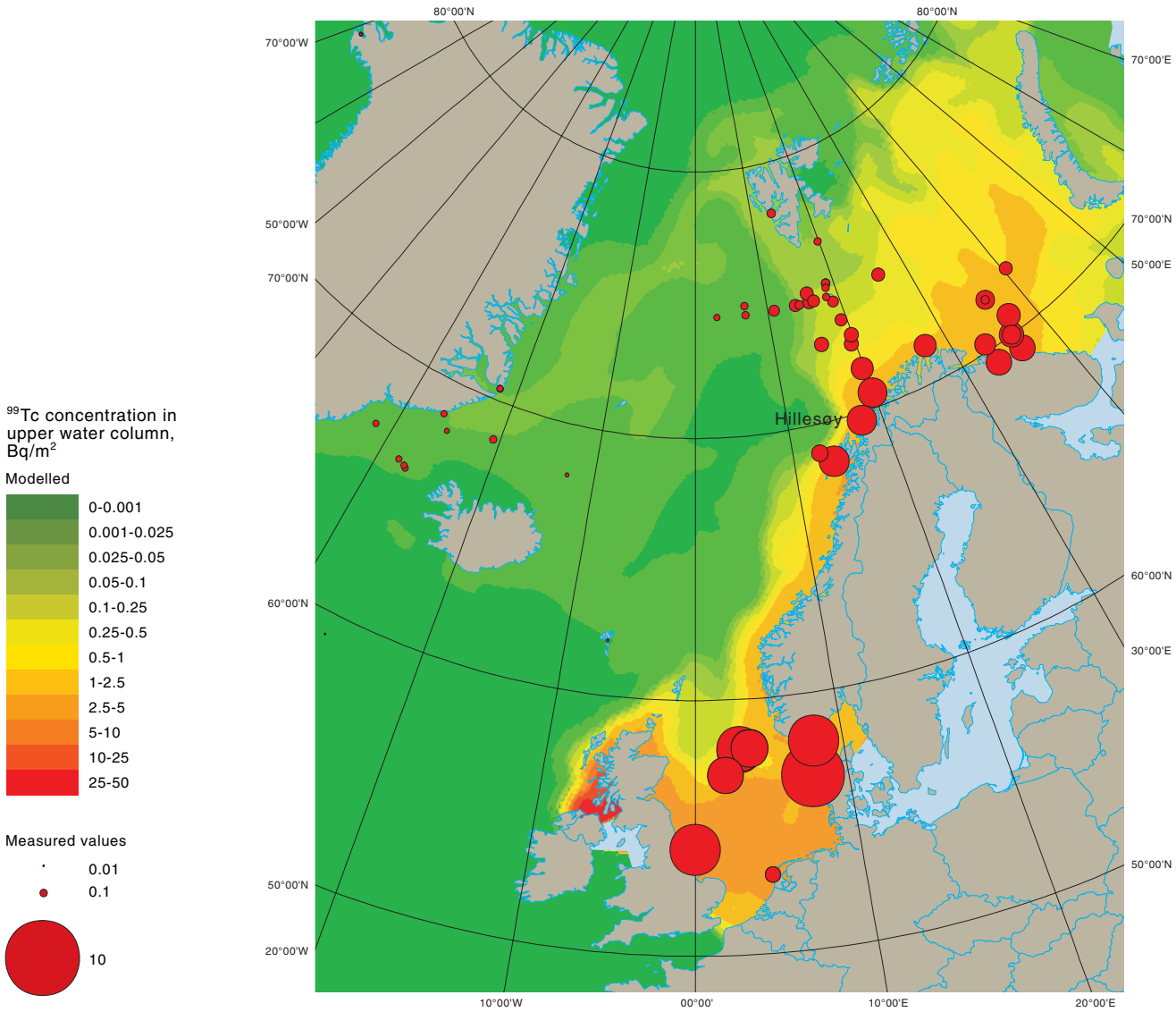

Figure 3·4. Simulated distribution of ^{99}Tc concentrations in the upper 20 m of the water column in June 1999 (Karcher *et al.*, 2003). Concentration is represented by the color coding. The red circles show measured values in 1999.

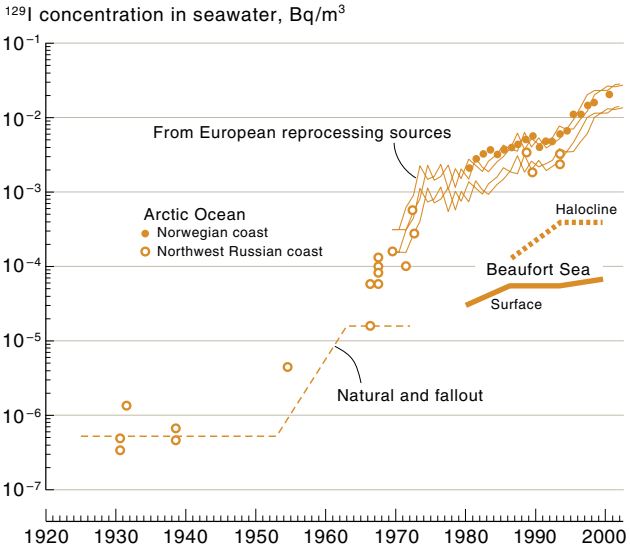

Figure 3·5. Activity concentrations of ^{129}I in the Arctic Ocean near the entry of Atlantic Water (Norway and northwestern Russia) and the Beaufort Sea (Cooper *et al.*, 1998, 1999; Matishov *et al.*, 2002; Raisbeck and Yiou, 1999; Smith *et al.*, 1998, 1999; Yiou *et al.*, 2002).

lantic Deep Water. Along the northern Norwegian and northwest Russian coasts, ^{129}I levels, and hence the isotope ratio, have been orders of magnitude higher than the global fallout level since the late-1970s. Measurements taken one month after the *Kursk* submarine accident in August 2000 did not indicate any leakage of ^{129}I from the *Kursk*, but confirmed the high levels and increasing trend of ^{129}I activity concentrations in Atlantic water entering the Arctic Ocean (Figure 3·5). The activity concentration of ^{129}I from European reprocessing sources in water entering the Arctic Ocean may be predicted using dilution factors of 1 to 2×10^{-14} yr/m^3 and transit times of 2 to 3 yr (Cap de la Hague) and 4 to 5 yr (Sellafield) (Figure 3·5). This is in agreement with earlier studies on other radionuclides (e.g., AMAP, 1998; Dahlgaard, 1994). In the Beaufort Sea, levels in the halocline water masses have increased as a result of ^{129}I transported from the Atlantic Ocean through the Arctic Ocean; whereas in surface waters it is mainly fallout-derived ^{129}I of Pacific origin that has been detected. The transport time for the Atlantic halocline waters across the Arctic Basin is in the order of one to two decades (AMAP, 1998; Smith *et al.*, 1998, 1999).

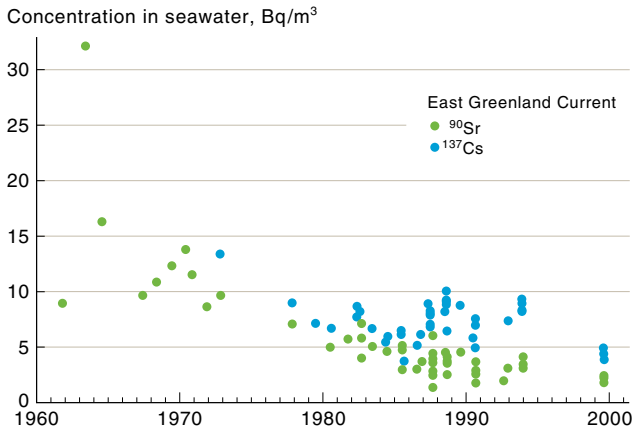

Figure 3·6. Time trends for ^{137}Cs and ^{90}Sr in the East Greenland Current (between 10.7°E and 22.35°E, and 70°N and 81.9°N).

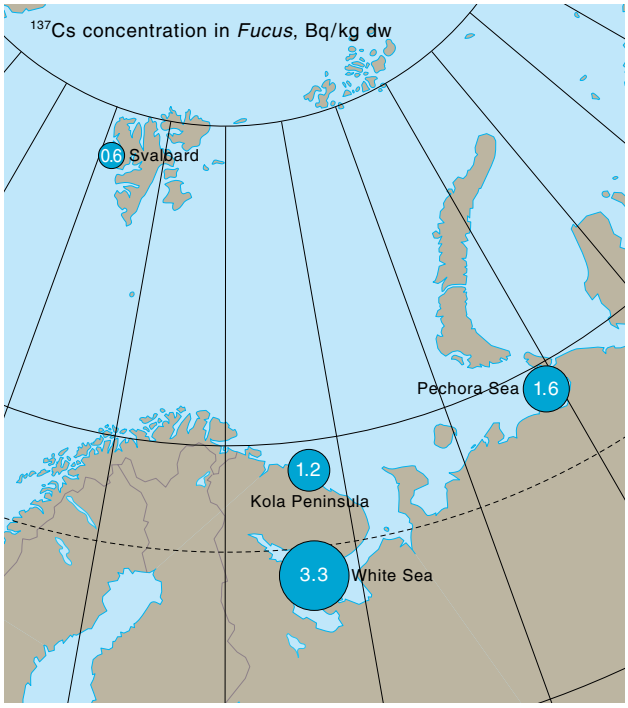

Figure 3·7. Activity concentrations of ^{137}Cs in *Fucus* (Rissanen *et al.*, 1995).

3.3.3. Cesium-137 and ^{90}Sr

Changes with time in ^{137}Cs and ^{90}Sr activity concentrations in surface seawater in the East Greenland Current (where the highest levels generally occur around Greenland) are shown in Figure 3·6. Whereas ^{90}Sr activity concentrations in seawater decrease with an effective ecological half-life (T_{eff}; Box 3·1) of approximately nine years, those of ^{137}Cs level off and even increase during the late-1980s and early-1990s due to fallout from the Chernobyl accident and inputs from European reprocessing plants.

Recent measurements of ^{137}Cs activity concentrations in *Fucus* spp. in the Barents, Pechora, and White Seas are shown in Figure 3·7. The highest values were detected in the White Sea.

Recent data (1993 to 1998) on the spatial distribution of ^{137}Cs in sediments from Arctic areas indicate considerable variability (Figure 3·8). Activity concentrations in areas with no known local sources of anthropogenic radionuclides are <20 Bq/kg dw. Higher levels occur in the White Sea and off the Norwegian coast (up to 60 Bq/kg dw) in areas strongly affected by Chernobyl fallout, in the outer parts of the Yenisey estuary (up to 80 Bq/kg dw), near the Atomflot base in the Kola fjord (up to 200 Bq/kg dw) and in the dumping areas along the western coast of Novaya Zemlya (up to 10^5 Bq/kg dw).

Consistently low ^{137}Cs activity concentrations have been reported in a range of marine fish species (Table 3·1). Except for dab (*Limanda limanda*) and capelin (*Mallotus villosus*), no significant differences were found between the various species. Dab (with low levels) were

Table 3·1. Activity concentrations for ^{137}Cs in marine fish species (Bq/kg ww) for 1995 to 2000 (AMAP Data Centre).

	n	Mean ± SD
Haddock (*Melanogrammus aeglefinus*)	65	0.25 ± 0.11
Cod (*Gadus* spp.)	394	0.22 ± 0.08
Shorthorn sculpin (*Myoxocephalus scorpius*)	10	0.31 ± 0.16
Flounder (*Platichthys flesus*)	6	0.33 ± 0.06
Capelin (*Mallotus villosus*)	3	0.16 ± 0.08
Dab (*Limanda limanda*)	247	0.09 ± 0.03

Box 3·1. *Effective ecological half-lives*

The effective ecological half-life (T_{eff}) describes the time required for the activity concentration of a radionuclide in an environmental compartment (often a food product) to be reduced to one half of its original activity concentration. It therefore incorporates physical decay. The ecological half-life (T_{ec}) does not take physical decay into account, and thus can be adapted for different isotopes of the same element. For example, the T_{eff} of ^{134}Cs and ^{137}Cs will differ because of the differences in physical half-lives, while the T_{ec} would be identical. The relationship between T_{eff} and T_{ec} for a radionuclide with a physical half-life (T_{phys}) will be:

$$T_{eff} = (T_{phys} \cdot 1/T_{ec})/(T_{phys} + 1/T_{ec})$$

Where data on long time trends are available, it is possible to model changes with time in activity concentrations. As a given radionuclide often exists in the environment in different physical/chemical forms, with different mobilities, a multiple exponential function is needed to describe the changes. In practice, a double exponential is often used, of the form:

$$A(t) = A_0 \cdot \left(a \cdot \exp\left(-\frac{\ln 2}{T_1} \cdot t\right) + (1-a) \cdot \exp\left(-\frac{\ln 2}{T_2} \cdot t\right)\right) \qquad \text{Eqn. 3.1}$$

where $A(t)$ is the activity concentration at time t, A_0 is the initial activity, T_1 and T_2 are the effective ecological half-lives (T_{eff1} and T_{eff2}) or the ecological half-lives (T_{ec1} and T_{ec2}); and a is a parameter partitioning the decay between the two half-lives. In this case, the shorter half-life will dominate the decay for the first period, and the longer half-life will govern the process on a longer time scale.

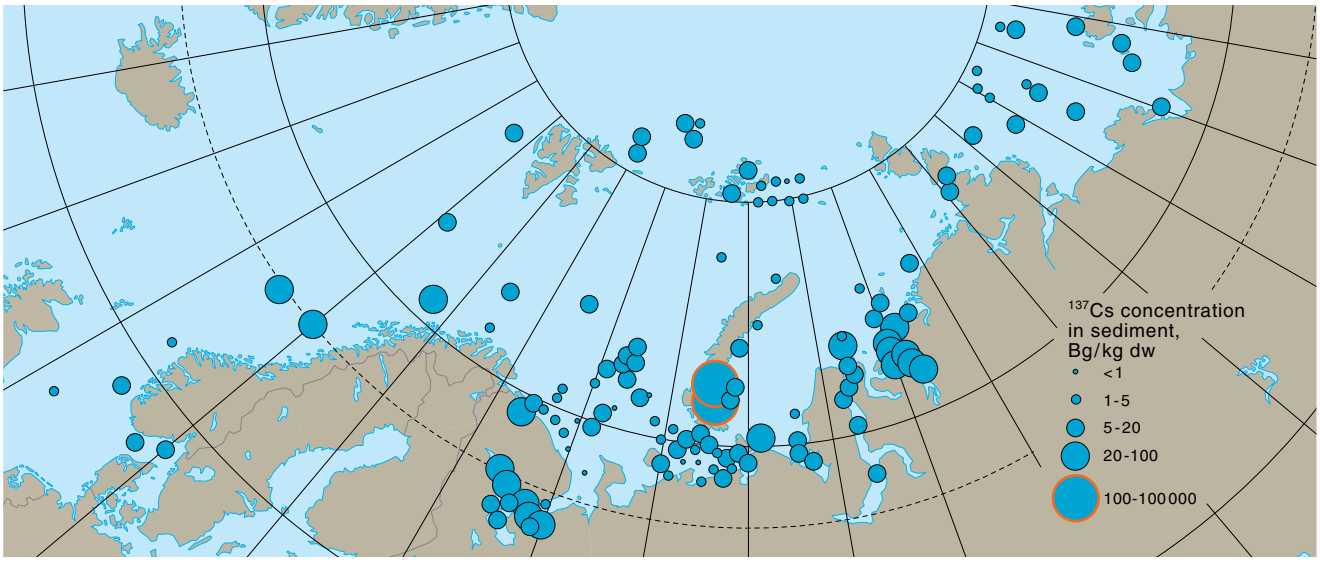

Figure 3·8. Distribution of ^{137}Cs in surface sediments from 1993 to 1998.

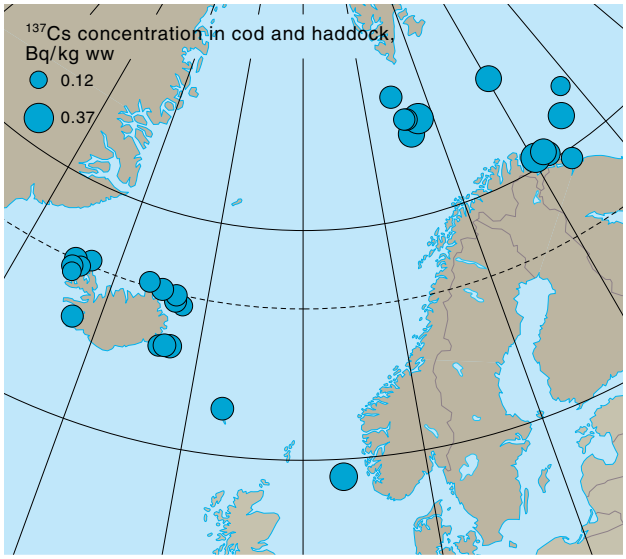

Figure 3·9. Average ^{137}Cs activity concentrations in cod and haddock from 1995 to 2000.

sampled near Iceland, where ^{137}Cs levels in seawater are generally low. Figure 3·9 shows the spatial variation in mean ^{137}Cs activity concentrations in cod (*Gadus* spp.) and haddock (*Melanogrammus aeglefinus*) from 1995 to 2000.

3.3.4. Plutonium isotopes

The ^{239,240}Pu activity concentration in surface waters of northern seas ranged from 2 to 66 mBq/m^3 in 1995 (Figure 3·10). The highest ^{239,240}Pu activity concentrations occurred in surface seawater off the north and northeast coasts of Scotland. This is consistent with findings on remobilized plutonium (Pu) from Irish Sea sediments as a newly identified source of Pu, with a typical ^{238}Pu : ^{239,240}Pu isotope ratio of around 0.2 (see Section 2.2.3).

The ^{238}Pu : ^{239,240}Pu isotope ratios in waters, particularly in the Norwegian Sea and Barents Sea, are elevated above the expected fallout ratio of ~0.04 and are, in some cases, close to Irish Sea ratios. The isotope ratios therefore suggest that Sellafield-derived Pu may have

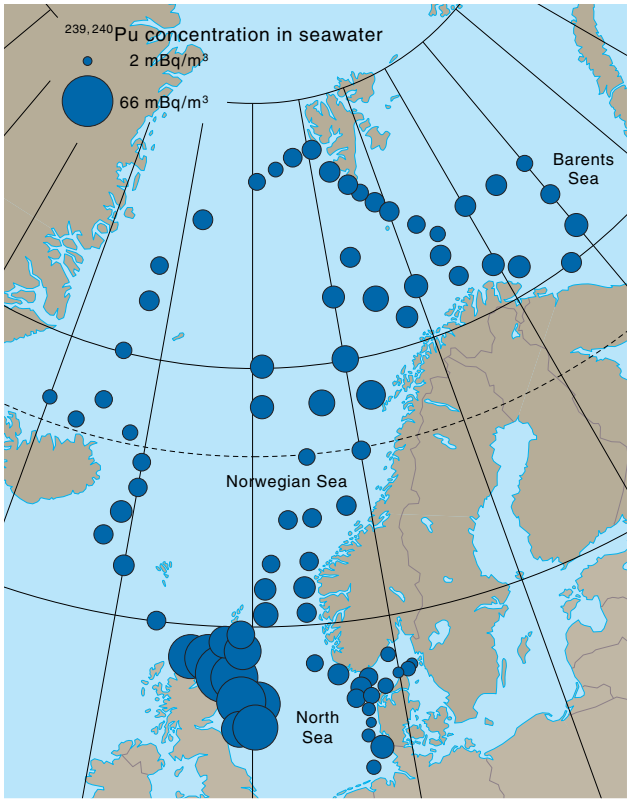

Figure 3·10. Distribution of ^{239,240}Pu in surface seawater of the northern seas in 1995 (Grøttheim, 2000).

been transported with the currents into the Norwegian Sea and the Barents Sea and, to a lesser extent, even the Greenland Sea. The ratios indicate that Sellafield could have been a major contributor to the Pu concentrations observed in Scottish and Norwegian waters (Grøttheim, 2000; Herrmann *et al.*, 1998; Kershaw *et al.*, 1999), whereas the primary source in the Arctic Ocean and the Greenland Sea remains global fallout. However, in two surveys of Pu in seawater in various parts of the Arctic Ocean, Vintró *et al.* (2002) explained the observed concentrations by advection of global fallout Pu from mid-North-Atlantic latitudes to the Arctic Ocean and failed to see evidence of Sellafield Pu.

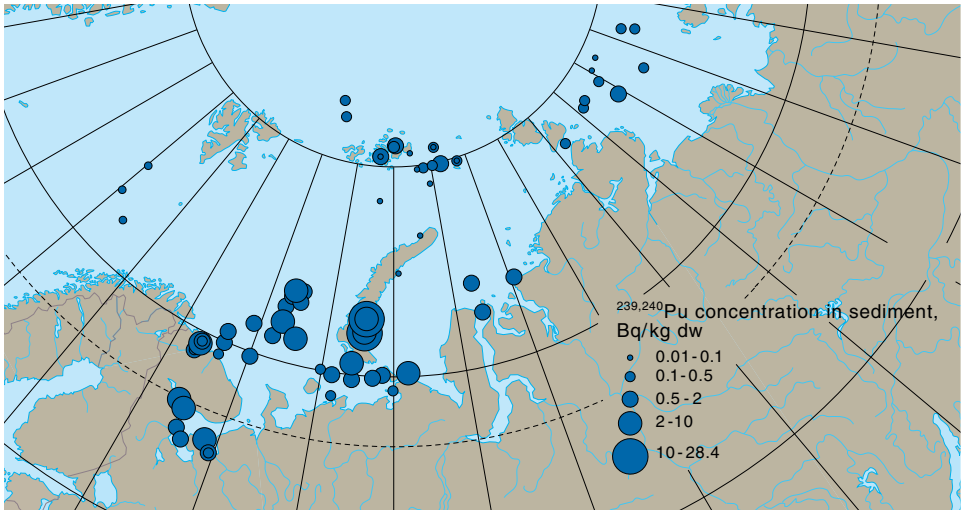

Figure 3·11. Activity concentrations of ^{239,240}Pu in surface sediments 1996 to 2000.

Figure 3·11 shows Pu activity concentrations in surface sediments. Global fallout levels in marine sediments depend on many factors such as sediment characteristics, depth, and proximity to river outflows. As is the case for seawater, isotope ratios may indicate different origins, but the actual global fallout concentrations in surface sediments vary considerably. Cooper *et al.* (2000) found Pu isotope ratios in Arctic Ocean sediments to be due to global fallout in most cases, and failed to show any indications of a Sellafield contribution, in accordance with the results for seawater (Vintró *et al.*, 2002).

Rissanen *et al.* (2000) analyzed 92 fish samples from commercial operations in the Barents Sea and found ^{239,240}Pu in one sample only: a ray, *Raja radiata*, containing 7.9 and 4.9 mBq/kg ww in flesh and bones, respectively. Plutonium activity concentrations were below detection limits of 1 to 3 mBq/kg ww in all other samples.

3.3.5. Radionuclide behavior in marine systems

3.3.5.1. Partitioning and uptake

Water movement and sedimentation are of major importance in determining transport pathways and the fate of radioactive material released to, or transported within, the marine environment. One of the important geochemical factors affecting the transport of radionuclides is particle-water exchange. In water, radionuclides are partitioned between the dissolved and particulate phases. Partitioning depends on the chemical form of the radionuclide, the physical–chemical properties of the chemical analogues (usually elemental analogues), and the characteristics of the environment.

Biological uptake and associated dose assessments for human exposure generally depend on the levels of dissolved radionuclides in water, although filter feeders (mussels and oyster) accumulate radionuclides from the particulate phase. For biota that are closely associated with marine sediments, such as benthic infauna and epifauna, sediment concentrations can exert the primary influence on the extent of uptake.

3.3.5.2. Transport of radionuclides in sea ice

Sea ice transport is a unique pathway in polar areas (Pfirman *et al.*, 1995, 1997; Strand *et al.*, 1996) that is partially independent of water mass movement (Pfirman *et al.*, 1997). During sea ice formation, dissolved contaminants are rejected together with salt. Levels of dissolved contaminants are therefore lower in sea ice than in seawater (Weeks, 1994). Consequently, the main focus in ice transport studies has been on the transport and fate of radionuclides associated with sediment particles incorporated into the ice, derived both from particles suspended in seawater and from bottom sediments.

A radiological assessment of sea ice transport has been considered by Iosjpe and Borghuis (2000) and Iosjpe (2002). Their approach is based on box modelling (Iosjpe *et al.*, 1997, 2002) that incorporates the various transfers of radioactivity: from the liquid phase to ice; from suspended sediment and bottom sediment to ice; through sea ice transport between sea areas; and into seawater during ice melt.

The potential significance of sea ice transport in the dispersion of radionuclides within the marine environment was illustrated by modelling the transport of radionuclides from the Kara Sea to the Fram Strait through the Arctic Ocean following the simulated release of 1 TBq of specific radionuclides into Ob Bay in the Kara Sea. Dispersion through the Kara Sea to the Fram Strait is particularly interesting because the model indicated that sea ice transport of contaminants in this region may represent a more rapid transport pathway than water. Furthermore, modelling showed the effect to be directly proportional to the partition coefficient (Kd) of each radionuclide. Thus, sea ice transport seems more relevant for high-Kd elements such as ^{241}Am and ^{60}Co and less important for ^{137}Cs and ^{90}Sr which are principally associated with the dissolved phase (Iosjpe and Strand, 2002).

3.3.6. Vulnerability in marine pathways

Marine ecosystems are relatively less vulnerable to atmospheric inputs of radiocesium than freshwater and terrestrial environments. This is due to the capacity of most marine ecosystems to rapidly dilute an input of radioactive contaminants through processes such as advection and mixing, coupled with the large volumes of water generally involved, and the high ionic strength of the saline waters. Thus, short-term consequences are likely to be more important in marine ecosystems as dilution is a long-term process. Vulnerable marine ecosys-

tems include those into which liquid discharges are released – especially if their exchange with the World Ocean is slow or restricted, as is the case for some fjords and inlets. In an Arctic context, this could include areas that receive radionuclides transported from nuclear facilities by marine currents or rivers such as the Ob and Yenisey. In addition to man-made radionuclides, marine areas may receive natural radioactivity from non-nuclear industries.

Radioecological vulnerability in marine ecosystems is affected by a number of factors, such as water exchange rates; residence times for radionuclides in the water column; sediment and sedimentation properties, including bioturbation and resuspension; freshwater inflow; salinity; oxygenation of the water column and sediments; and ice conditions.

Marine zones with high biological productivity are considered radioecologically vulnerable, when considering collective doses. From an economic point of view, highly productive areas such as the Barents Sea and areas used for aquaculture of mollusks, fish, or crustaceans are potentially important.

3.4. Freshwater environment

Estimates of the initial activity concentration in water bodies following radionuclide deposition can be made by assuming dilution of activity 'deposited' to the river or lake surface. Therefore, deep rivers and lakes would be expected, initially, to be less vulnerable than shallow water bodies. However, deposition times can be long, as was the case for global fallout, compared to river water transit times. Catchment runoff can also make a significant long-term contribution to water activity concentrations. Activity concentrations in runoff water decline significantly with time after deposition. At a given point in time, the activity concentration in river or lake water per unit of deposition to the catchment (the *runoff coefficient*) is a measure of radioecological vulnerability of the catchment. Organic, boggy catchments, such as those prevailing in some Arctic areas, have much higher ^{137}Cs runoff coefficients than catchments with a high proportion of mineral soils (Kudelsky *et al.*, 1996).

Loss of the initial input of radioactivity from lake and reservoir water may be estimated using the water residence time of the lake and simple models for the removal of radioactivity to sediments. Long-term activity concentrations in lakes with relatively short water residence times are primarily controlled by inputs of radioactivity from the surrounding catchment. Long-term activity concentrations in closed lakes, which have relatively long water residence times, are controlled by the transfer of radioactivity to and from bottom sediments. Lakes with high vulnerability for radiocesium include shallow lakes with long water residence times, especially those having large catchments with a high percentage of organic boggy soils, as these allow higher runoff than catchments with other soil types. In areas such as Finnish Lapland, vulnerability to radiocesium is high due to the presence of boggy soils (30 to 60% of catchments) combined with a hilly topography that may increase surface runoff.

When contaminated snow melts, radionuclides may be transported with runoff waters and may then contaminate soils and freshwater systems. Contamination from snow melt can be highly heterogeneous and radionuclide contents in soil can be higher in areas where water from melting snow accumulates. The effect varies with the size of the catchments and lakes. Ice overlying lakes can, to some extent, protect freshwater biota from radioactive contamination. However, when the ice melts there is a sudden influx of contamination, in addition to that from melting snow. Thus, there can be a delayed pulse of enhanced activity concentrations in spring.

Currently, atmospheric and surface terrestrial activity concentrations of ^{129}I in the Northern Hemisphere exceed those due to global fallout by up to six orders of magnitude near point sources (Sellafield, Cap de la Hague, Chernobyl), three to four orders of magnitude in most of Western Europe, and at least an order of magnitude outside Europe (e.g., Moran *et al.*, 1999). In the Arctic, little is known about terrestrial ^{129}I levels, but some indications based on analyses of freshwaters and precipitation suggest an increase of about two orders of magnitude above the nuclear weapons test (NWT) contaminated background levels (Moran *et al.*, 1999), and four orders of magnitude above the natural background level in most of the Arctic. For freshwaters, where current ^{129}I:^{127}I ratios are typically up to an order of magnitude lower than in terrestrial food chains within the same region, very few data are available for rivers and estuaries without evident contamination (Beasley *et al.*, 1997; Fehn and Snyder, 2000; Meili *et al.*, 2002). Levels are far higher in some Russian rivers into which discharges from point sources, such as the Mayak reprocessing plant, occur, reaching about three and four orders of magnitude above the NWT background level near the mouths of the Yenisey and Ob rivers, respectively (Cochran *et al.*, 2000; Cooper *et al.*, 1998).

3.4.1. Rivers

The first AMAP assessment reported temporal changes in ^{137}Cs and ^{90}Sr activity concentrations in Finnish and Russian rivers. Figure 3·12 shows the extended time series for ^{90}Sr activity concentrations in the water of different Russian rivers. Current levels are low in all areas of the Russian Arctic.

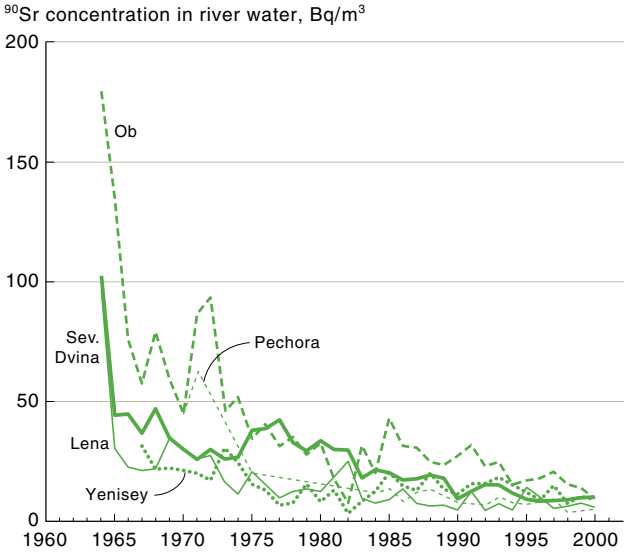

Figure 3·12. Changes in ^{90}Sr activity concentrations in Russian rivers since the mid-1960s.

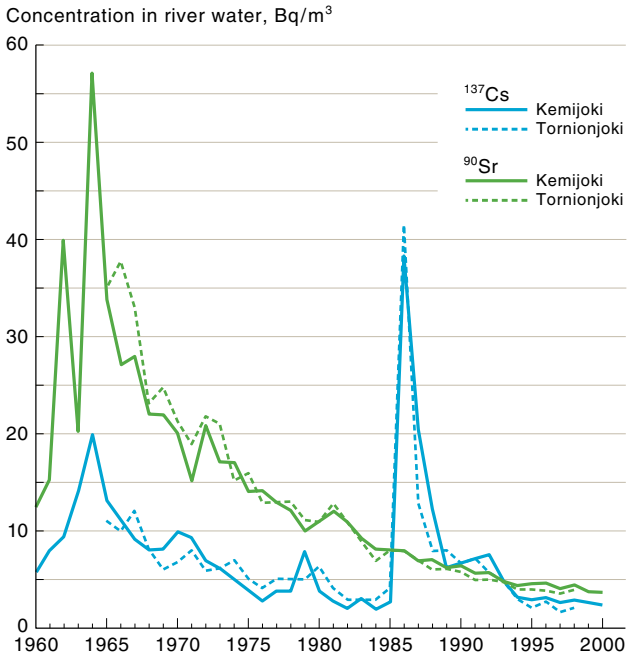

Figure 3·13. Annual mean levels of ^{137}Cs and ^{90}Sr in two Finnish rivers since 1960.

Updated information for two Finnish Arctic rivers (Figure 3·13) shows further declines over the last five years, following the Chernobyl accident (1986). In the global weapons testing period, ^{90}Sr activity concentrations in rivers were consistently two- to three-fold higher than for ^{137}Cs. Chernobyl deposition reversed this situation such that activity concentrations of ^{137}Cs were higher than ^{90}Sr for some years following deposition. However, owing to the shorter effective half-life (T_{eff}; Box 3·1) of ^{137}Cs compared with ^{90}Sr, activity concentrations of ^{90}Sr have been slightly higher than for ^{137}Cs since 1994.

Table 3·2 compares T_{eff} values for ^{90}Sr in various Russian rivers.

Table 3·2. T_{eff} values for ^{90}Sr in Russian rivers.

	T_{eff_1} (yr)	T_{eff_2} (yr)
Severnaya Dvina	12.9±1.7*	
Ob	0.7±0.3	13.2±2.4
Lena	0.2±0.15	14.3±2.2

Half-lives calculated using non-linear least squares (www.r-project.org).

* It was not possible to calculate a double exponential for Severnaya Dvina.

Data for ^{90}Sr and ^{137}Cs in two Finnish rivers (Saxen, 2003) show that post-Chernobyl T_{eff} values for ^{137}Cs are about a factor of two shorter than the bomb-fallout half-lives (Table 3·3). As different periods of time were used to calculate the half-lives in Tables 3·2 and 3·3, the first components of the T_{eff} for the Finnish and Russian rivers are not comparable, but the second components are not significantly different.

3.4.2. Fish

Long-term data for Russia which enable a comparison of ^{137}Cs activity concentrations in freshwater fish from two Arctic regions indicate that concentrations in fish from the Kola Peninsula are statistically significantly

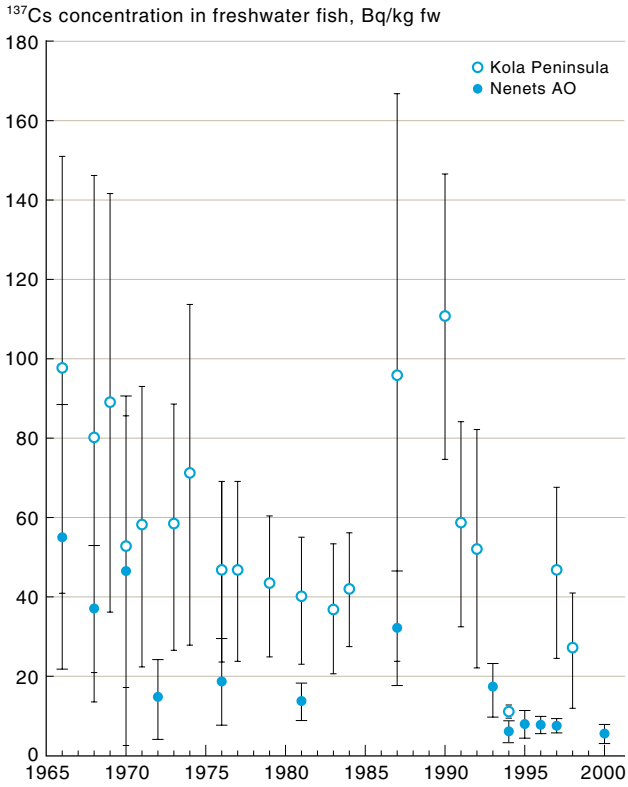

Figure 3·14. Activity concentrations (±SD) for ^{137}Cs in freshwater fish from the Kola Peninsula and Nenets AO.

higher than in fish from lakes and rivers in the Nenets Autonomous Okrug (NAO), both after the period of global fallout and after the deposition of Chernobyl fallout (Figure 3·14). A contributory factor was the higher Chernobyl fallout on the Kola Peninsula than in the NAO.

Table 3·3. T_{eff} values (yr) for ^{90}Sr and ^{137}Cs in Finnish rivers from weapons test and Chernobyl fallout (based on Saxen, 2003).

		Weapons tests	Chernobyl	
		1965-1985	1988-2000	1965-2000
^{90}Sr	Tornionjoki	9.1	11.6	9.5
	Kemijoki	10.5	12.8	11.6
		1965-1985	1986-1988	1989-2001
^{137}Cs	Tornionjoki	9.3	0.6	4.2
	Kemijoki	10.4	1.0	5.1

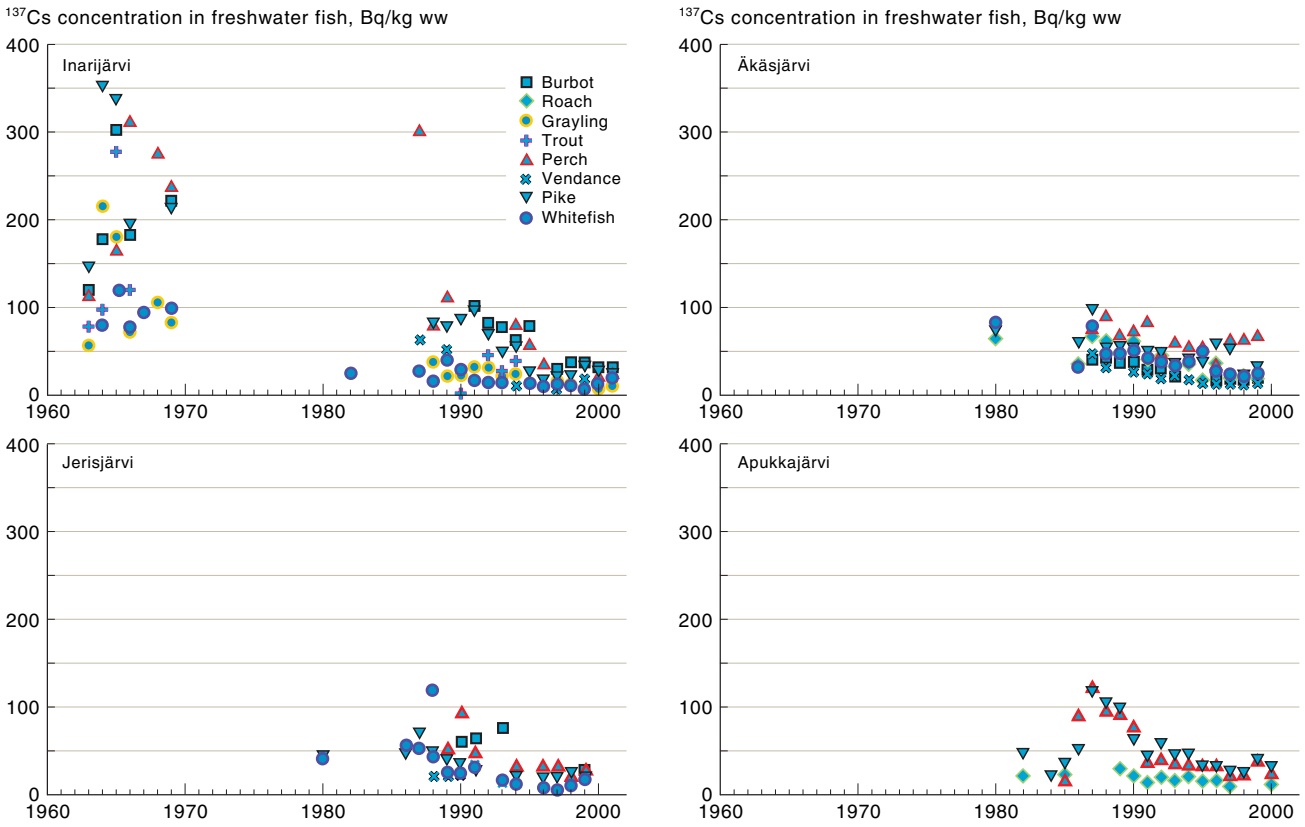

Figure 3·15. Changes in ^{137}Cs activity concentrations with time in various fish species in four lakes in Arctic Finland.

Also, fish from the Kola Peninsula are mainly caught in lakes (particularly, Lake Lovozero), whereas they are mainly from rivers in the NAO, and fish caught in lakes are generally more contaminated than fish from rivers.

3.4.2.1. Species differences

Activity concentrations are available for ^{137}Cs in a large number of different fish species for four Finnish lakes: Inarijärvi, a large lake in northeast Finland (Pasvik area, Barents Sea catchment); Apukkajärvi, a small highly eutrophic lake in the vicinity of Rovaniemi (Kemijoki catchment area); and Äkäsjärvi and Jerisjärvi, which are two medium-sized lakes in the Tornionjoki system 200 km northwest of Rovaniemi (Figure 3·15). In Inarijärvi, the only lake sampled in Arctic Finland during the global fallout period, ^{137}Cs activity concentrations of up to 356 Bq/kg ww were measured in pike in 1964 (Kolehmainen *et al.*, 1966). In 1982, activity concentrations of 25 Bq/kg ww occurred in whitefish from Inarijärvi (STUK, 1983). After the Chernobyl accident, when more data became available, there was a single high measurement of 305 Bq/kg ww in perch, but otherwise ^{137}Cs activity concentrations were much lower than in the 1960s with maximum values of ~100 Bq/kg ww.

Radiocesium activity concentrations in fish are inversely related to the potassium (K) concentration of the surrounding water (e.g., Blaylock, 1982; Kolehmainen *et al.*, 1967). Similarly, an inverse relationship has been found between ^{90}Sr activity concentrations in fish and water calcium (Ca) concentrations (e.g., Blaylock, 1982). High K or Ca concentrations in water are often a result of the runoff of agricultural fertilizers, but this is not particularly relevant to Arctic ecosystems. Transfer rates to fish also depend on feeding habit, with ^{137}Cs activity concentrations in the more radioecologically-vulnerable predatory fish generally a factor of two or more higher than for non-predatory fish. The Finnish Arctic data are diverse and thus difficult to analyze, however, it is clear that some species generally contain lower radionuclide activity concentrations than others. For each lake sampled, ^{137}Cs activity concentrations are higher in pike (a predatory species) and perch (partially predatory) than in other species, and whitefish contamination is consistently low.

3.4.2.2. Migratory fish

Salmon (*Salmo salar*) from Arctic seas migrate up the Tana River, which flows from northern Finland, through Norway to the Barents Sea, to spawn. They are an important traditional food resource for the indigenous peoples living near the river and the present total catch varies from 90 to 180 t/yr. Annual samples show the activity concentrations of ^{137}Cs in flesh to have decreased from 1 Bq/kg ww in 1988 to 0.3 Bq/kg ww in 2000, with T_{eff} values of 6 yr over the study period. In comparison, Baltic salmon from the more heavily Chernobyl-contaminated Bothnian Bay had activity concentrations 100 times higher (Rissanen and Ikäheimonen, 2000).

3.5. Terrestrial environment

3.5.1. Soil and humus

Recent data demonstrate that ^{137}Cs deposition from 1995 to 2002 at the Zapolyarie and North monitoring stations, covering the Asian and European parts of the Russian Arctic respectively, has not exceeded 1.3 Bq/m^2/yr.

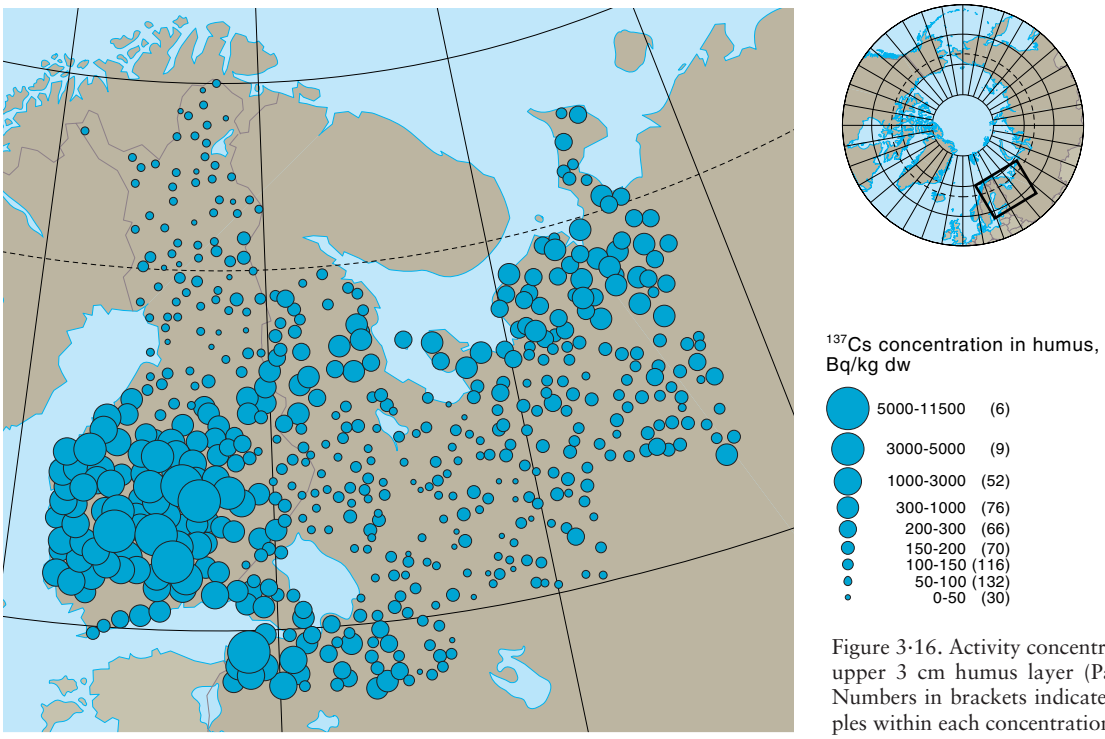

Figure 3·16. Activity concentrations of ^{137}Cs in the upper 3 cm humus layer (Paatero *et al.*, 2002). Numbers in brackets indicate the number of samples within each concentration range.

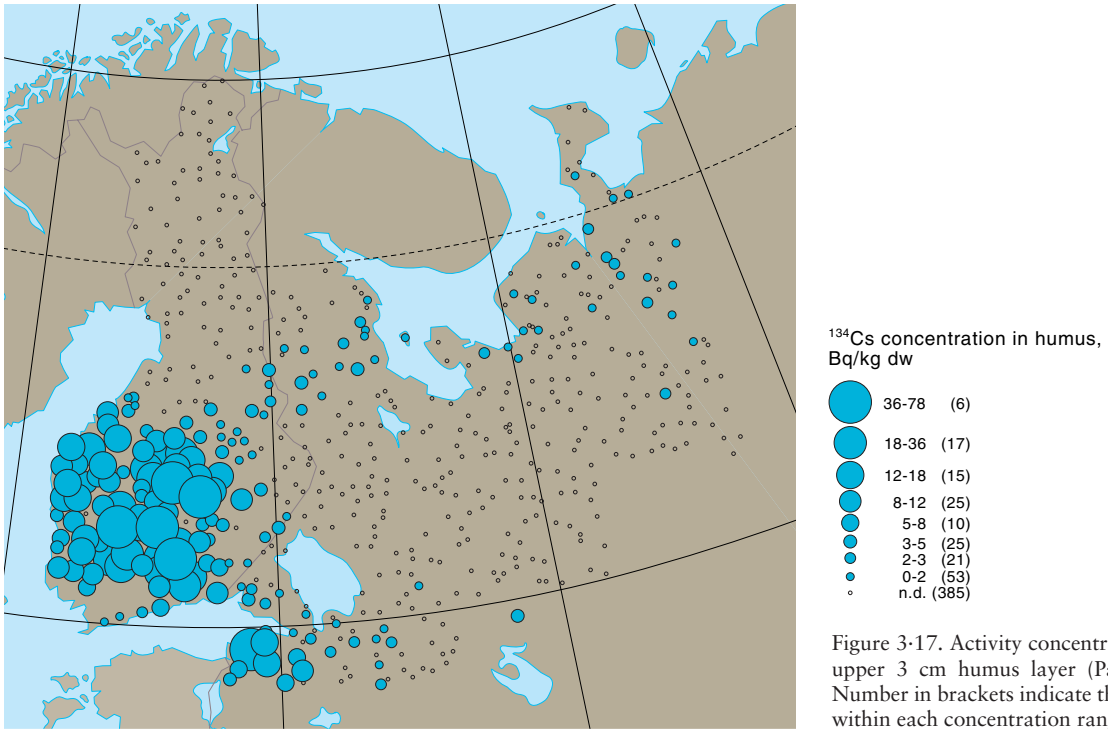

Figure 3·17. Activity concentrations of ^{134}Cs in the upper 3 cm humus layer (Paatero *et al.*, 2002). Number in brackets indicate the number of samples within each concentration range.

A comprehensive survey of ^{137}Cs and ^{134}Cs in the upper 3 cm humus layer in Finland and northwest Russia in 2000 (Figures 3·16 and 3·17) found ^{134}Cs (from the first plume) from the Chernobyl accident could still be detected in southern Finland and near St Petersburg. The presence of ^{134}Cs suggests that the elevated ^{137}Cs levels in this area are also due to the Chernobyl accident. Paatero *et al.* (2002) found a similar pattern to that of the humus survey for Chernobyl-derived Pu fallout in Finland.

Activity concentrations of 9 to 32 Bq/m^2 were found for ^{239,240}Pu in surface vegetation and in the upper 3 to 5 cm of soils near the coast in northwest Russia and Svalbard between 1993 and 1996. The variation was due to the type and density of the surface vegetation (Rissanen *et al.*, 2001). The ^{238}Pu : ^{239,240}Pu ratio at these sites and in Franz Josef Land suggests the primary source is global fallout.

The time of deposition is critical for Arctic ecosystems, particularly for short-term deposition, such as might occur after an accident. In subarctic areas, snow cover is present for about seven to eight months of the year, the period during which the land is snow-covered increasing with increasing latitude and factors such as

altitude, and distance from coasts. When deposition occurs onto snow, the radioactivity is available for uptake by vegetation and biota only after the snow has melted. Surface contamination of plants, lichens, and mosses occurs as snow melts, by a process similar to interception of wet deposition. The extent of foliar uptake depends on the rapidity of snow melt, the topography of the landscape and the morphology of the vegetation. Subsequent lateral transport of radionuclides from melting snow depends on the extent of interception and catchment characteristics at both the large and small scale. If deposition occurs in the few months when vegetation is exposed to the atmosphere, then the transfer of radioactivity to herbivores is more rapid. Furthermore, for short-lived radionuclides, especially ^{131}I, deposition only leads to contamination of foodstuffs if it occurs just prior to, or during, the growing season.

The uptake of radioactivity by plants from soil occurs via the soil solution. The processes controlling radionuclide transfer between soil components and the soil solution are critical for bioavailability. For example, sorption of many radionuclides on non-specific cationic exchangeable sites is weaker than on more specific sites such as clay minerals. In addition, soil solution composition is important because of the competition between radionuclides and their stable analogues, e.g., strontium and Ca, and cesium and K. Therefore, soils with low potassium and clay mineral content will be more radioecologically vulnerable to radiocesium than soils with high potassium and clay mineral content. Strong sorption enhances retention of radionuclides in upper soil layers where most roots absorb nutrients. It is not clear whether the presence of a thin, organic layer in many Arctic ecosystems will enhance or reduce radionuclide mobility, nor is the effect of permafrost known.

3.5.2. Mushrooms

After the Chernobyl accident, the potential importance of mushroom consumption as a source of radiocesium intake became apparent, especially in the mid- to long-term after the accident. The first AMAP assessment found the importance of mushroom consumption to vary considerably between countries and population groups. Although a potentially important source, data on ^{137}Cs contamination of mushrooms at Arctic sites were not generally available. Such data are now available for Finland, Russia, and Norway.

3.5.2.1. Finland

Figure 3·18 shows the results of an extensive survey in Arctic Finland of fruiting bodies from a wide range of mushroom species (Rissanen *et al.*, 2002) The samples were obtained from 1983 onwards from four forest types at a site 70 km southeast of Rovaniemi. In 1993, the average ^{137}Cs deposition to the soil at the site was 0.8 to 0.9 kBq/m^2 declining to 0.7 to 0.8 kBq/m^2 in 1999. Owing to the low level of Chernobyl fallout in Finnish Lapland, about half the total ^{137}Cs present was due to global fallout.

There was no significant difference in the ^{137}Cs activity concentration of the mushroom species between the four types of forest stand. Analysis of the ^{134}Cs content showed that significant pre-Chernobyl ^{137}Cs was present in many species, again with about half the total ^{137}Cs

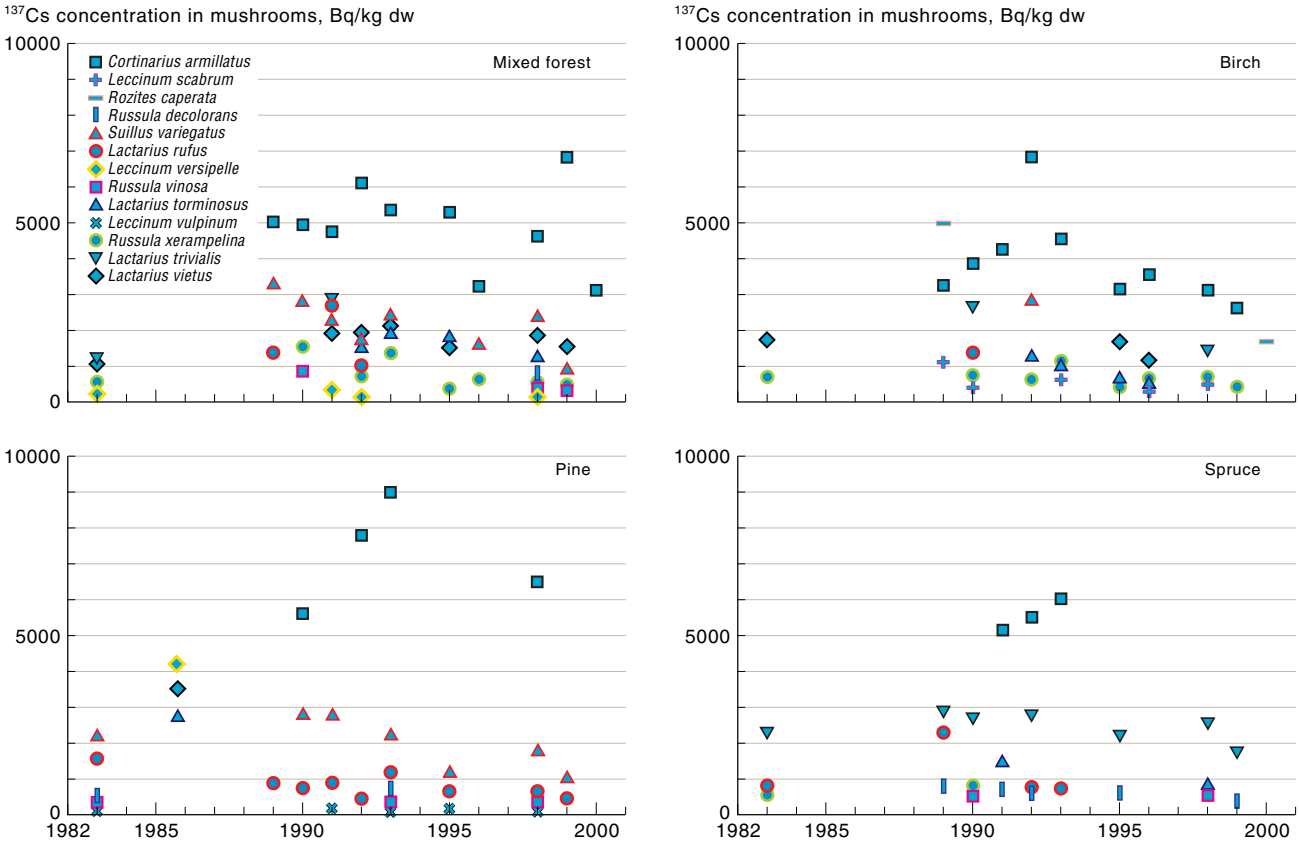

Figure 3·18. Activity concentrations of ^{137}Cs in a range of mushroom species in Finnish forest stands.

Box 3·2. *Aggregated transfer coefficients*

The transfer of radionuclides is quantified using the aggregated transfer coefficient (T_{ag}) defined as the activity concentration in an environmental compartment (often a food product) (in Bq/kg) divided by the corresponding radionuclide deposition in soil (in Bq/m^2); with units of m^2/kg.

$$T_{ag} = \frac{Activity\ concentration}{Deposition}$$

T_{ag} values were most commonly used in the former Soviet Union to quantify transfer to food products. In other countries, they are most often used for semi-natural products. They are therefore the most commonly used transfer quotient for Arctic ecosystems. High T_{ag} values, such as those derived for highly organic soils for radiocesium, indicate radioecologically vulnerable areas. T_{ag} values are time dependent, and can be combined with ecological half-lives to quantify changes with time. They are not appropriate for use when considering surface depositions onto plants during fallout.

The T_{ag} may be combined with the effective ecological half-life as follows:

$$C(t) = A \cdot T_{ag_0} \cdot \exp(-\lambda_r \cdot t) \cdot \left(a_1 \cdot \exp\left(-\frac{\ln 2}{T_{eff1}} \cdot t\right) + (1 - a_1) \cdot \exp\left(-\frac{\ln 2}{T_{eff2}} \cdot t\right)\right) \quad \text{Eqn. 3.2}$$

where $C(t)$ is the activity concentration of a given radionuclide in a food product at time t (Bq/kg); A is the surface deposition of radionuclide p (Bq/m^2); T_{ag_0} is the initial value of the aggregated transfer coefficient (m^2/kg); λ_r is the radioactive decay constant for radionuclide p (1/days); t is the time after deposition (days); and the other parameters are the same as in Equation 3.1 (Box 3·1).

The values of parameters T_{ag_0}, a_1, T_{eff1}, and T_{eff2} can be estimated on the basis of long-term measurements of ^{137}Cs and ^{90}Sr activity concentrations in different food products during the period of global fallout (about 40 years) and after the Chernobyl accident.

Using these techniques, it is possible to calculate integrated transfer coefficients (ITC) for ^{137}Cs and ^{90}Sr in different regions (AMAP, 1998). The calculation, assuming a single deposition event uses the formula:

$$ITC = \int_0^\infty \frac{C(t)dt}{A} = T_{ag_0} \cdot \int_0^\infty \exp(-\lambda_r \cdot t) \cdot \left(a_1 \cdot \exp\left(-\frac{\ln 2}{T_{eff1}} \cdot t\right) + (1 - a_1) \cdot \exp\left(-\frac{\ln 2}{T_{eff2}} \cdot t\right)\right) dt \quad \text{Eqn. 3.3}$$

or, after integration:

$$ITC = T_{ag_0} \cdot \left(a_1 \cdot \frac{T_{eff1}}{\ln 2} + (1 - a_1) \cdot \frac{T_{eff2}}{\ln 2}\right), \quad \text{(Bq/yr)/kg per kBq/m}^2 \quad \text{Eqn. 3.4}$$

In the case of ^{137}Cs, some areas received an additional input from the Chernobyl accident. As this input was a pulse input and possibly in a different chemical form, separate half-lives must be calculated for the Chernobyl input after removing the contribution from global fallout by extrapolating the pre-Chernobyl data using the model.

from Chernobyl and half from global fallout. The highest pre-Chernobyl value was 2390 Bq/kg dw in *Lactarius trivialis* in 1983. Although the variable nature of the data makes derivation of half-lives difficult, it is clear that ^{137}Cs uptake persists for many years in a wide variety of mushroom species.

After the Chernobyl accident, the most highly contaminated species in all four forests was the non-edible *Cortinarius armillatus*, with a maximum recorded value of 9030 Bq/kg dw in 1993. Of the edible species, the most contaminated were *Rozites caperata*, *Lactarius trivialis*, and *Suillus variegatus*, which is consistent with data for mushrooms in temperate areas.

3.5.2.2. **Russia**

Activity concentrations of ^{137}Cs in mushroom fruiting bodies from northwest Russia in 1989 to 1999 were low compared to those recorded in temperate areas of Europe. Highest activity concentrations were recorded in *Lactarius flexuosus* and *Xerocomus* spp. Over the period 1987 to 2000, ^{137}Cs T_{ag} values (Box 3·2) from soil to mushroom for two regions in northwest Russia were relatively constant (RTCP, 1999, 2000; Shutov *et al.*, 1999). Therefore, in Table 3·4, data from the whole sampling period have been collated to quantify transfer for the Kola Peninsula and the Mezen and NAO regions.

Table 3·4. Activity concentrations for ^{137}Cs in mushroom species (Bq/kg, mean ± SD, air dw) in northwest Russia for 1992 to 2000 and associated T_{ag} values (m^2/kg ± SD) for soil to mushroom (Borghuis *et al.*, 2002).

	Kola Peninsula			Mezen and Nenets AO regions		
	n	Activity concentration	T_{ag}	n	Activity concentration	T_{ag}
Leccinium aurantiacum	126	27.6±1.7	0.013±0.0007	27	21.8±3.9	0.0072±0.0012
Leccinium scabrum	63	55.2±5.3	0.026±0.0025	28	27.8±5.4	0.0082±0.0017
Suillus bovines	–	–	–	2	35.0±7.1	0.0104±0.0031
Russula spp.	70	63.9±4.5	0.029±0.002	19	54.4±11.3	0.016 ±0.0032
Paxillus spp.	–	–	–	5	45.3±20.3	0.016 ±0.0073
Xerocomus spp.	36	117 ±9	0.053±0.0044	4	77.4±20.9	0.028 ±0.0075
Lactarius rufus	6	133 ±21	0.067±0.01	5	59.4±8.5	0.018 ±0.0025
Lactarius flexuosus	3	235 ±69	0.144±0.042	4	98.3±36.8	0.035 ±0.013
Lactarius necator	14	50.3±9.5	0.026±0.0048	2	40.3±3.7	0.014 ±0.0013
Boletus edulis	1	8.9	0.0047	5	22.4±5.3	0.0075±0.0017
Suillus luteus	11	80 ±14	0.037±0.0066	1	40	0.013

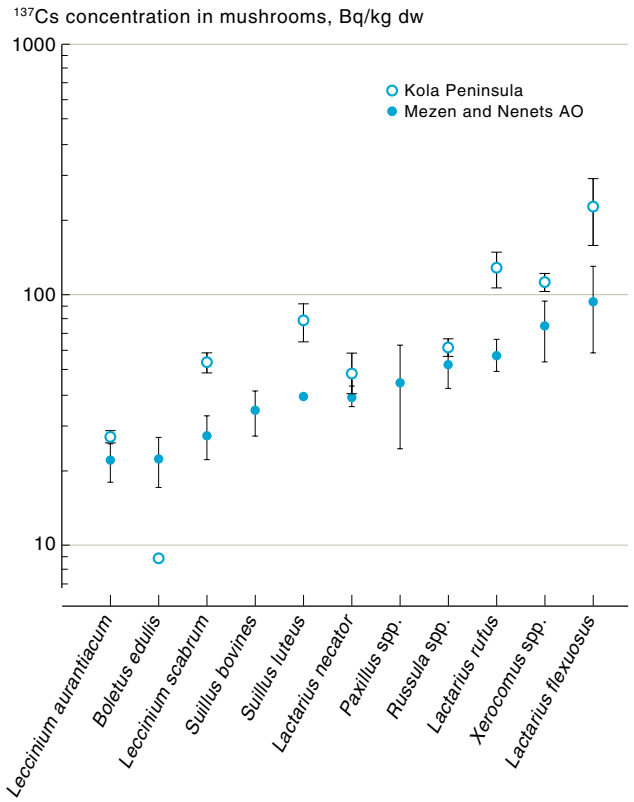

Figure 3·19. Activity concentrations of ^{137}Cs in various mushroom species from the Kola Peninsula and the Mezen and Nenets AO regions (Borghuis *et al.*, 2002).

The average ^{137}Cs T_{ag} values from soil to mushroom on the Kola Peninsula were significantly higher than for the NAO region (P = 0.00024, Table 3·4 and Figure 3·19).

3.5.2.3. Norway

Activity concentrations of ^{137}Cs in mushroom species from Troms and Finnmark collected in 1998 to 1999 are given in Table 3·5. Activity concentrations vary significantly between species. The activity concentration in species such as *Rozites caperata*, which is known to accumulate cesium, was a factor of ten to fifteen higher than in *Leccinum* spp. The activity concentration of ^{137}Cs in fungi with a high cesium transfer was about 500 times higher than in plant species from the same site, whereas the factor was approximately 50 for fungi with a moderate transfer.

Table 3·5. Activity concentrations for ^{137}Cs in mushroom species (Bq/kg dw ± SD) in Arctic Norway for 1998 to 1999 and corresponding T_{ag} values (m^2/kg ± SD) for soil to mushroom (Salbu, pers. comm., 2001).

	n	Activity concentration	T_{ag}
Boletus edulis	1	2097	2.17
Leccinum spp.	2	354 ± 175	0.37 ± 0.14
Rhozites caperata	4	3929 ± 3557	4.38 ± 4.47
Russula spp.	5	595 ± 417	0.48 ± 0.43

The T_{ag} values also varied between the different species. Species with a known high uptake of ^{137}Cs had a ten-fold higher T_{ag} value for ^{137}Cs than those with a lower uptake.

Overall, the data indicate lower activity concentrations in Arctic mushroom species than in those from nearby temperate areas of NW Europe due to the low levels of global and Chernobyl fallout in most of the areas sampled. Nevertheless, the ^{137}Cs activity concentrations are higher in mushrooms than in many other Arctic foodstuffs. There is, however, evidence of large T_{ag} values. Although activity concentrations in the most contaminated species vary between areas, the data are not directly comparable as different weight bases have been used.

3.5.3. Berries

Activity concentrations of ^{137}Cs in the berries of cloudberry (*Rubus chamaemorus*), bilberry (*Vaccinium myrtillus*), and cowberry (*Vaccinium vitis-idaea*) were measured at four sites in Lapland. In 1980 to 1981, before the Chernobyl accident, ^{137}Cs activity concentrations at Kittilä were 25 to 45 Bq/kg ww in cloudberry, 11 to 19 Bq/kg ww in bilberry, and 6 to 22 Bq/kg ww in cowberry (Rissanen *et al.*, 1987). Sufficient data were available at one of the four sites, Salla-Kuusamo, to indicate a slow change in concentration since the Chernobyl accident (Figure 3·20) and higher ^{137}Cs activity concentrations in cloudberry berries than in those of bilberry or cowberry.

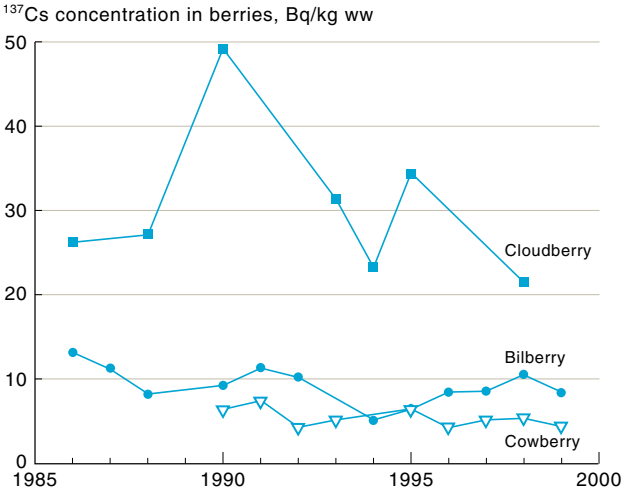

Figure 3·20. Changes over time in ^{137}Cs activity concentrations in three species of berries at Salla-Kuusamo in Lapland, Finland (Rissanen *et al.*, 1987).

The range in activity concentrations in these species at the four sites in Lapland since the Chernobyl accident is shown in Table 3·6. Despite the difficulties in comparing data with some time dependency after the Chernobyl accident, the sampling frequencies for the four species are sufficiently similar to conclude that there are differences between the species and that cloudberry is the most contaminated.

Table 3·6. Range in ^{137}Cs activity concentrations (Bq/kg ww) for berry species in Lapland for 1986 to 2001 (Rissanen, pers. comm., 2002).

	Rovaniemi	Inari	Salla-Kuusamo	Kittilä
Cloudberry	30-63	16-31	21-49	30-38
Bilberry	6-15	2-9	5-13	10-16
Cowberry	4-25	2-7	4-8	2-12

Table 3·7. Average ^{137}Cs activity concentrations for various berry species (Bq/kg ww; mean ± SE) in northwest Russia for 1998 to 1999 and associated T_{ag} values (m^2/kg) for soil to berries (Borghuis *et al.*, 2002).

	Kola Peninsula			Mezen and Nenets AO regions		
	n	Activity concentration	T_{ag}	n	Activity concentration	T_{ag}
Cloudberry	28	31.6±1.4	0.014±0.0007	50	26.9±2.3	0.0091±0.0008
Bilberry	64	11.1±1.0	0.0048±0.0004	24	14.2±1.3	0.0045±0.0004
Cowberry	192	7.2±0.4	0.0032±0.0002	40	3.7±0.4	0.0013±0.0002
Cranberry	5	17.9±4.7	0.0091±0.0024	38	11.6±1.1	0.0037±0.0004

Activity concentrations and T_{ag} values for ^{137}Cs in various berry species collected in 1998 to 1999 in northwest Russia are shown in Table 3·7. Activity concentrations and transfer of ^{137}Cs to cloudberry are higher than for the other species, including bilberry which is often the most contaminated berry in temperate areas.

Recent data on ^{137}Cs activity concentrations in the berries of bilberry and cloudberry from Arctic Norway in 1998 to 1999 are shown in Table 3·8 (note the data are on a dry and not wet weight basis). Although, there was a tendency for higher ^{137}Cs activity concentrations in cloudberry than bilberry the difference was not statistically significant. Some samples also contained ^{134}Cs from the Chernobyl accident (with maximum values of 33 Bq/kg dw in cloudberry and 45 Bq/kg dw in bilberry).

Table 3·8. Activity concentrations for ^{137}Cs (Bq/kg dw (Bq/kg ww*)) in bilberry and cloudberry in Arctic Norway for 1998 to 1999 (Salbu, pers. comm., 2001).

	n	Mean	Minimum	Maximum
Cloudberry	11	74.8 (11.2)	32.0 (4.8)	175.0 (26.5)
Bilberry	15	63.1 (9.5)	nd	167.0 (25.1)

nd: not detectable;
* fresh weight values estimated assuming 85% water content in berries.

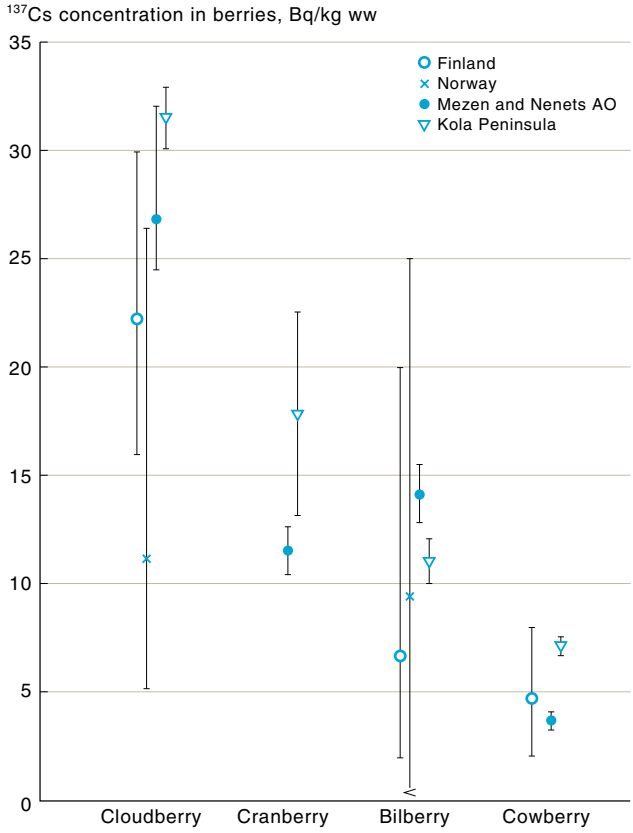

Figure 3·21. Activity concentrations of ^{137}Cs in berries from 1998 and 1999. Russian data: mean ± SD. Finnish and Norwegian data: mean and range. < : Under detection limit.

The Finnish, Russian, and Norwegian data are compared in Figure 3·21. Overall, the data indicate that cloudberry, which is a species typical of Arctic ecosystems, has the highest ^{137}Cs activity concentrations and is relatively vulnerable to radiocesium deposition compared to the other berry species. Cloudberry (and cranberry; *Vaccinium oxycoccus*) grow on wet, highly organic bogs, conditions which would be expected to lead to high radiocesium plant uptake from soil. Bilberry and cowberry grow on dry land, mainly in forests, in which the depth of the organic layer varies and sand is often present under the upper organic horizons.

The limited Finnish data suggest that the reduction with time in the ^{137}Cs content of Arctic berries is slow.

3.5.4. Milk

Sufficient data are now available on ^{137}Cs activity concentrations in milk to report on changes over time during the period of global fallout and after the Chernobyl accident for several locations in Finland, Sweden (annual averages), the Faroe Islands, Iceland, Russia, and Norway. The sample sites are shown in Figure 3·22. For ^{137}Cs, all time series show a peak in activity concentrations in the early-1960s with nearly 100 Bq/L detected in the Faroe Islands and Iceland. After the Chernobyl accident, some fallout was detected in milk in parts of Sweden, Arctic Finland, northern Norway, northwest Russia, and the Faroe Islands, with peak values of up to 20 Bq/L.

In most time series with an adequate sampling frequency, a strong seasonal signal can be seen with higher ^{137}Cs activity concentrations in the summer, when cows are put out to pasture or fed fresh grass. In some cases, the completeness of directly comparable time series has been affected by dairies closing down and consequent changes in the collection areas for those remaining.

Long-term time series on ^{90}Sr activity concentrations in milk are available for northern Finland, Sweden, Norway, and the Faroe Islands. Peak ^{90}Sr activity concentrations in the late-1960s were around 2 Bq/L in Fennoscandia. Little input from the Chernobyl accident was detected, although the weighted mean value in Sweden for ^{90}Sr in milk increased marginally from 0.10 Bq/L just before the Chernobyl accident to 0.13 Bq/L just after. In all time series with adequate sample numbers, a strong

Figure 3·22. Sites at which milk samples were collected.

seasonal signal in ^{90}Sr activity concentrations is seen with higher values in the summer.

The ^{137}Cs and ^{90}Sr activity concentration data are summarized in Table 3·9. Despite more ^{90}Sr being deposited during the period of global fallout, there were consistently higher ^{137}Cs activity concentrations in milk due to a roughly 15-fold higher transfer of ^{137}Cs to milk compared to ^{90}Sr.

3.5.4.1. Finland

Milk has been sampled at several sites in Finnish Lapland since the 1960s. Sampling was undertaken from 1963 to 1987 in Kursu, an area with boggy soils. Samples of milk powder were collected from a dairy in Rovaniemi from 1966 to 1975 and dairy milk from 1986 onwards. Samples from individual farms were collected from Apukka from 1975 to 1977 and 1986 to 1991 and from Vikajärvi from 1991 onwards. Kostiainen and Rissanen (2003) summarized the findings as follows: '*The highest ^{90}Sr and ^{137}Cs concentrations in the 1960s in Finnish milk were recorded in Lapland even though the deposition of ^{90}Sr and ^{137}Cs was no greater than in the rest of Finland. This was mainly due to the high proportion of peat soils and nutrient deficiency of the pastures in Lapland. Cs-137 deposition after the Chernobyl accident in 1986 in Lapland was less than 1 kBq/m^2, and ^{90}Sr deposition was so low that there was no detectable increase in the ^{90}Sr concentration in milk*.'

Although total ^{137}Cs deposition from nuclear weapons tests was similar in parts of Finland Lapland to that for Chernobyl fallout, ^{137}Cs activity concentrations in milk during the 1960s were considerably higher than after the Chernobyl accident. The pre-Chernobyl fallout, characterized by an annual maximum in summer, resulted in significant direct contamination of growing crops, whereas the Chernobyl deposition occurred before the start of the growing season in Lapland. The direct contamination of growing crops resulted in higher contamination levels compared to the short-term deposition following the Chernobyl accident when, before the start of the growing season, the food chain is hay to milk. The

Table 3·9. Activity concentrations for ^{137}Cs and ^{90}Sr in milk (Bq/L; mean (range)) (AMAP Data Centre).

		1964	1986	1998
^{137}Cs	Faroe Islands	51 (22-97)	5.8 (0.9-19)	0.8 (0.2-1.9)
	Arctic Finland	31 (27-36)	2.4 (0.5-5.2)	0.55 (0.5-0.6)
	Iceland	27 (7-83)	–	1.4 (0.9-2.4)
	Arctic Norway	16 (7-37)	6.2 (1-20)	–
	NW Arctic Russia	–	6.4 (4.0-10)	1.3 (0.04-0.3)
	Arctic Sweden	22 (14-30)	6.0 (2.0-13)	2.5 (1.5-4)
^{90}Sr	Faroe Islands	7.2 (3.3-12)*	–	0.04 (0.04-0.05)*
	Arctic Finland	1.1 (0.9-1.4)	–	0.05 (0.04-0.05)
	Iceland	–	–	–
	Arctic Norway	1.7 (1.3-2.2)	–	–
	NW Arctic Russia	–	–	0.17 (0.05-0.54)
	Arctic Sweden	1.4 (1.0-1.8)**	–	0.06 (0.03-0.09)

* the Faroese ^{90}Sr data were calculated from values in Bq/kg Ca, using an average of 1.2 g Ca in 1 kg milk;

** data from 1965, for Sweden, on average, ^{90}Sr levels declined by 18% from 1964 to 1965.

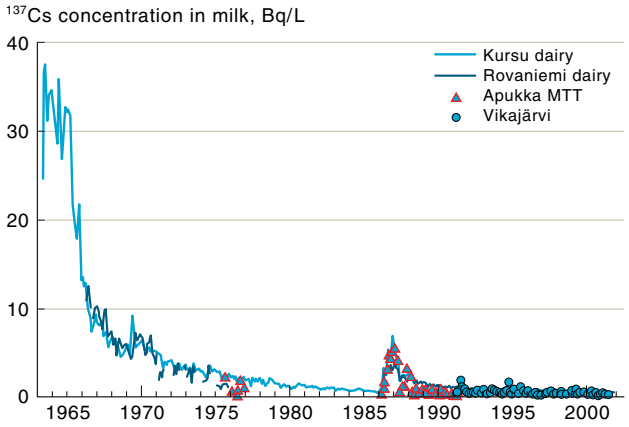

Figure 3·23. Time series for ^{137}Cs activity concentrations in milk from sites in northern Finland (Kostiainen and Rissanen, 2003).

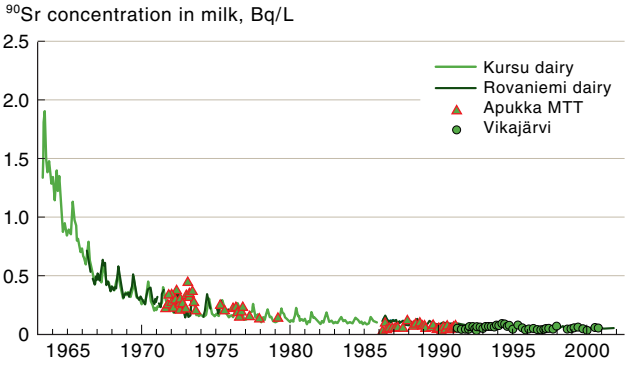

Figure 3·24. Time series for ^{90}Sr activity concentrations in milk from sites in northern Finland (Kostiainen and Rissanen, 2003).

time trends in ^{137}Cs and ^{90}Sr activity concentrations in milk are shown in Figures 3·23 and 3·24, respectively.

The increase in the activity concentrations of ^{137}Cs in milk due to the Chernobyl fallout was clearly visible in July 1986, and the peak lasted until summer 1987. In early-1987, activity concentrations in Kursu dairy milk were twice as high (5 to 7 Bq/L) as those from the Rovaniemi dairy (3.5 Bq/L), which collects milk from most of Lapland. The difference was due to the high frequency of peat soils and to the higher fallout in the area of the Kursu dairy compared to the average deposition in Lapland. The peak activity concentrations in the Apukka farm milk were about the same as in Kursu dairy milk, and decreased after summer 1988 to below the level of the Rovaniemi dairy milk. The activity concentrations of ^{137}Cs in Vikajärvi farm milk were similar to those in Rovaniemi dairy milk.

In contrast, ^{90}Sr in milk in Lapland is mainly from the fallout from nuclear weapons tests and the activity concentrations in milk from different areas of northern Finland were similar.

After the period of nuclear weapons tests, fallout continued to be deposited at lower, but not negligible levels, for several years. The T_{eff} values for ^{137}Cs and ^{90}Sr in milk after the peak values in 1963 were similar, at 2 yr in 1963 to 1966 and 5 yr in 1966 to 1975 (Table 3·10). During 1975 to 1985, the effective half-life of ^{137}Cs was still 5 yr, but for ^{90}Sr had increased to 10 yr. The effective half-lives of ^{137}Cs in milk after the peak concentrations resulting from the Chernobyl accident were similar to those in the 1960s to 1970s. The effective half-lives for ^{137}Cs in dairy milk and farm milk were about the same during the 1990s (at 7 to 8 yr), as were those for ^{90}Sr during the 1970s. There were larger fluctuations in the monthly activity concentrations of ^{137}Cs in farm milk than in dairy milk.

Table 3·10. T_{eff} values (yr) for ^{137}Cs and ^{90}Sr in milk at the Kursu and Rovaniemi dairies after the period of global fallout and the Chernobyl accident (Kostiainen and Rissanen, 2003).

	^{137}Cs		^{90}Sr	
	Kursu	Rovaniemi	Kursu	Rovaniemi
1963 to 1966	2.0		2.2	
1966 to 1975	4.8	3.2	5.1	6.1
1975 to 1985	5.3		10	
1987 to 1989		2.2		14
1989 to 1993		4.1		6.9
1993 to 2001		7.6		16

Kostiainen and Rissanen (2003) conclude that the whole of Lapland is vulnerable to radioactive contamination. The transfer of ^{137}Cs into milk from peat soils was more than twice that for the clay soils of southern Finland, and the T_{eff} values for milk in Lapland are twice those for intensively-cultivated clay pastures.

3.5.4.2. Sweden

Annual average ^{137}Cs activity concentrations at two sites in Arctic Sweden are shown in Figure 3·25. These indicate consistently higher values at Tärnaby than at Vittangi. A peak due to the Chernobyl accident can be seen in the values for Tärnaby milk.

A regular seasonal variation was evident for ^{137}Cs, and to a lesser degree for ^{90}Sr, during the 1950s and 1960s. Generally, peak levels occurred during the third quarter of the year coincident with the months of highest precipitation. Seasonal variations are especially pronounced in milk from the dairy in Tärnaby where cows graze in summer on natural pastures and in forests. A similar, but not as regular, seasonal variation was apparent after the Chernobyl accident.

T_{eff} values for ^{137}Cs in Swedish milk, estimated for the period from the peak of the atmospheric testing fallout until the Chernobyl accident, exhibit a fast and a slow component. The $T_{eff\,1}$ values were 1.4 and 1.8 yr, and the $T_{eff\,2}$ values 9.1 and 6.2 yr for Tärnaby and Vittiangi, respectively. The $T_{eff\,2}$ of 9.1 yr for Tärnaby is the longest $T_{eff\,2}$ for milk found in this study, but is not significantly longer than $T_{eff\,2}$ values found in northern Norway and on the Faroe Islands. Insufficient data were available to calculate post-Chernobyl T_{eff} values.

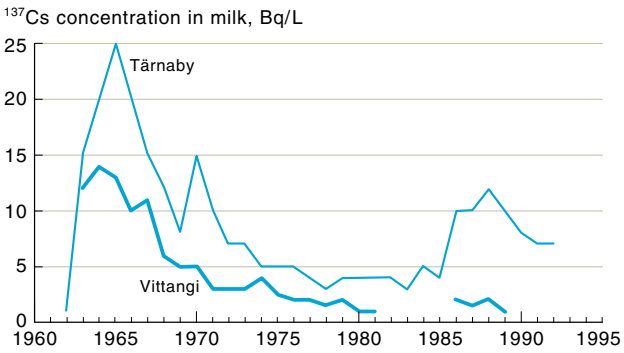

Figure 3·25. Annual average ^{137}Cs activity concentrations in milk for two sites in northern Sweden.

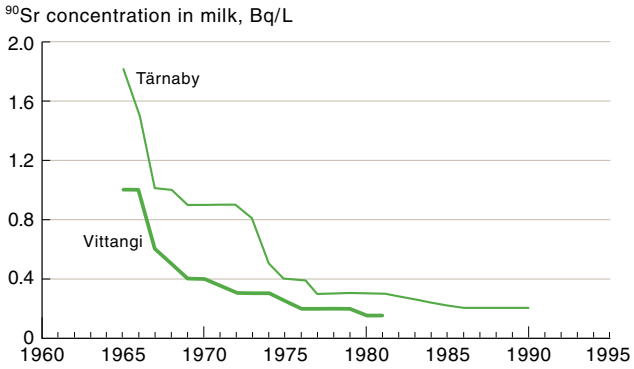

Figure 3·26. Annual average ^{90}Sr activity concentrations in milk for two sites in northern Sweden.

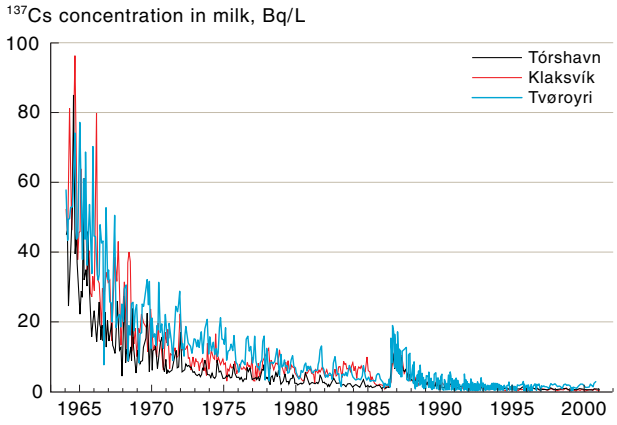

Figure 3·27. Weekly ^{137}Cs activity concentrations in milk at three locations in the Faroe Islands.

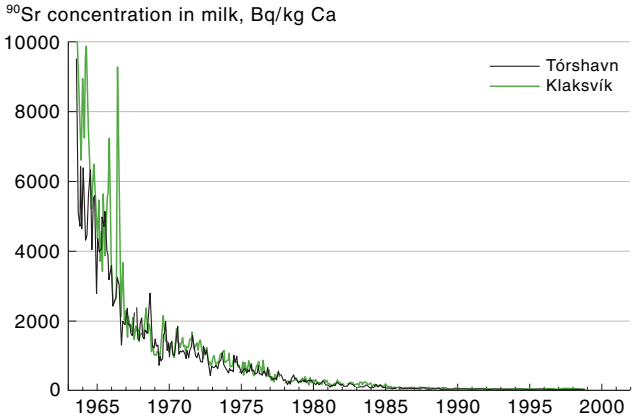

Figure 3·28. Weekly ^{90}Sr activity concentrations in milk (per kg Ca) from two sites in the Faroe Islands. The average calcium content of milk in the Faroe Islands is 1.2 g/L.

High precipitation combined with the high transfer of ^{137}Cs via the grass → cow food chain in natural pastures and forest environments was the main cause of the high activity concentrations and relatively slow decrease rates in milk at Tärnaby.

Annual average ^{90}Sr activity concentrations at the two sites in Arctic Sweden are shown in Figure 3·26. Again, consistently higher values occurred at Tärnaby than Vittangi. For ^{90}Sr, the short component (T_{eff1}) after 1963 was 2.1 yr, while the longer component (T_{eff2}) before and after the Chernobyl accident was 9.1 and 9.2 yr, respectively.

3.5.4.3. Faroe Islands

Milk has been sampled weekly in Tórshavn, Klaksvík, and Tvøroyri. The Klaksvík and Tvøroyri samples are from locally produced milk, while the Tórshavn samples are from a dairy which collects milk from most of the country. The ^{137}Cs activity concentrations are shown in Figure 3·27.

The T_{eff} for ^{137}Cs in milk from the Faroe Islands has been calculated to 6.5 to 8.8 yr for the long (second) component in global fallout, whereas the short (first) component was between 1.0 and 1.8 yr. For Chernobyl fallout, the T_{eff} was from 1.3 to 1.8 yr.

Activity concentrations of ^{90}Sr in milk have been measured as Bq/kg Ca (Figure 3·28). With an average Ca concentration of 1.2 g/L, the peak values of ~10000 Bq/kg Ca correspond to ~12 Bq/L. The short and long components in the T_{eff} for ^{90}Sr were found to be 1.1 to 1.4 and 5.2 to 5.5 years, respectively.

3.5.4.4. Iceland

The ^{137}Cs activity concentrations in Icelandic milk in the mid-1960s are shown in Figure 3·29. The average T_{eff} for ^{137}Cs in milk from these dairies during 1964 to 1967 was 3.0 yr, similar to that in Kursu in northern Finland. Soon after this period, the rate of reduction of Cs con-

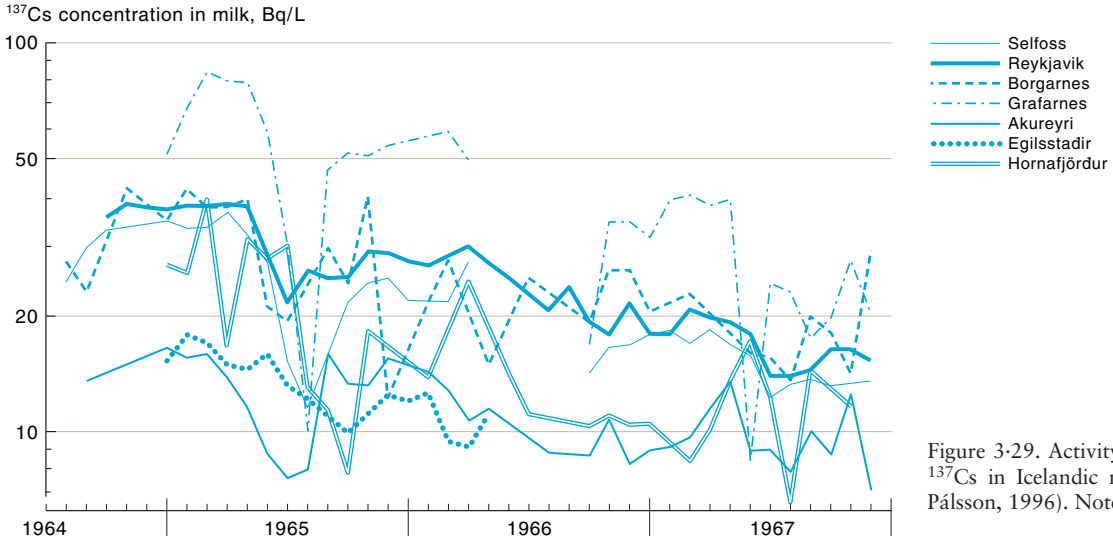

Figure 3·29. Activity concentrations for ^{137}Cs in Icelandic milk (adapted from Pálsson, 1996). Note logarithmic scale.

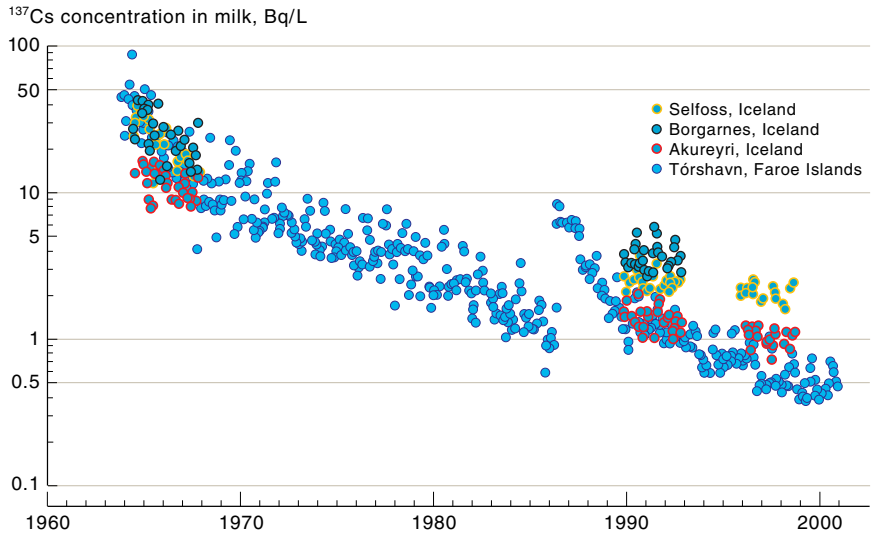

Figure 3·30. A comparison of ^{137}Cs activity concentrations in Icelandic and Faroe Island milk.

centration in milk decreased. The Icelandic data exhibit a high spatial variability between sample areas, with relatively high maxima compared to other Arctic areas. In particular, the values at Grafarnes are relatively high for many years although there are occasional dips. A comparison with the ^{137}Cs deposition map in Section 3.7.2.2 (Figure 3·52) shows good agreement between the pattern of deposition and ^{137}Cs activity concentrations in milk, with the highest values in the southwest and the lowest in the north.

Activity concentrations of ^{137}Cs in Icelandic (Selfoss, Borgarnes, and Akureyri) and Faroe Island (Tórshavn) milk are compared in Figure 3·30. This shows that activity concentrations in milk in the 1960s were slightly lower in Iceland than the Faroe Islands. However, in the 1990s, the levels in Iceland were higher in the three study areas than in Tórshavn, although the relative difference in concentration between the milk from the three Icelandic regions remained the same (Pálsson *et al.*, 2002b). This is despite the Faroe Islands receiving some fallout from the Chernobyl accident (see Figure 3·30), in contrast to Iceland where it was hardly detectable. The comparison indicates that the reduction in ^{137}Cs activity concentrations in milk has been slower in Iceland than the Faroe Islands. This is also reflected in the lower slope of the Icelandic data sets during the 1990s. The explanation may lie in the lower cesium-binding properties of the young volcanic Icelandic soils compared to those of the Faroe Islands (Sigurgeirsson *et al.*, 2002). It is thus concluded, as was also the case for Lapland, that Iceland is vulnerable to radioactive contamination due to high transfer of ^{137}Cs from soil to milk and very high T_{eff} values.

3.5.4.5. Norway

Figures 3·31 and 3·32 show ^{137}Cs and ^{90}Sr activity concentrations in milk from four dairies in Arctic Norway, since 1960. Because precipitation in Arctic Norway is low compared to precipitation on the southwest coasts, the deposition of global fallout in Arctic Norway is also relatively low compared to other parts of the country. In general, the spatial variation in activity concentrations broadly follows that for precipitation and therefore global fallout. The highest values were measured at Bodø, a coastal area. However, there is an anomaly in that levels at Kautokeino are higher than would be expected from the low rate of precipitation in this area. A possible explanation is that cows in Kautokeino were fed lichen as fodder (Eikelmann, pers. comm., 2002).

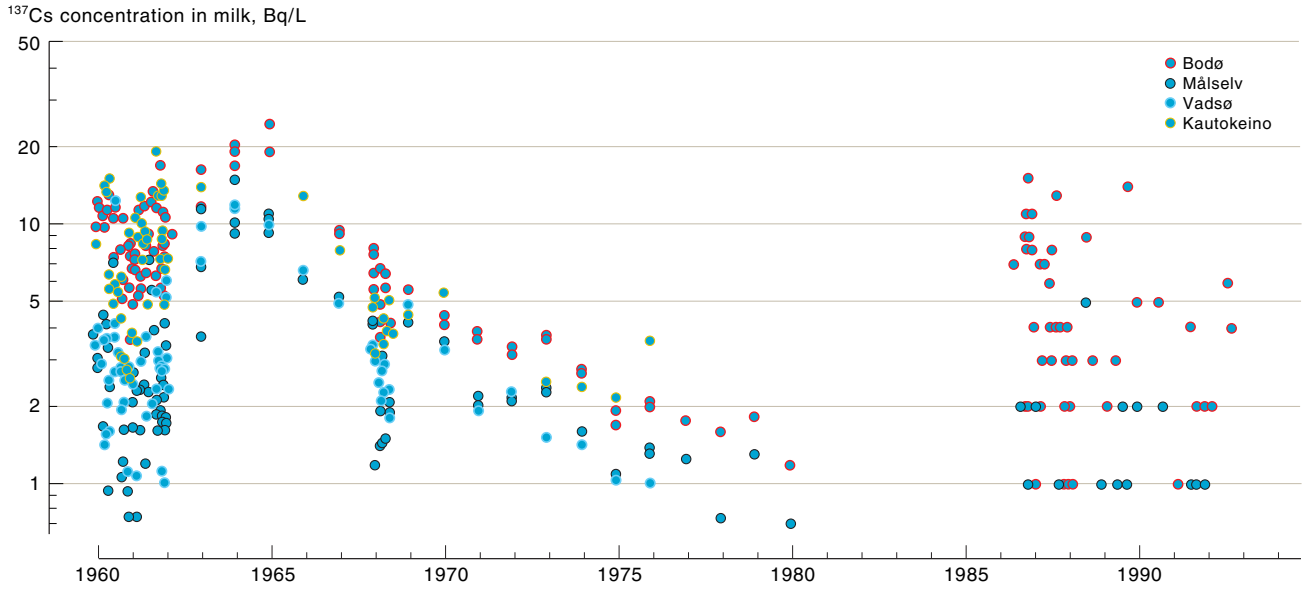

Figure 3·31. Activity concentrations of ^{137}Cs in milk at four locations in northern Norway.

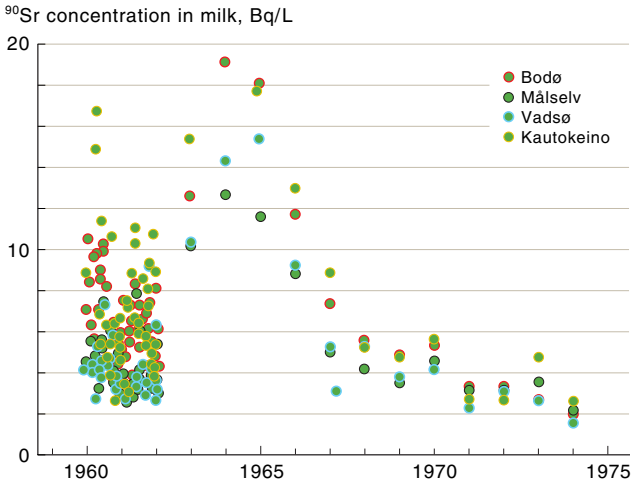

Figure 3·32. Activity concentrations of ^{90}Sr in milk at four locations in northern Norway.

The impact of the Chernobyl accident is variable, with fallout most noticeable at Bodø.

3.5.4.6. Russia

Sporadic data on ^{137}Cs activity concentrations in Russian milk from 1986 onwards support the findings in sections 3.5.4.1. to 3.5.4.5. Due to the limited amount of data available for milk in the Russian Arctic, a detailed assessment is not possible.

3.5.4.7. Trends

The data in sections 3.5.4.1 to 3.5.4.6 were collected using various techniques over different periods of time, which makes direct comparisons difficult. Nevertheless, a rough comparison between countries is attempted in Table 3·11 using double exponential T_{eff} values and a consistent method for the various sites in the different countries. All the half-lives were calculated using the statistical package 'R', and its nls- (nonlinear least squares) library. The T_{eff} values are estimated for global fallout data from 1964 to 1985, and for Chernobyl fallout from 1986 onward. The global fallout values are expected to be greater than for a single pulse input as global fallout continued to be deposited after the peak values in the mid-1960s.

A comparison of ^{137}Cs activity concentrations in milk from a range of locations in northern Scandinavia and the Faroe Islands in 1964, the year for which most data were available, shows that even though some samples came from dairies taking milk from a wide geographical area, a relationship is still evident between milk contamination and global fallout, which is linked to precipitation rates (Figure 3·33). Thus, for instance, within Norway and Iceland the west coast, which receives the highest rate of precipitation, produced the most contaminated milk. Some of the lowest values were recorded in the areas of Norway closest to the Novaya Zemlya test site.

Table 3·11. T_{eff} values (yr ± SD) for ^{137}Cs and ^{90}Sr activity concentrations in milk from various Arctic areas.

	Faroe Islands			Finland			Norway				Sweden	
	Klaksvík	Tórshavn	Tvøroyri	Apukka	Kursu	Rovaniemi	Bodø	Vadsø	Målselv	Kautokeino	Tärnaby	Vittiangi
^{137}Cs												
Global fallout												
T_{eff_1}	1.5±0.1	1.0±0.1	1.8±0.2		1.0±0.1		1.9±0.6	1.6±0.4	1.5±0.3	1.1±0.3	1.4±0.4	1.8±0.6
T_{eff_2}	7.1±0.5	6.5±0.4	8.8±0.7		4.5±0.7		4.5±1.2	5.1±1.3	6.1±1.2	6.0±2.0	9.1±1.1	6.2±1.0
Chernobyl fallout												
T_{eff}	1.3±0.1	1.8±0.1	1.8±0.1	0.7±0.1		3.4±0.1						
^{90}Sr												
T_{eff_1}	1.0±0.1	1.4±0.1			1.3±0.1		1.8±0.6	n.a.	n.a.	1.5±0.4	3.0±1.0	1.4±0.3
T_{eff_2}	5.2±0.4	5.5±0.5			8.4±0.3		4.0±1.0	n.a.	n.a.	4.6±1.3	9.0±2.0	8.5±1.0

n.a.: no statistically valid half-life could be calculated.

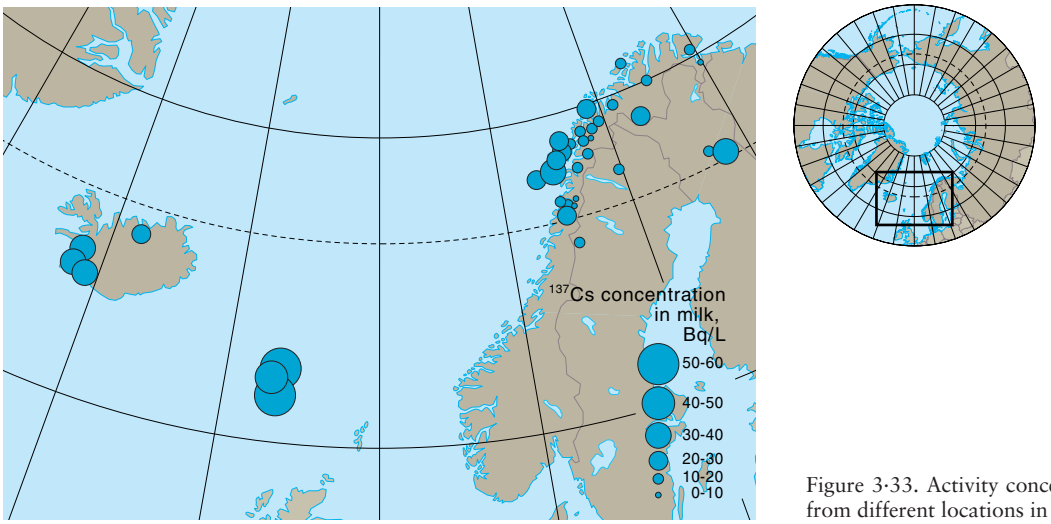

Figure 3·33. Activity concentrations of ^{137}Cs in milk from different locations in Nordic areas in 1964.

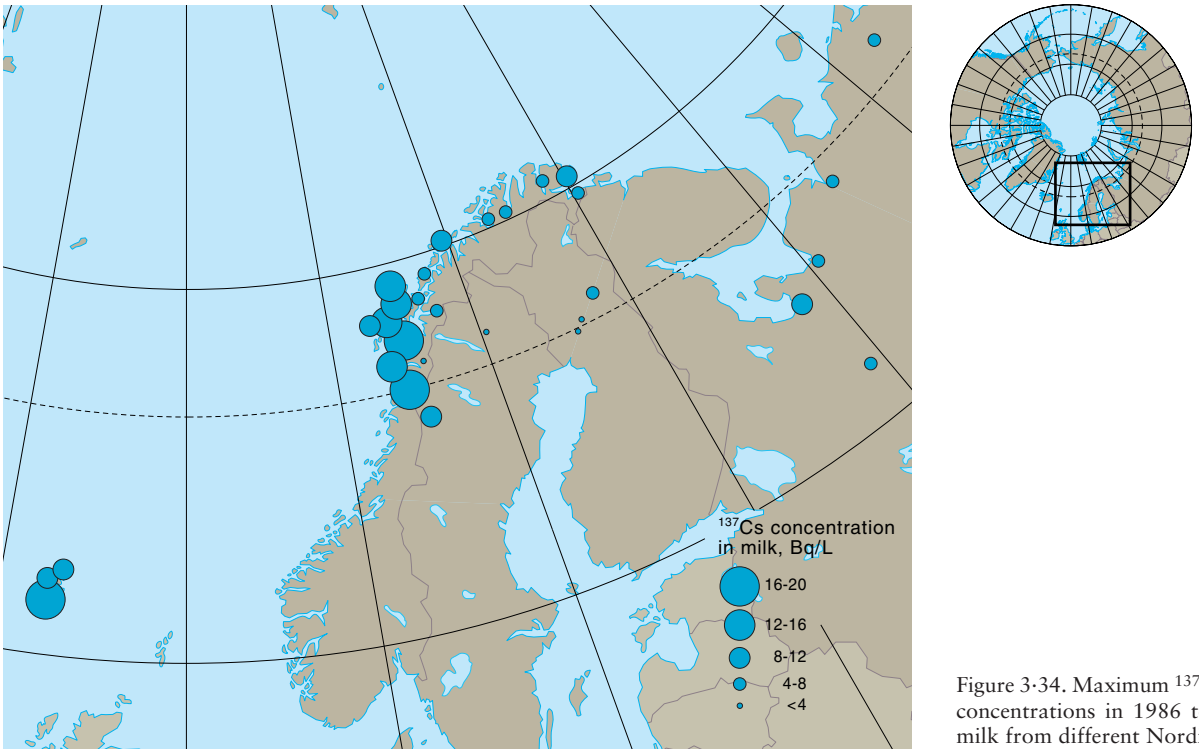

Figure 3·34. Maximum ^{137}Cs activity concentrations in 1986 to 1987 in milk from different Nordic areas.

The highest ^{137}Cs activity concentrations in milk, after the Chernobyl accident, are shown in Figure 3·34. In all Arctic areas, levels were higher in the global fallout period than after the Chernobyl accident.

There were no statistically significant differences in the average ^{137}Cs T_{ag} values from soil to milk (and beef) produced in two Russian Arctic regions (Table 3·12) after global fallout.

Table 3·12. ^{137}Cs T_{ag} values (10^{-3} m^2/kg) for beef and milk in northwest Russia (Borghuis *et al.*, 2002).

	n	Mean ± SD	*t*-test
Beef, 1978-1985			
Kola	7	0.28 ± 0.08	P = 0.10
Nenets AO	34	0.24 ± 0.12	
Milk, 1974-1978			
Kola	21	0.14 ± 0.07	P = 0.68
Nenets AO	7	0.12 ± 0.05	
Milk, 1978-1985			
Kola	12	0.082 ± 0.030	P = 0.081
Nenets AO	39	0.062 ± 0.035	

3.5.5. Lichen and reindeer

The long-term trend in ^{137}Cs activity concentrations in reindeer meat from the Kola Peninsula and the NAO region in Russia is shown in Figure 3·35. Over a 40-year period, the ^{137}Cs activity concentrations in meat from the Kola Peninsula were consistently statistically significantly higher than in the NAO region. One reason for the difference is the greater amount of global fallout on the Kola Peninsula than in the NAO region, but there may also be other contributory factors connected with pasture conditions.

Tables 3·13 and 3·14 present parameters describing the transfer and temporal variability in ^{137}Cs activity concentrations, such as T_{ag_0}, A_0, T_1, and T_2 (see Boxes 3·1 and 3·2), for lichen and reindeer meat, in northern Russia and northern Fennoscandia. These values are based on the assumption that the interception rate was the same for global fallout and Chernobyl fallout. Over the 40-year observation period, activity concentrations in lichen and reindeer meat are well described in all regions by a double exponential fit.

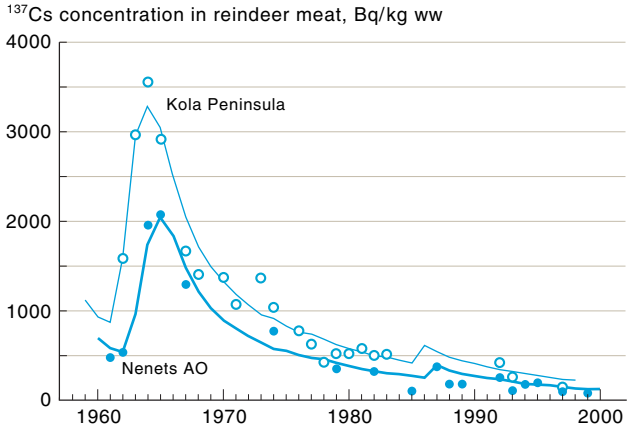

Figure 3·35. Changes with time in ^{137}Cs activity concentration in reindeer meat collected in the Kola Peninsula and NAO regions since 1961 (Borghuis *et al.*, 2002).

The T_{ag_0} values in Tables 3·13 and 3·14 were estimated on the basis of annual data. However, significant seasonal fluctuations are possible. Thus, for example, if the deposition in winter occurs onto snow, lichen is contaminated during the period of snowmelt and the interception fraction decreases through runoff as part of the activity is lost in snowmelt. In contrast, for a single dry deposition event, the interception fraction can be much higher than the annual mean value. Annual T_{ag_0} values must therefore be used with care as they are likely to underestimate single pulse deposition and do not take seasonal variation or (wet versus dry) deposition conditions into account.

Table 3·13. Initial values of the aggregated transfer coefficient (m^2/yr), ecological half-lives (T_{ec}, yr), and effective ecological half-lives (T_{eff}, yr) for ^{137}Cs and ^{90}Sr in lichen (dw). a is a parameter partitioning the decay between the two half-lives (see Box 3·1 and Box 3·2)

		T_{ag0}	a_1	T_{ec1}	T_{ec2}	T_{eff1}	T_{eff2}
^{137}Cs	Kola Peninsula	1.4	0.80	2.0	20	1.9	12
	Northern Sweden	1.4	0.52	3.4	14	3.0	10
^{90}Sr	Kola Peninsula	0.7	0.72	0.70	20	0.7	12

Table 3·14. Initial values of the aggregated transfer coefficient (m^2/yr), ecological half-lives (yr), and effective ecological half-lives (yr) for ^{137}Cs activity concentrations in reindeer meat (ww).

	T_{ag0}	a_1	T_{ec1}	T_{ec2}	T_{eff1}	T_{eff2}
Kola Peninsula	1.7	0.82	2.0	18	1.9	11
Nenets AO	1.2	0.81	1.8	15.6	1.5	10
Kautokeino (Norway)	1.8	0.83	1.2	13	1.1	9.0

Activity concentrations of ^{137}Cs in reindeer meat also vary over the year due to changes in food selection. In summer, reindeer eat herbaceous vegetation. In autumn they can eat large quantities of mushrooms. In winter, they mainly eat ground and arboreal lichens, which have a higher radiocesium content than herbaceous vegetation which they dig out from under the snow. The high interception of radionuclides by lichen, particularly radiocesium, is one of the key factors contributing to the most vulnerable Arctic food pathway, lichen → reindeer → man (Figures 3·36 and 3·37; Åhman and Nylén, 1998).

If contamination occurs during deep snow cover, then reindeer will ingest contaminated snow. The extent of intake is determined by the amount of snow ingested, but also by further snowfalls that may be less contaminated. The extent of contamination received from lichen depends on whether the contaminated snow overlying the lichen melts allowing the lichen to intercept the radioactivity. There can be a significant delay in reindeer attaining high levels of radiocesium from lichen owing to the protection of lichen by snow.

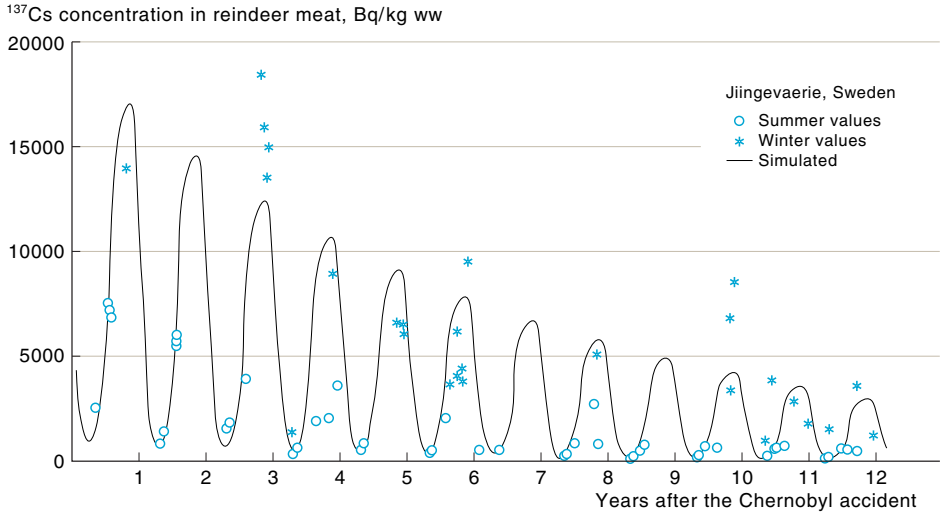

Figure 3·36. Activity concentrations of ^{137}Cs in fresh reindeer meat from the Jiingevaerie herding district in Sweden (Åhman and Nylén, 1998).

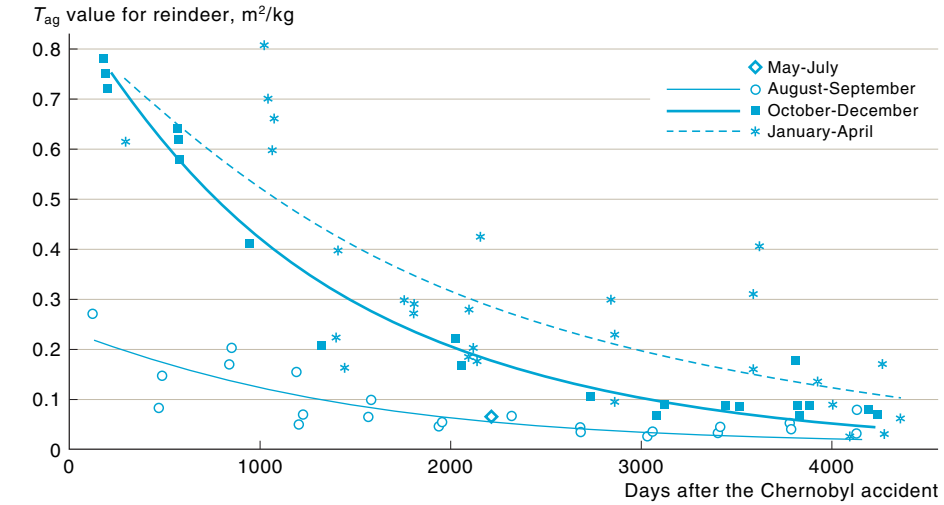

Figure 3·37. Seasonal variation in T_{ag} values for reindeer following the Chernobyl accident (Åhman and Nylén, 1998).

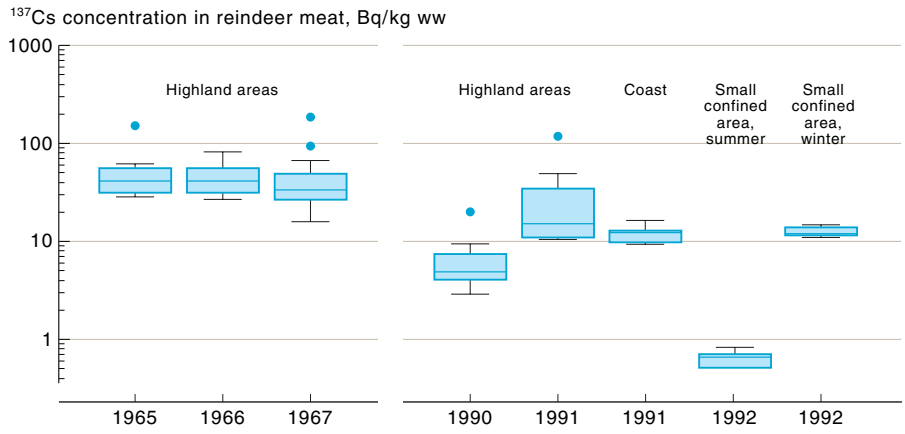

Figure 3·38. ^{137}Cs activity concentrations in meat from free-ranging reindeer from Iceland for different hunting seasons. Datasets for 1965 to 1991 are from highland areas and the coast, and the 1992 data from a more homogeneous restricted area.

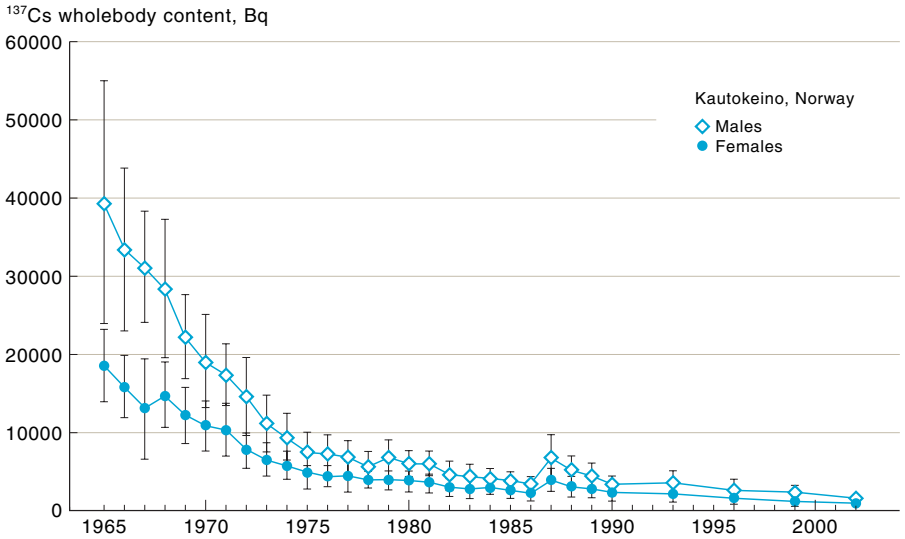

Figure 3·39. Wholebody measurements of ^{137}Cs in reindeer herders from Kautokeino, Norway.

Significant changes from year to year due to changes in food selection and variability in the ^{137}Cs activity concentrations in plants are also clearly visible in a study of wild free-ranging reindeer in Iceland. They are relatively few in number (around 3000) and roam within a highland region of different types of vegetation. A few samples were obtained each year by inspectors monitoring the hunting together with information about the location in which the animals had been shot and where they had grazed (Figure 3·38).

If the herd had grazed a large area with different types of vegetation, this was reflected in greater variability in the activity concentrations in the meat. If the herd had grazed a relatively small or uniform area, then activity concentrations showed little variability. The grazing areas selected by the herd varied from year to year, influenced for example by climate and the state of the vegetation. The lowest values (<1 Bq/kg ww) occurred in samples from 1992 from a herd that had grazed a relatively homogeneous area confined by glacial rivers and close to the glacier Vatnajökull.

3.6. Humans

Trends in wholebody measurements of ^{137}Cs presented in the first AMAP assessment have been extended by new data on wholebody measurements for northern Norway, Finland, and the Kola Peninsula and NAO regions of northwest Russia (see Table 3·15). An example of the temporal trend since 1965 is shown in Figure 3·39.

Table 3·15. Wholebody measurements (Bq) for reindeer herders (STUK; NRPA; Borghuis *et al.*, 2002).

		Males	Females	Average
Russia				
Kola	1999			3250±250
Finland				
Inari	1995			3300
	1997			3000
Norway				
Kautokeino	1996	2600±1400	1400±600	
	1999	2200±800	1100±400	
	2002	1414	872	

3.7. Site-specific data

3.7.1. Faroe Islands

The Faroe Islands were not addressed in the first AMAP radioactivity assessment. They comprise 18 islands between 6°15'W and 7°41'W and 61°20'N and 62°24'N with a total land surface area of 1399 km^2 (Figure 3·40). The land is mountainous, with the highest peak 882 m above sea level. There were 46 180 inhabitants on 31 December 2000. There is no woodland on the Faroe Islands. Land cover is dominated by rough, semi-natural pasture, and is grazed throughout the year by around 70 000 sheep and some cattle.

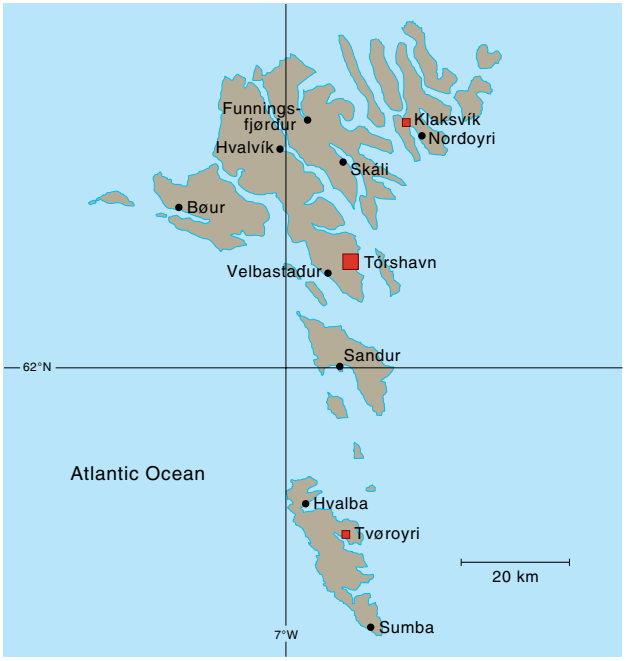

Figure 3·40. The Faroe Islands.

3.7.1.1. Climate

The climate is milder than might be expected at a latitude of 62°N due to the influence of the North Atlantic Current (the 'Gulf Stream'). Measurements at synoptic weather stations within 100 m of sea level, indicate an average annual air temperature of 6 to 7°C, with average winter and summer air temperatures of 3 to 4°C and 9 to 10°C, respectively (Cappelen and Laursen, 1998; Lysgaard, 1969). Only minor differences in air temperature occur between the synoptic stations. There is significant spatial variation in precipitation rates, however, due to the combined effects of meteorology and topography. These synoptic data are not totally representative of the Faroese climate however, as the zone within 100 m of sea level only covers 10% of land area. The climatic conditions change gradually from cool temperate oceanic conditions at the coast to Arctic conditions in the mountains (Mortensen, 2002). The annual average temperature at a new weather station on the mountain Sornfelli, 722 m above sea level, was 1.71°C in 2000 (Mortensen, 2002).

3.7.1.2. Cesium-137 and ^{90}Sr in precipitation and foodstuffs

Measurements of environmental radioactivity have been carried out on samples from the Faroe Islands since 1962, with an emphasis on terrestrial and marine foodstuffs.

Precipitation

Monthly precipitation samples have been obtained for radioactivity analyses since the 1960s. Annual average ^{137}Cs deposition rates in Tórshavn in the central part of the country and in Klaksvík in the north, shown in Figure 3·41, were highest in the early-1960s and showed a pronounced peak following the Chernobyl accident in 1986. Pre-1986 ^{137}Cs data are based on ^{90}Sr measurements (using a ^{137}Cs : ^{90}Sr fallout ratio of 1.6); after 1986 they are actual measurements (Figures 3·41 and 3·42).

Lamb meat

Figures 3·43 and 3·44 show annual average ^{137}Cs and ^{90}Sr activity concentrations in lamb meat, based on a few samples collected mainly in October. Trend analyses were not possible because the samples were collected from different places and there is significant spatial vari-

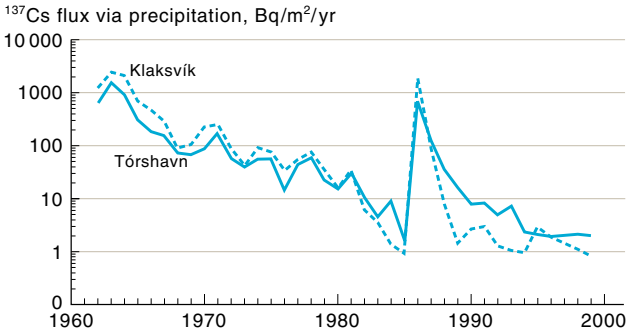

Figure 3·41. Annual average ^{137}Cs levels in precipitation from Tórshavn and Klaksvík, Faroe Islands.

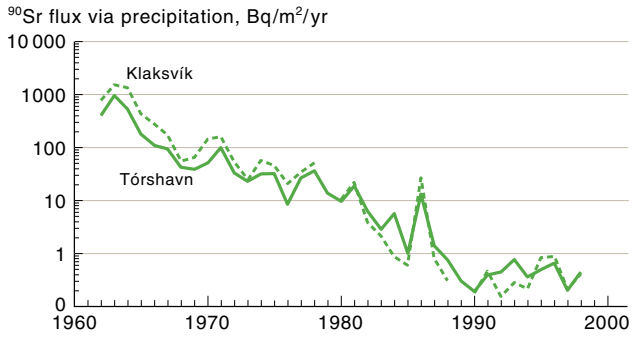

Figure 3·42. Activity concentrations of ^{90}Sr in precipitation from Tórshavn and Klaksvík, Faroe Islands.

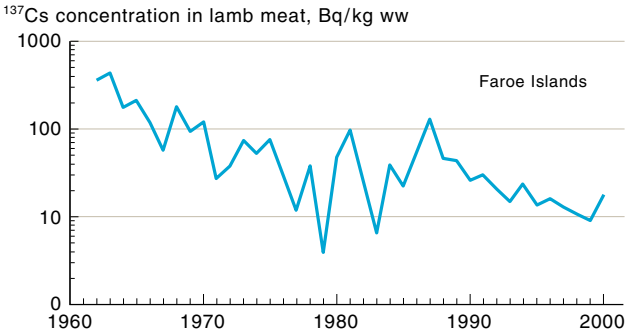

Figure 3·43. Annual average ^{137}Cs activity concentrations in lamb meat, Faroe Islands.

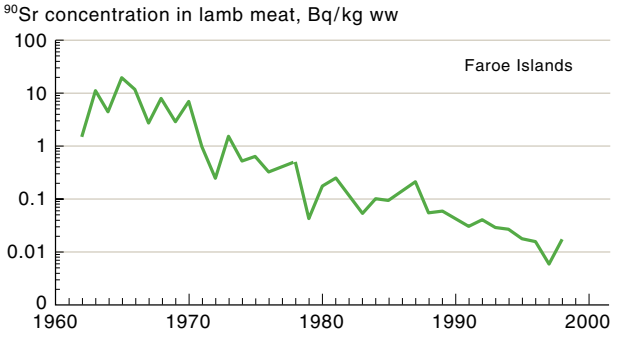

Figure 3·44. Annual average ^{90}Sr activity concentrations in lamb meat, Faroe Islands.

ation in contamination across the country (Joensen, 1999). During the 1990s however, the samples were collected consistently from the same places. Figure 3·43 indicates increased ^{137}Cs activity concentrations after the Chernobyl accident. In contrast, ^{90}Sr activity concentrations in lamb meat are lower and are not affected by Chernobyl fallout (Figure 3·44).

Drinking water

Figure 3·45 shows annual average activity concentrations for ^{90}Sr in tap water from Tórshavn and Klaksvík since 1962. Faroese drinking water is obtained from surface water. Sampling frequency has varied from monthly in the early-1960s to an annual summer value in the 1990s. No ^{90}Sr from Chernobyl was observed in drinking water.

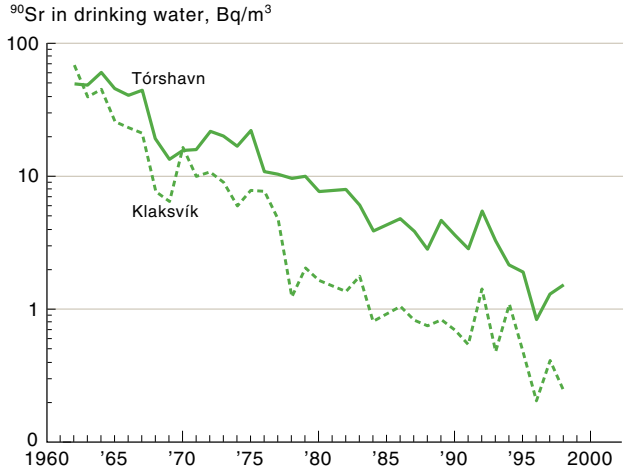

Figure 3·45. Annual average activity concentrations for ^{90}Sr in drinking water from Tórshavn and Klaksvík.

Effective ecological half-lives

Table 3·16 presents T_{eff} values for ^{137}Cs and ^{90}Sr in milk, lamb meat, precipitation, and drinking water. These are estimated by regressing the logarithm of the measured activities against time. The ^{137}Cs T_{eff} values in foodstuffs range from 4.9 to 8.7 yr, while those for ^{90}Sr range from 3.7 to 4.5 yr. Figures 3·41 to 3·45 and Table 3·16 indicate a tendency for increasing T_{eff} values with time. Despite the small geographical extent of the Faroe Islands, spatial variation in the T_{eff} values is apparent.

3.7.1.3. Transfer of ^{137}Cs within the lamb food chain in semi-natural pastures

Spatial variability in ^{137}Cs transfer to lamb has been evaluated by comparing its characteristics at nine uncultivated pastures during 1990 to 2000. Their locations are shown in Figure 3·40 (locations marked by black symbols). T_{eff} values and transfer factors for various components of the lamb food chain were estimated for the nine pastures. The soil at each site was previously characterized by Hove *et al.* (1994) and Joensen (1999).

Soil

There are large temporal and spatial variations within and between pastures in terms of ^{137}Cs deposition in the upper 10 cm soil layer (Figure 3·46). Between 50 and 80% of the deposition in this layer occurs in the upper 5 cm. Deposition ranges from 2000 to 8000 Bq/m^2. There was no consistent pattern of change with time for the nine pastures; with clear declines in some pastures, but not in others.

Soil pH in the nine pastures was between 4.4 and 5.3, and loss on ignition was 50 to 70% (Joensen, 1999). Thus these soils are acidic with a high organic matter content: conditions that favour a high uptake of radiocesium.

Grass

Activity concentrations of ^{137}Cs in mixed grass decreased in most pastures during the 1990s, with the highest levels in Hvalvík and the lowest in Hvalba. Although Figure 3·46 shows little clear difference in deposition between the nine sites, change over time in the grass varied widely (Figure 3·47). In some pastures early declines were evident, which were presumably due to the declining Chernobyl input, while others showed no overall decline, and others an approximate 20-fold decline.

Soil-to-grass T_{ag} values for the nine pastures are shown in Figure 3·48. The highest values occurred in Hvalvík and the lowest in Hvalba and Sandur. A multiple linear regression analysis between T_{ag} values and pH, loss on ignition, and potassium, showed loss on ignition to be the most significant factor (Joensen, 1999), and that the regression coefficient is negative for pH and potassium, and positive for loss on ignition. There were two orders of magnitude between the lowest and highest

Table 3·16. Estimated T_{eff} values (yr) based Figures 3.41 to 3.45. r^2 (shown in brackets) from linear regression of the natural logarithm of the measured activity concentrations against time.

		Milk	Lamb meat	Precipitation	Drinking water
^{137}Cs*	Klaksvík	6.2 (0.846)			
	Tórshavn	4.9 (0.959)			
	Tvøroyri	5.8 (0.971)			
	'Faroes'		5.5 (0.555)		
^{137}Cs**	Klaksvík	5.3 (0.980)		3.3 (0.940)	
	Tórshavn	6.4 (0.977)		4.3 (0.932)	
	Tvøroyri	7.1 (0.987)			
	'Faroes'		8.7 (0.772)		
^{90}Sr***	Klaksvík	4.4 (0.980)		2.8 (0.941)	5.1 (0.958)
	Tórshavn	4.4 (0.985)		3.2 (0.953)	6.9 (0.966)
	Tvøroyri	4.5 (0.986)			
	'Faroes'		3.7 (0.935)		

*only pre-Chernobyl data; **all data except for 1986 to 1992 (to avoid the Chernobyl peak); ***all data.

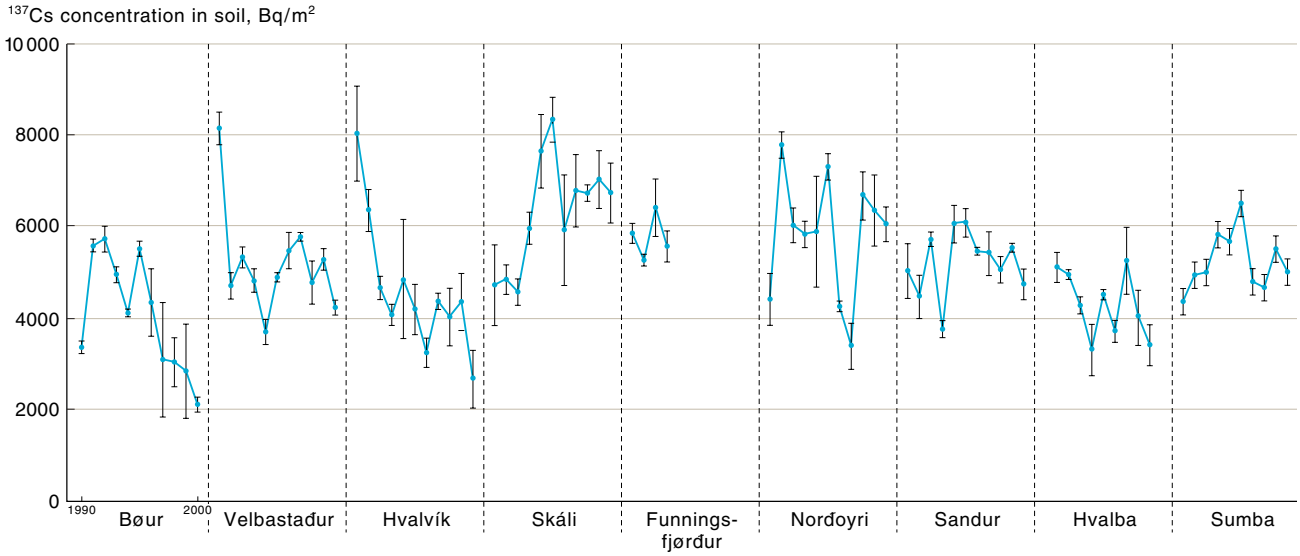

Figure 3·46. Annual average (±SE) ^{137}Cs deposition to surface soil (upper 10 cm) between 1990 and 2000 (1999 for Hvalba and Sumba) in the Faroe Islands (for locations see Figure 3.40).

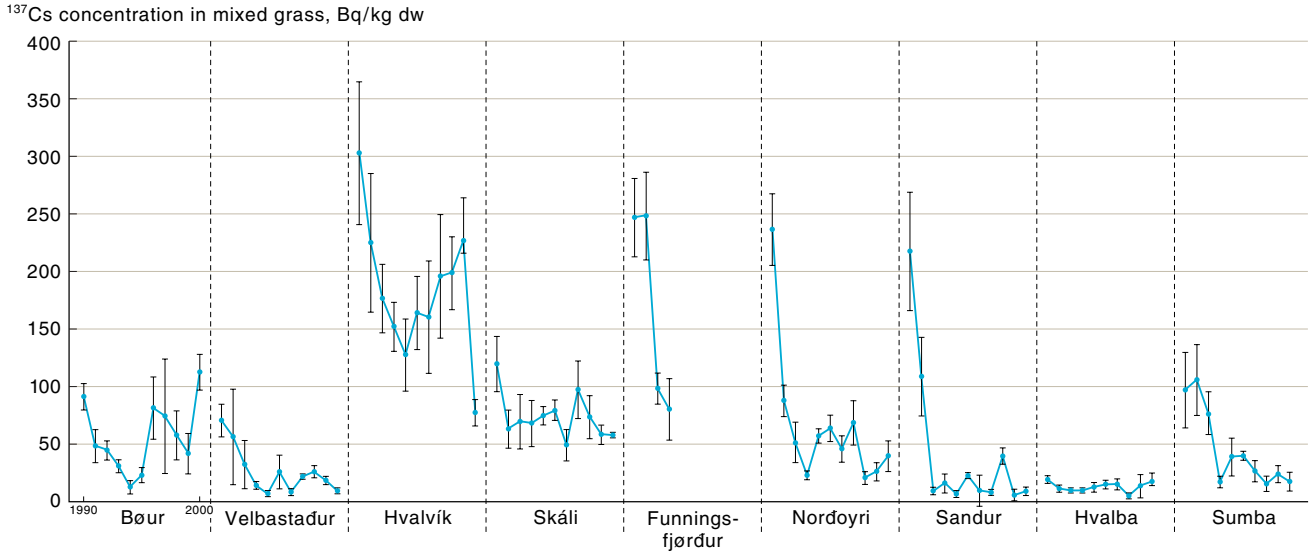

Figure 3·47. Annual average (±SE) ^{137}Cs activity concentrations in mixed grass between 1990 and 2000 (1999 for Hvalba and Sumba) in the Faroe Islands (for locations see Figure 3.40).

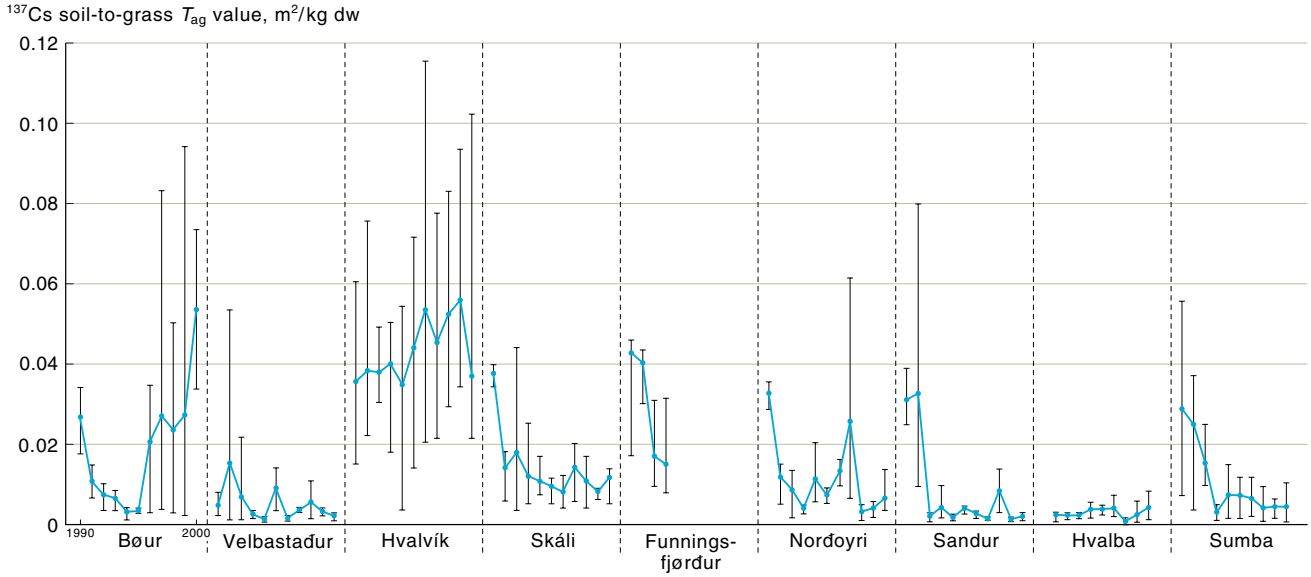

Figure 3·48. Annual average and ranges of soil-to-grass T_{ag} values for ^{137}Cs between 1990 and 2000 in the Faroe Islands (for locations see Figure 3.40).

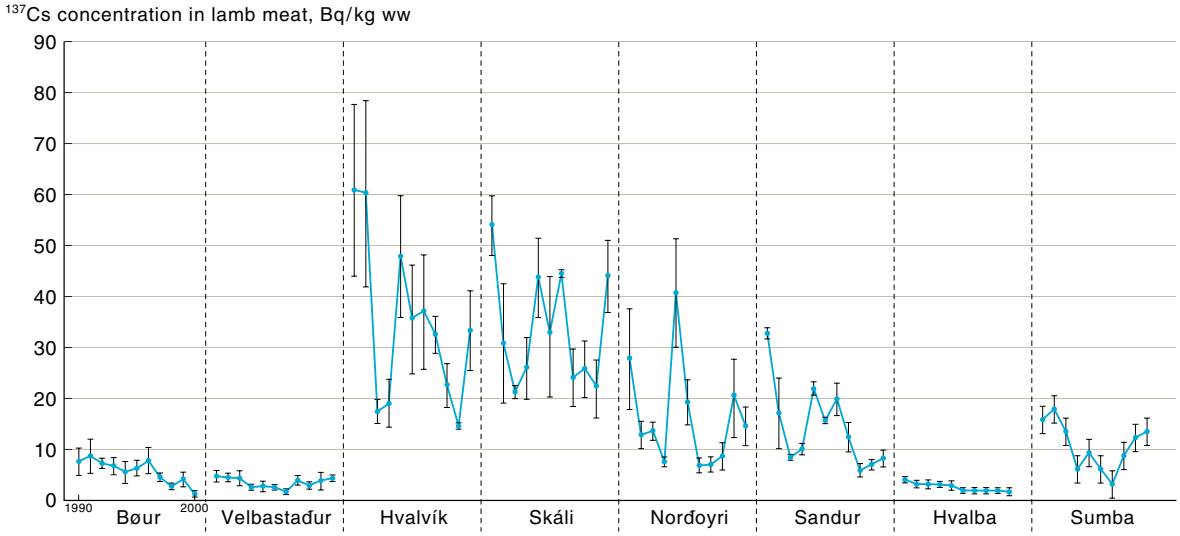

Figure 3·49. Annual average (±SE) ^{137}Cs activity concentrations in lamb meat between 1990 and 2000 (1999 for Hvalba and Sumba) in the Faroe Islands (for locations see Figure 3.40).

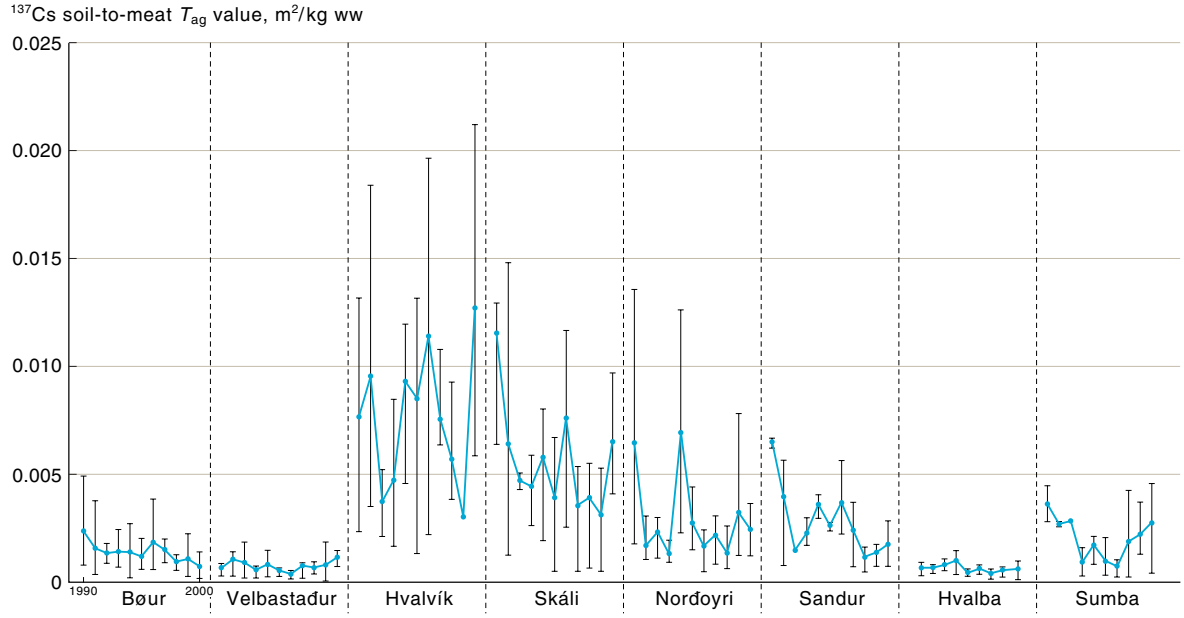

Figure 3·50. T_{ag} values for ^{137}Cs transfer to lamb meat at nine different sites in the Faroe Islands (for locations see Figure 3.40). Yearly averages and ranges 1990-2000.

T_{ag} values over the ten-year period. Some individual pastures showed a similar degree of variation. There was no clear time dependency in T_{ag} values at most sites.

Lamb meat

Figure 3·49 shows ^{137}Cs activity concentrations in lamb meat for 1990 to 1999. Large standard errors reflect large individual variation between animals. The highest values occurred at Hvalvík, Skáli, and Norðoyri.

The T_{ag} values at the nine sites are shown in Figure 3·50. Again, the highest values occurred at Hvalvík, Skáli, and Norðoyri. There is significant variation in T_{ag} values within and between sites.

Effective ecological half-lives

Effective ecological half-lives were derived for ^{137}Cs activity concentrations in grass and meat (Table 3·17). They could only be estimated for some pastures.

Table 3·17. T_{eff} values (yr) based on measurements for 1990 to 2000. r^2 (shown in brackets) from linear regression of the natural logarithm of the measured ^{137}Cs activity concentrations against time. No estimates given when $r^2 < 0.3$.

	Bøur	Velbastaður	Hvalvík	Skáli	Norðoyri	Sandur	Hvalba	Sumba
Grass	-	5.3	–	–	5.3	3.1	–	3.6
	(0.027)	(0.306)	(0.235)	(0.167)	(0.93)	(0.379)	(0.005)	(0.667)
Meat	5.1	–	–	–	–	6.9	8.0	–
	(0.668)	(0.033)	(0.199)	(0.031)	(0.060)	(0.392)	(0.873)	(0.069)

Conclusion

Even though the Faroe Islands cover a small geographical area, there was considerable spatial and temporal variation in the transfer of ^{137}Cs from soil to both grass and lamb meat. Owing to this high variability it is inappropriate to use a single T_{ag} value for either grass pasture or lamb meat. In other countries, there is generally greater variation in the key soil characteristics influencing radiocesium uptake than was measured at these sites. Even higher variability could thus be expected in other countries and country-wide generalizations about transfer are open to considerable error.

3.7.2. Iceland

3.7.2.1. Site description

Iceland is the second largest island in Europe, located in the North Atlantic just south of the Arctic Circle. The total surface area is 103 000 km^2, of which 23 805 km^2 (23%) are vegetated, 11 922 km^2 glaciers, 2757 km^2 lakes, and the remaining 64 538 km^2 (63%) barren. The coastline, including fjords and inlets, is about 4970 km long.

Iceland is the most sparsely populated country in Europe with an average of 2.8 inhabitants per km^2. On 31 December 2000 the number of inhabitants was 283 361.

The Icelandic diet is western European in most respects. Nevertheless it retains some characteristics of a subarctic region, making it unique among European nations. Fish, meat, and milk are traditionally the main foods produced in Iceland. Icelanders consume more fish than any other nation in Europe (73 g/d/cap) and, in general, food of animal origin constitutes a large proportion of the Icelandic diet. During the 1990s, the consumption of lamb meat decreased, beef consumption increased slightly, and the consumption of pork and poultry increased significantly (50 to 100%). This is reflected in agricultural production figures, since most of the products are consumed domestically.

In 2000, there were 466 000 sheep and 72 000 cattle on Iceland. During summer the sheep graze freely on rangelands in the interior, the same applies for a proportion of the 74 000 horses and around 4000 wild reindeer (*Rangifer tarandus*, the original herd imported from Scandinavia between 1771 and 1787) which inhabit the northeast of the country. Although reindeer constitute a minor part of the Icelandic diet, hunting is an increasingly popular sport, which also provides an important source of income for local communities.

Thus, sheep, horses, and reindeer would be affected by contamination of the highland areas of Iceland, whereas lowland contamination would affect cattle, and pig and chicken farming.

Volcanic eruptions are frequent in Iceland, producing lava fields and volcanic ash deposits of various extent. The unstable barren areas of the highlands and the floodplains of glacial rivers act as sources of aeolian material. The parent materials of Icelandic soils are largely of volcanic origin. Icelandic soils are mostly andosols, which are characterized by low cohesion and a high capacity to absorb water (>100% on a dry weight basis). This high water-holding capacity intensifies freezing effects, resulting in solifluction, landslides, needle ice formation, and the formation of hummocks (Arnalds, 1999).

The uneven surface of the rangeland areas and the sparse vegetation can make it difficult to obtain representative deposition estimates by sampling the soils.

The first AMAP assessment (AMAP, 1998) identified Iceland as one of the Arctic areas receiving the most fallout from atmospheric nuclear weapons tests, owing to relatively high precipitation rates compared with much of the rest of the Arctic and subarctic (Wright *et al.*, 1999).

The ^{137}Cs in the Icelandic terrestrial ecosystem originates almost entirely from nuclear weapons tests carried out in the atmosphere until the early-1960s. Fallout was greatest in the mid-1960s. Additional fallout from the Chernobyl accident was relatively small (Pálsson, 1996). This section provides data on radionuclide contamination in Iceland and uses recently acquired data to test the methodology and conclusions of the first AMAP assessment regarding global fallout.

Measurements of fallout from nuclear weapons tests in soil, vegetation, and agricultural products started in Iceland over 40 years ago (Pálsson, 1996). Considerable variability was present in the results, even between adjacent sites, probably due to the mountainous terrain, variable and strong winds, and highly variable levels of precipitation. This variability is particularly noticeable for soils. Early measurements of nuclear fallout were restricted to cultivated lowland areas. The importance of uncultivated rangelands in Icelandic agriculture (e.g., for sheep farming) makes their inclusion desirable for current and future estimates of radionuclide transfer into agricultural products.

Since summer 2000, spatial variation in ^{137}Cs deposition in Iceland has been studied systematically. The objectives of the study are to measure the spatial variation of radiocesium inventories in Icelandic soils and to compare the results with predicted ^{137}Cs soil levels (Sigurgeirsson *et al.*, 2002).

In summer 2000, soil samples were collected to a depth of 25 cm at 14 sites. The sites were located close to meteorological measurement stations so that representative precipitation data were available. Deposition at each site was estimated by assuming the ^{137}Cs activity concentration in precipitation was the same at all sites during any given period. Thus, deposition at each site is estimated by measuring the activity concentration of ^{137}Cs in precipitation at one reference site and then estimating deposition at the other sites by summing the product of precipitation (in m) at the site and ^{137}Cs in precipitation (in Bq/m^3) at the reference site for the period of interest (Pálsson *et al.*, 2002a,b).

A reference station close to Reykjavík (Rjúpnahæð, location 7 on Figures 3·51 and 3·52) was used for predicting fallout in Iceland, and quarterly measurements of fallout radioactivity in precipitation were undertaken regularly by the U.K. Atomic Energy Authority from 1959 to 1982 (Pálsson, 1996). Precipitation data for the reference site were supplied by the Icelandic Meteorological Office (Veðráttan 1959-1983). The reference station data show that 82.9% of the decay-corrected deposition of ^{137}Cs occurred during the first eight years, i.e., 1960 to 1967. The emphasis of the study was thus placed on meteorological stations that were operational during this eight-year period; estimates of deposition for these years were based on 1960 to 1967 precipitation data. The results were subsequently scaled up to cover the entire study

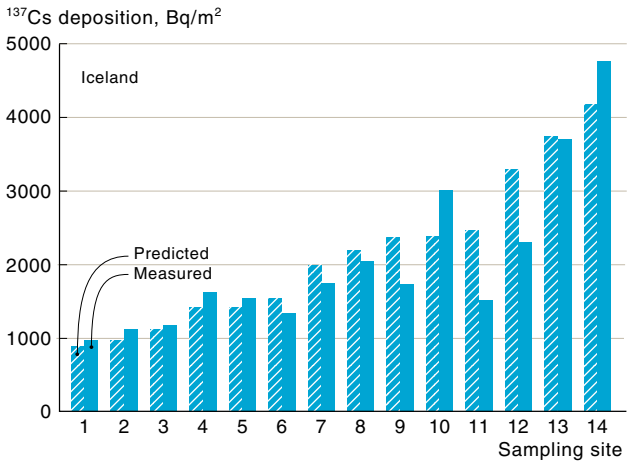

Figure 3·51. Comparison of predicted ^{137}Cs deposition based on precipitation data and measured values at fourteen sites (see Figure 3·52 for locations) close to meteorological stations in Iceland (Pálsson *et al.*, 2002a).

period, 1960 to 1982, assuming in all cases that 82.9% of the deposition had occurred during the first eight years used for calculations. Fallout data prior to 1960 were not included in this study and thus the measured values should be slightly higher than those predicted.

The measured ^{137}Cs content per unit area of soil varied from 900 to 4700 Bq/m^2, with deposition greater in the south of Iceland which receives more precipitation.

There are various ways of estimating the correlation between predicted and measured ^{137}Cs deposition. The method used in the comprehensive AMAP study (Wright *et al.*, 1999) was to force the regression line through the origin and calculate correlation coefficients on that basis. This gives a higher value for the correlation coefficient than for an unbound regression line, but can be justified in that the assumption being tested is that deposition is directly proportional to precipitation. This approach was used in the present study. Figure 3·51 compares measured and predicted deposition at the 14 sites.

The correlation between predicted and measured values for Iceland was much stronger than that reported by Wright *et al.* (1999). The AMAP study was based on 50 samples obtained from Greenland, Norway, and Russia between 1961 and 1985. A line through the origin was fitted to the data using least squares regression and gave an r^2 value of 0.51 based on a coarse precipitation data set and disparate sources of measured ^{137}Cs deposition using different sampling methods. In Iceland, a comparison of predicted and measured values gave a corresponding r^2 value of 0.96. This same value was obtained using both the AMAP method and when prediction was based on average annual precipitation for 1960 to 1967. Some of the improved correlation relative to the AMAP study is probably due to the proximity of meteorological stations, where precipitation has been measured in a consistent manner. Also, soil sampling was conducted by the same team, with a consistent, rigorous methodology over a short period of time. However, the strength of the correlation is surprising considering that dry deposition is not accounted for, although the high precipitation rate in much of Iceland means dry deposition is unlikely to contribute much to the total ^{137}Cs deposition. In addition, lateral transport by erosion would be expected in some Icelandic areas (Arnalds, 1999). A map of predicted ^{137}Cs deposition for Iceland, based on the AMAP methodology is given in Figure 3·52.

Overall, the AMAP methodology has been successful and has the advantage that it is possible to predict the ^{137}Cs deposition at any location in any year since 1960. Allthough fallout did occur prior to 1960, this was at lower levels than during the 1960s.

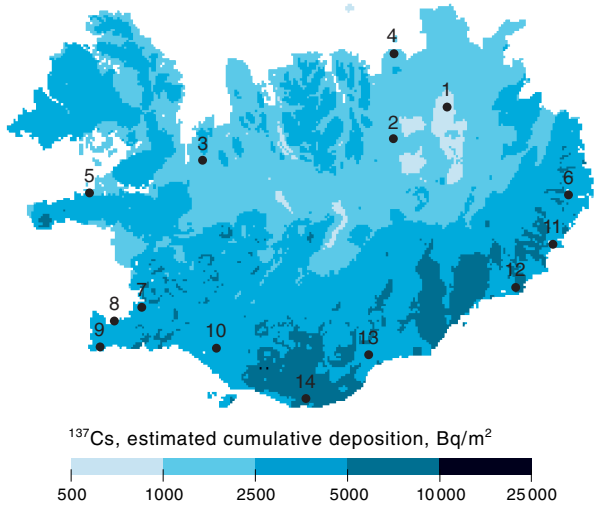

Figure 3·52. Preliminary map of estimated cumulative deposition of ^{137}Cs from atmospheric nuclear weapons tests, decay-corrected to 1995 (AMAP Data Centre). The map is based on a preliminary estimate of the average annual precipitation in Iceland, using a model developed by Crochet (2002) and precipitation data from 1960 to 1990. The conversion to deposition was achieved using a method equivalent to the AMAP method (Pálsson *et al.*, 2002a). Numbers indicate locations of sites represented in Figure 3·51.

3.7.3. Amchitka Island

In November 1971, the project Cannikin was conducted by the U.S. Atomic Energy Agency, now the Department of Energy, at the Amchitka Island underground nuclear test area; this was its largest underground nuclear test, with a yield of about 5 Mt. Preceding Project Cannikin were Projects Long Shot and Milrow; tests of approximately 80 kt and 1 Mt. These three tests represented an estimated 15 to 16% of the total effective yields of all the U.S. underground nuclear tests. In total effective yield, this site is the second largest and the only island underground nuclear test area in the United States. The location is shown in Figure 3·53.

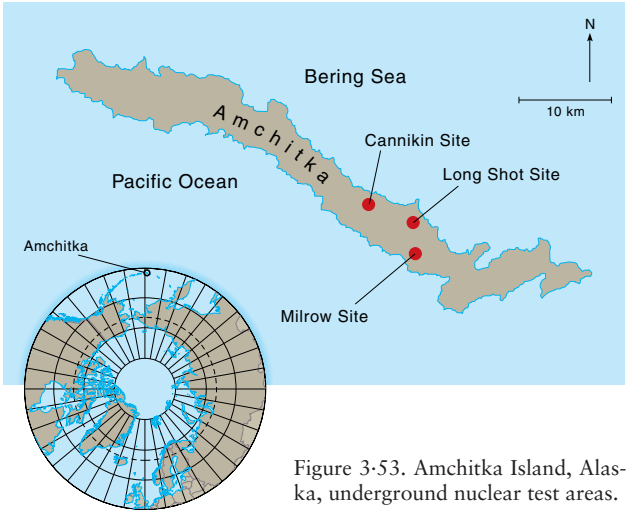

Figure 3·53. Amchitka Island, Alaska, underground nuclear test areas.

3.7.3.1. **Sampling**

Since the late-1970s there has been no marine sampling around Amchitka to assess the status of the anthropogenic radionuclides present or to determine trends. Groundwater contaminated by the three underground nuclear tests is transported toward discharge points on the ocean floor. Conceptual groundwater transport models have shown, based on a range of geohydrological assumptions, that discharge of radionuclides could have started as early as 1975; ten years after the first test.

In 1996, Greenpeace reported that leakage of ^{241}Am and $^{239+240}$Pu had been detected from these underground test sites to the terrestrial and freshwater environment (Miller and Buske, 1996). The marine environment was not specifically addressed in the Greenpeace report. In response, a federal, state, tribal, and non-governmental team conducted a terrestrial and freshwater radiological sampling program in 1997. Additional radiological sampling was conducted in 1998. An assessment of the reported leakage to the freshwater environment was evaluated by assessing tritium (^{3}H) values in surface waters and ^{240}Pu : ^{239}Pu ratios in various sample media (Dasher *et al.*, 2002). Tritium values ranged from 0.41 Bq/L ± 0.11 (2 SD) to 0.74 Bq/L ± 0.126 (2SD) at the surface water sites sampled, including the reported leakage sites. Only at the Long Shot test site, where leakage of radioactive gases to the near surface occurred in 1965, were higher ^{3}H levels of 5.8 Bq/L ± 0.19 (2SD) still observed in 1997; in mud pit #3 (Faller and Farmer, 1998). The mean ^{240}Pu : ^{239}Pu ratio for all Amchitka samples was 0.199 ± 0.014 (1 SD), with values ranging from 0.182 ± 0.0007 (1 SD) to 0.24 ± 0.02 (1 SD).

For the macroalga *Fucus distichus* the mean ^{240}Pu : ^{239}Pu atom ratio of 0.217 ± 0.016 (1 SD) is slightly outside the 95% confidence interval (±2 SD) of the reported global ratio of 0.176 ± 0.014 (1 SD) (Krey *et al.*, 1976). The mean ^{240}Pu : ^{239}Pu atom ratio of 0.216 ± 0.023 (1 SD) for the littoral zone marine sediment samples was consistent with the higher ratio seen for *F. distichus*. Deviations from the global fallout mean ^{240}Pu : ^{239}Pu atom ratio observed in marine algae, sediment, and pooled Amchitka samples may suggest another source of Pu to the marine environment. In an investigation of Bering Sea sediments Hameedi *et al.* (1999) reached similar conclusions. However, uncertainties in analyses and environmental processes must be fully assessed before making conclusions. Further work is needed to determine whether there are any other sources of Pu to the Bering Sea and North Pacific regions besides global fallout.

Results of the 1997 and 1998 sampling based on the measured ^{240}Pu : ^{239}Pu ratios and ^{3}H levels do not provide any evidence for leakage of ^{241}Am or other radionuclides from the underground test shot cavities into the terrestrial or freshwater environments on Amchitka Island (Dasher *et al.*, 2002). In addition, the hydrogeological regime as understood for Amchitka does not provide the physical means to transport transuranics from the test cavities to the reported surface locations.

Clearly, these results do not mean that leakage from the Amchitka underground nuclear tests is not occurring or will not occur into the North Pacific Ocean or the Bering Sea. Hydrogeological modelling predicts leakage can begin initially for ^{3}H from the test sites into the marine water over periods of 20 to 3000 years (Claassen, 1978; Dudley *et al.*, 1977). These periods bridge the various hydrogeological parameter assumptions that can be made. No sampling has been conducted in the marine environment surrounding Amchitka since the late-1970s and, thus, it remains an important area to be addressed.

3.7.3.2. **Geological forces**

Since the underground nuclear tests, dramatic changes have taken place in the field of geosciences and in the understanding of the geological forces acting on the Aleutian Islands, including Amchitka (Eichelberger *et al.*, 2002). In the 1960s, the site was considered geologically stable, showing little evidence of vertical tectonic motion or massive slope failures. Large ongoing horizontal displacements were not considered a possibility.

A paradigm shift occurred in the years following the nuclear tests with the acceptance of the theory of plate tectonics. Amchitka Island is now understood to be a fragment of an island arc crest at the intersection of the subduction of the Pacific Plate beneath the North American Plate (Figure 3·54). Recent field measurements (Eichelberger *et al.*, 2002) indicate that Amchitka is undergoing westward movement of about 1 cm/yr. This sug-

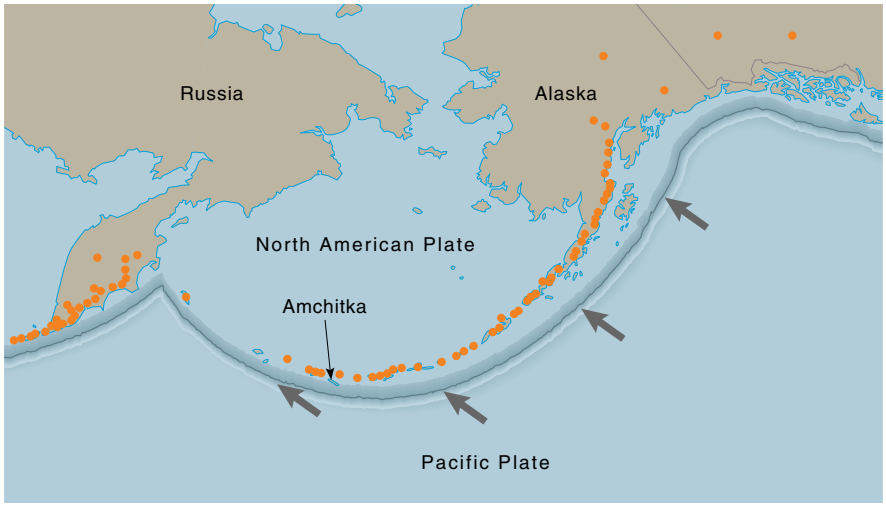

Figure 3·54. The Aleutain Vocanic Arc. Orange circles indicate active volcanoes. Arrows show the motion of the Pacific Plate relative to the North American Plate (Eichelberger *et al.*, 2002).

gests potential major faults in Amchitka Pass and also a strike-slip boundary north of Amchitka that is moving at a rate approximately two-thirds that of the San Andreas fault. The stresses induced would tend to open fractures perpendicular to the island leading into the marine environment.

The acceptance that Amchitka is part of a crustal block rotating clockwise within the fore-arc of an obliquely converging subduction zone raises concerns over possible enhancement of 'fast pathways' for the release of radionuclides from the underground nuclear test sites to the marine environment. Figure 3·55 provides a schematic illustration of the Cannikin underground nuclear test site over the adjacent Teal Creek Fault. A limited survey of the coastline near the Cannikin test area revealed a low density of joints, but did identify apparent fluid transport along these structures in the geological past. A better understanding of the geological forces affecting Amchitka is needed to determine their potential impact on leakage and in designing appropriate long-term monitoring programs.

3.7.3.3. Summary

The knowledge that environmental pathways, potentially enhanced by geological forces, exist on Amchitka Island to transport radionuclides into the nearshore marine environment is a cause for concern. The nearshore environment is commercially and environmentally important. An independently conducted radiological assessment is required to protect the indigenous peoples of the region (the Aleut), U.S. and Russian citizens, and people living off the vast commercial fisheries of the region (State of Alaska and Aleutian/Pribilof Island Association, 2001). The recent radiological assessment of the French nuclear test sites in the South Pacific provides an example of what is needed (IAEA, 1998b).

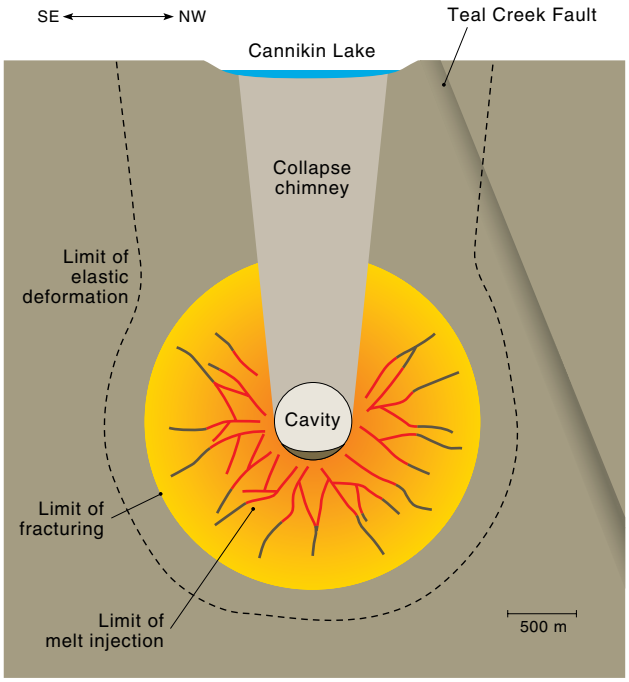

Figure 3·55. Schematic cross-section showing the effects of the Cannikin underground test, based on work by R. Laczniak and colleagues at the Nevada Test Site (Eichelberger *et al.*, 2002).

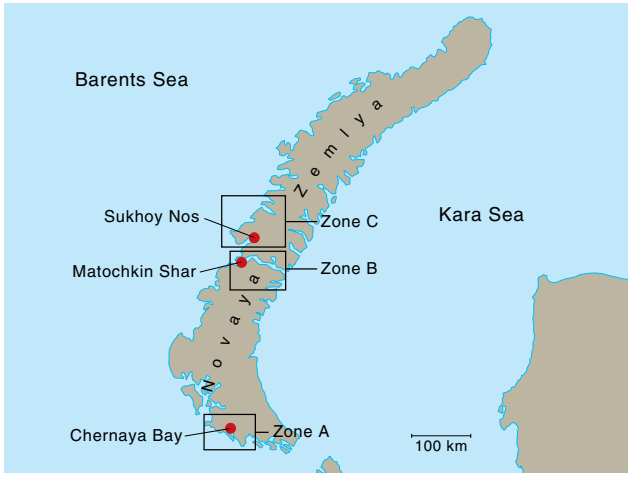

Figure 3·56. The central test site of the Russian Federation at Novaya Zemlya.

3.7.4. Novaya Zemlya

Since the first AMAP assessment, two further sources of information on Novaya Zemlya have been published: a report by Ivanov *et al.* (1997) and a major review of the Novaya Zemlya test site by Logachev (2000). The following information is based on these sources.

Novaya Zemlya was one of two major nuclear test areas for the former Soviet Union. Altogether, 130 nuclear tests had been carried out at the north test site by 25 October 1990 (the date the effective moratorium was announced). Nuclear tests were conducted in three areas (Figure 3·56):

Zone A. A series of nuclear tests were conducted from 1955 to 1962. The first underwater nuclear test occurred on 21 September 1955, the date of commencement for the site. The explosion comprised the experimental blasting of a T-5 torpedo with a warhead of about 3.5 kt at a depth of about 12 m. Another underwater test was conducted later. Six underground nuclear tests were conducted in vertical blast holes between 1972 and 1975, and there was a near-surface nuclear explosion on 7 September 1957.

Zone B. Thirty-three underground nuclear tests were conducted in tunnels within mountains between 1964 and 1990.

Zone C. A series of elevated and atmospheric nuclear tests, were undertaken commencing 24 September 1957 and ending 25 December 1962. The largest nuclear test explosion ever, a 58 Mt atmospheric blast, occurred in this zone on 30 October 1961.

3.7.4.1. Soil contamination

The data used by Ivanov *et al.* (1997) and Logachev (2000) are mainly from surveys by the E.K. Fedorov Institute of Applied Geophysics (in 1976 to 1978 and 1990) and the V.G. Khlopin Radium Institute (in 1992).

The ^{137}Cs contamination density outside the test areas ranged from 1.7 to 5.6 kBq/m^2 (45 to 150 mCi/km^2) in the 1976 to 1978 survey, with a mean of 3.4 kBq/m^2 (91 mCi/km^2). The contamination density in 1990 ranged from 1.5 to 6.7 kBq/m^2 (40 to 180 mCi/km^2) with a mean of 3.3 kBq/m^2 (90 mCi/km^2).

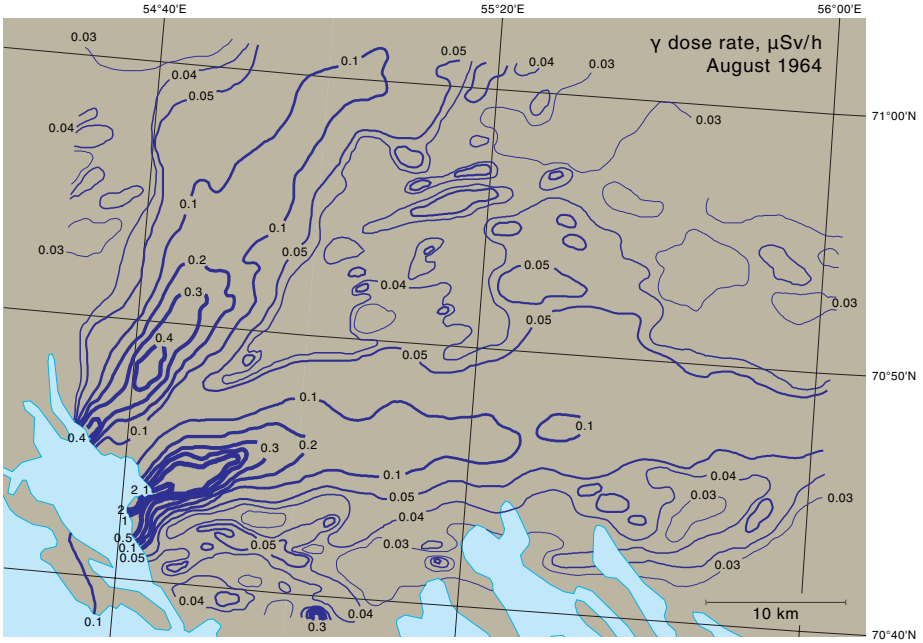

Figure 3·57. Radiation situation in the area around Chernaya Inlet in August 1964 (Ivanov *et al.*, 1997; Logachev, 2000). Contour lines show γ dose rates 1 m above ground.

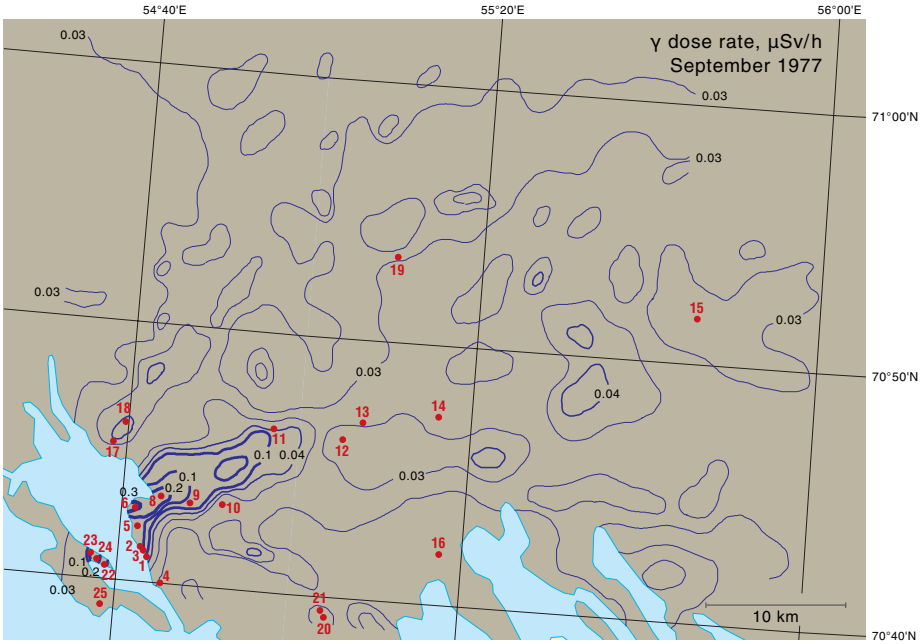

Figure 3·58. The radiation situation in the area of the Chernaya Inlet in September 1977 (Ivanov *et al.*, 1997). Contour lines show γ dose rates 1 m above ground. Red dots and associated numbers indicate sampling sites referred to in Table 3·19.

The surveys placed particular emphasis on studying radioactive traces due to deposition of radioactive products after the nuclear explosions. The zones of increased radioactive contamination detected during the survey include:

- the area around Chernaya Bay (Zone A);
- Sukhoy Nos Peninsula (Zone C);
- the area around Bashmachnaya Inlet (Zone B); and
- the tidal area of the Matochkin Shar Strait (Zone B).

The present assessment contains improved data for the first two sites; data for the others are available in the first AMAP assessment (AMAP, 1998).

3.7.4.2. **Area around Chernaya Bay**

Of the 90 atmospheric nuclear weapons tests conducted on the archipelago between 1955 and 1962 at least four were in contact with the underlying surface, and all occurred in the area of Chernaya Bay.

The epicenter of one of the explosions conducted in 1957 was about 100 m from the shore. The trace of the radioactive fallout from this explosion, as followed in the first three days after the explosion to a distance of 1500 km from the epicenter, covered part of the South Island and the Yamal, Gydansky, and Taymir Peninsulas. In 1964, another aerial survey was carried out over the extent of the trace covering 70 km along the axis and the results are shown in Figure 3·57. Analyses revealed the presence of the following radionuclides: ^{137}Cs, ^{144}Ce, ^{125}Sb, ^{106}Ru, ^{90}Sr, ^{60}Co, ^{152}Eu, ^{154}Eu, and ^{155}Eu. The epicenter of this near-surface explosion is the most contaminated zone on the archipelago. The measurements in this zone were repeated in 1976 to 1977. Results of the 1977 aerial survey and the location of soil sampling points are shown in Figure 3·58. The level of gamma radiation near the 1957 explosion funnel was as high as 5 μSv/h (the transverse size of the zone is several tens of meters). The distribution of the radionuclides was investigated using depth samples collected from two

Table 3·18. Radionuclide profiles for soils (Bq/kg (10^{-9} Ci/kg)) collected in 1977 from the epicenter of the near-surface nuclear explosion conducted in 1957 in the area of Chernaya Inlet, Novaya Zemlya (Ivanov *et al.*, 1997; Logachev, 2000).

Depth, cm	^{137}Cs	^{60}Co	^{152}Eu
Site 1			
0-1	67000 (1800)	52000 (1400)	Trace
1-2	70000 (1900)	44000 (1200)	Trace
2-3	44000 (1200)	33000 (900)	17000 (460)
3-4	19000 (520)	22000 (590)	18000 (480)
4-5	12000 (320)	22000 (590)	22000 (590)
10-15	2300 (61)	27000 (720)	54000 (1450)
20-25	–	8700 (235)	17000 (460)
30-35	70 (1.9)	13000 (360)	2500 (67)
40-45	330 (8.9)	190 (5.1)	440 (12)
50-55	230 (6.1)	–	–
60-65	160 (4.2)	–	–
Site 2			
0-2	41000 (1100)	32000 (870)	4400 (120)
2-4	46000 (1250)	26000 (710)	6700 (180)
4-6	50000 (1350)	26000 (715)	7000 (190)
6-8	27000 (725)	16000 (425)	8000 (215)
8-10	23000 (610)	12000 (325)	7800 (210)
10-15	16000 (430)	13000 (360)	6800 (185)
20-25	13000 (360)	4100 (110)	8500 (230)
30-35	190 (5)	–	1900 (50)
40-45	110 (3)	120 (3.3)	250 (6.8)
50-55	–	–	–
60-65	46000 (3.8)	–	–

pits; located 10 m southwest of the funnel crest (site 1, Figure 3·58) and 100 m north of the funnel (site 2, Figure 3·58). Data on the local distribution of γ-emitters for these samples are shown in Table 3·18.

The extent of the radioactive trace is decreasing with time. The area of the trace exceeding 0.1 μSv/h decreased from 133 km^2 in 1964 to 10 km^2 in 1977 (see Figures 3·58 and 3·59). Moreover, the trace has become heterogeneous in character, due to radioactive decay of short-lived γ-emitters.

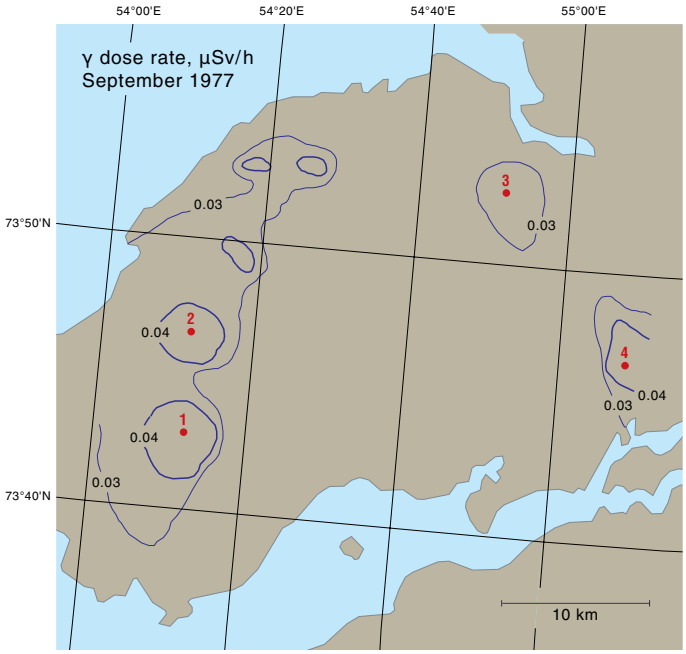

Figure 3·59. The radiation situation on the Sukhoy Nos Peninsula in September 1977. The radiation dose rate was in excess of natural background (Logachev, 2000). Red dots and associated numbers indicate contaminated sites referred to in the text.

In addition to the trace resulting from the near-surface explosion of 1957, traces of radioactive contamination from an above-water explosion in 1961 and an underwater explosion in 1955 can be seen in Chernaya Bay. Data on the density of contamination by radionuclides in these traces obtained in 1977 are shown in Table 3·19.

3.7.4.3. Sukhoy Nos Peninsula

Most of the atmospheric nuclear weapons tests were conducted above the Sukhoy Nos Peninsula to the north of the Matochkin Shar strait. Several areas of contamination resulted from these tests. A gamma survey and soil sampling campaign carried out in 1977 showed four contaminated areas (see Figure 3·59):

- the western trace is an area (0.5 km^2) 3 km east of the Fedorov Mountain (Site 1);
- the central trace (0.3 km^2) is in the center of the peninsula (Site 2);
- the northern trace (0.3 km^2) is 10 km from the Tsivolki Cape (Site 3); and
- the eastern trace (0.4 km^2) is 12 km northeast of the Klochkovsky Peninsula (Site 4).

Table 3·20 shows the man-made radionuclides in soil from the eastern trace. These data suggest that contamination after the elevated air explosions, during which mineral (soil) particles were generally not entrained in the plume, was primarily due to radionuclides formed as a result of interaction between the neutron flux of the penetrating radiation arising from the nuclear explosion and the soil.

3.7.5. Thule

In January 1968, a B-52 aircraft carrying four nuclear weapons crashed onto the sea ice ~11 km from Thule Air Base in northwest Greenland (see Section 2.4). As a consequence, the benthic marine environment of Bylot Sound (180 to 230 m deep) became contaminated by ^{239,240}Pu. The site was revisited in August 1997, 29 years after the accident. Sections 3.7.5.1. to 3.7.5.4. are extracted from Dahlgaard *et al.* (2001).

3.7.5.1. Plutonium in water and seaweed

Activity concentrations of ^{239,240}Pu in *Fucus distichus* (a brown alga) around Thule and 750 km to the south near Uummannaq ranged from 0.15 to 1.14 Bq/kg dw. The source of most of this Pu is global fallout – except possibly the highest value seen in a single sample near the accident site. With the exception of a near-bottom water sample taken at the point of impact containing 30 mBq/m^3, no clear effect of the accident was seen in any of the water samples. Of this elevated level, 42% was particulate indicating that resuspended sediments containing accident-related Pu are an important source. The general level of ^{239,240}Pu within Bylot Sound was 5 to 10 mBq/m^3 in unfiltered surface water, which is regarded as global fallout background. These data indicate that Pu from contaminated sediments is not transported into surface waters in significant quantities.

Table 3·19. Radionuclide profiles for soils (kBq/m^2 (mCi/km^2)) collected in 1977 from within the traces of nuclear explosions in the Chernaya Bay area (Ivanov *et al.*, 1997).

	No. of site in Fig 3·58	Sample No.	Depth, cm	^{137}Cs	^{90}Sr	^{60}Co	^{152}Eu
Epicenter zone	1		0-5	4800 (1.3×10^5)	–	2800 (7.5×10^4)	780 (2.1×10^4)
and near-surface	2		0-5	4100 (1.1×10^5)	–	2400 (6.4×10^4)	280 (7600)
explosion trace	3		0-3	26 (690)	–	160 (4200)	230 (6300)
1957			3-6	20 (540)	–	160 (4200)	190 (5200)
			6-10	7.8 (210)	–	130 (3500)	200 (5400)
	4		0-5	14 (380)	–	–	–
	5		0-5	85 (2300)	–	59 (1600)	–
	6	1	0-5	160 (4400)	100 (2700)	190 (5200)	–
		2	0-2	21 (560)	–	24 (650)	5.4 (140)
			2-4	5.2 (140)	–	4.8 (130)	0
			4-6	–	–	–	–
		3	0-2	37 (1000)	–	78 (2100)	8.1 (220)
			2-4	–	–	–	–
	7		0-5	74 (2000)	–	89 (2400)	–
	8		0-5	11 (310)	4.8 (130)	10 (280)	–
	9		0-5	7.8 (210)	–	5.9 (160)	–
	10		0-5	30 (800)	–	24 (660)	–
	11		0-5	2.6 (69)	–	–	–
	12		0-2	11 (310)	–	10 (270)	–
			2-4	3.5 (94)	–	3.3 (88)	0
			4-6	–	–	–	0
	13		0-5	5.7 (155)	0.93 (25)	2.7 (72)	–
	14		0-5	9.3 (250)	–	5.2 (140)	–
	15		0-5	3.9 (106)	3.7 (100)	–	–
	16		0-5	2.6 (69)	–	–	–
Trace of above-	17		0-5	48 (1300)	70 (1900)	–	–
water explosion	18		0-5	5.2 (140)	–	–	–
1961	19		0-5	2.3 (61)	–	–	–
Spot resulting from	20	1	0-5	–	–	120 (3300)	220 (5900)
near-surface			10-15	–	–	56 (1500)	110 (2900)
explosion			20-25	–	–	12 (330)	23 (630)
			30-35	–	–	–	–
		2	0-1	0.96 (26)	–	4.4 (120)	7.4 (200)
			1-2	–	–	13 (350)	23 (610)
			2-3	–	–	28 (770)	46 (1250)
			3-5	–	–	63 (1700)	96 (2600)
	21		0-5	2.6 (69)	2.9 (79)	–	4.4 (120)
Trace of the	22*		0-3	300 (8000)	–	48 (1300)	–
the underwater			3-6	160 (4400)	–	34 (930)	–
explosion 1955			6-10	44 (1200)	–	10 (270)	–
	23		0-2	410 (11000)	–	56 (1500)	–
			2-4	160 (4300)	–	29 (780)	–
			4-6	26 (700)	–	7 (190)	–
			6-8	41 (1100)	–	9.8 (265)	–
			8-10	59 (1600)	–	–	–
	24		0-5	44 (1200)	3.7 (100)	8.5 (230)	–

* no vegetation on the sampling location.

Table 3·20. Radionuclides in soil (kBq/m^2 (mCi/km^2)) from the eastern part of the experimental zone (Site 4) on the Sukhoy Nos Peninsula in early-1993 (Ivanov *et al.*, 1997; Logachev, 2000).

Sample No.	Depth, cm	^{137}Cs	^{60}Co	^{152}Eu
#1	0-1	1.4 (37)	Trace	2.7 (72)
	1-2	1.4 (37)	Trace	3.7 (100)
	2-3	0.85 (23)	Trace	4.4 (120)
	3-4	Trace	Trace	5.6 (150)
	4-5	Trace	Trace	5.9 (160)
#2	0-5	2.6 (70)	19 (520)	36 (980)
	10-15	Trace	Trace	23 (610)
	20-25	Trace	Trace	Trace
	30-35	Trace	Trace	Trace
#3	0-5	2.2 (60)	15 (400)	14 (380)
#4	0-5	3.5 (95)	Trace	20 (550)
#5	0-5	4.6 (125)	12 (330)	31 (830)

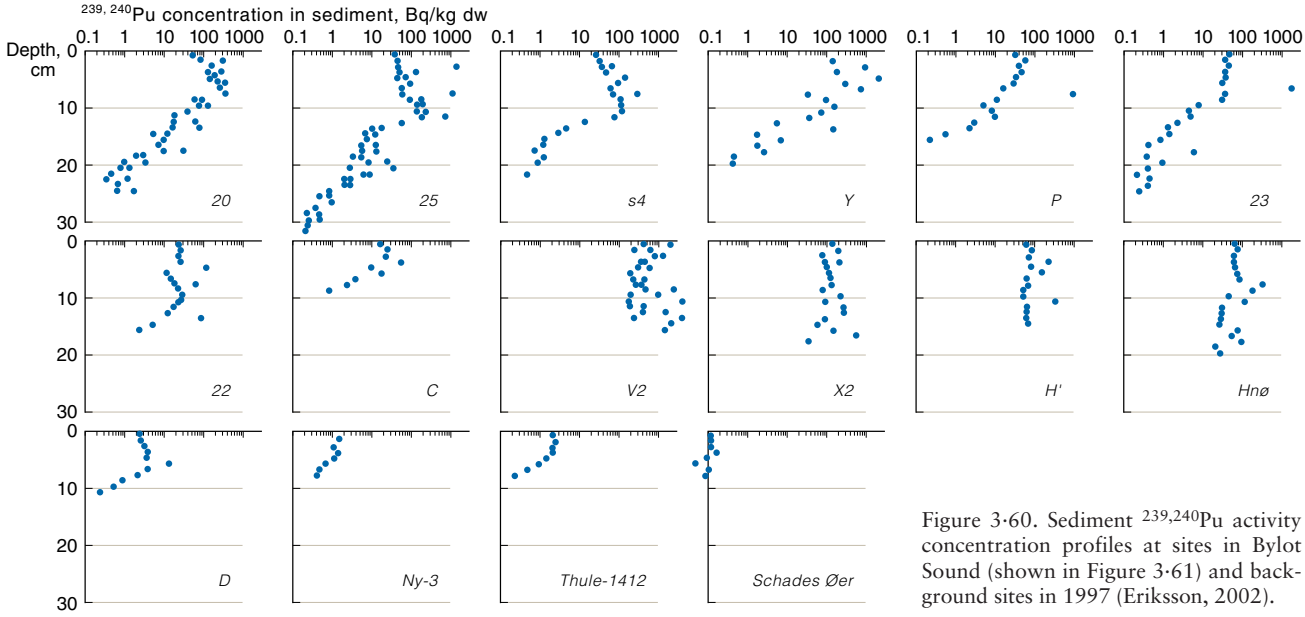

Figure 3·60. Sediment ^{239,240}Pu activity concentration profiles at sites in Bylot Sound (shown in Figure 3·61) and background sites in 1997 (Eriksson, 2002).

3.7.5.2. **Sediments**

Figure 3·60 shows Pu depth profiles for contaminated sediment cores from Bylot Sound, plus background cores taken outside Bylot Sound (Ny-3, Thule-1412, and Schades Øer; the latter about 750 km southeast of Thule). In all cases, the Pu appears well mixed throughout the upper 3 to 5 cm layer (note the logarithmic concentration axis in Figure 3·60). Despite logarithmic axes, the large variation in Pu concentrations is clear. This is caused by 'hot particles'. In a recent thesis Eriksson (2002) stated that these 'hot particles' hold more Pu than previously anticipated (see Section 2.4).

Plutonium concentrations in surface (0 to 3 cm) sediments are shown in Figure 3·61. The highest concentrations are centered on the accident site, with a fairly even distribution in the remaining deep part of Bylot Sound, and almost fallout background concentrations outside Bylot Sound. The water depths at the accident site – close to location V2 – are 180 to 230 m. The two assumed background sites outside Bylot Sound, Ny-3 and Thule-1412, have depths of 500 and 640 m. A surface (0 to 3 cm) ^{239,240}Pu activity concentration of 0.12 Bq/kg dw occurred 750 km further south near Schades Øer. Surface concentrations outside Bylot Sound (at Thule-1412 and Ny-3) are an order of magnitude higher. It is unclear whether this reflects accident Pu or a natural perturbation caused by differences in sedimentological parameters. At other Arctic marine locations similar levels of Pu have been attributed to global fallout (see Section 3.3.4).

3.7.5.3. **Benthic biota**

Plutonium concentrations in biota have been compared with concentrations in surface (0 to 3 cm) sediments (Figure 3·61) to give 'concentration ratios' (Table 3·21). Although the biota live within or on the sediments the concentration ratios indicate that the bioavailability of the weapons Pu is low. Most of the concentration ratios fall within the range 0.01 to 0.1, i.e., Pu concentrations in benthic biota are around one to two orders of magnitude lower than in surface sediments. Furthermore, a significant proportion is probably not metabolized but is bound to particles within the gut and adheres to the surface structure of the animals. One single bivalve sample had a much higher level, which was probably due to a 'hot particle'.

3.7.5.4. **Isotope ratios**

A number of the sediment samples were analyzed for ^{240}Pu : ^{239}Pu atom ratios by High Resolution Inductively Coupled Plasma/Mass Spectrometry. These had ratios in the range 0.027 to 0.057. The calculated uncertainties for most of the samples were 2 to 10%. The samples with highest activity – which have been identified as containing 'hot particles' – show significant variation in the ^{240}Pu : ^{239}Pu atom ratios, i.e., there is a variation in Pu isotope ratios in the Thule debris significantly above measurement error. This supports the conclusion by Mitchell *et al.* (1997) that the Thule Pu originates from at least two sources of different quality. Plutonium concentrations in the samples for this study were dominated by the Thule weapons accident. Therefore, the higher

Table 3·21. Plutonium (^{239,240}Pu) concentration ratios (based on values in Bq/kg dw) for benthic biota and surface (0 to 3 cm) sediments (Dahlgaard *et al.*, 2001).

		Mean	SD	n
Mollusks	Bivalves	0.025	0.024	13
	Macoma calcarea	37*		1
	Snails	0.0033	0.0018	9
	Squid, *Rossia* sp.	0.00036		1
Echinoderms	Starfish	0.0094	0.0139	9
	Brittle stars	0.013	0.016	4
	Feather stars	0.0070	0.0060	4
	Sea urchins	0.12	0.16	4
	Sea cucumber	0.0080	0.0083	4
Crustaceans	Shrimp	0.0048	0.0088	4
	Various	0.038	0.039	4
Annelids	*Pectinaria* sp.	0.068	0.052	4
	Various	0.023	0.033	10
	Tube	0.28	0.29	6
Fish	*Liparis* sp.	0.00035		1

* outlier, probably caused by 'hot particle'.

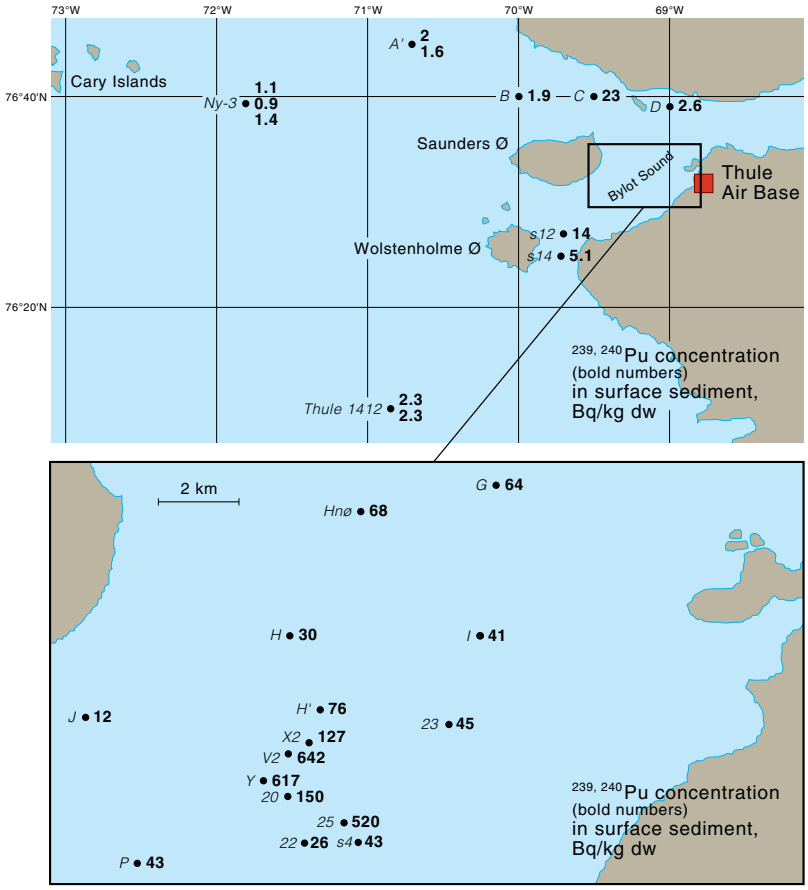

Figure 3·61. Activity concentrations of ^{239,240}Pu in surface sediments (0 to 3 cm layer) near Thule in 1997. Location names are shown in italics, concentrations in bold. The point of impact was on the sea ice (180 m water depth) at the location marked V2 (Eriksson, 2002).

^{240}Pu : ^{239}Pu atom ratio observed in global fallout, approximately 0.18, will not affect these results. Any influence of the higher ^{240}Pu : ^{239}Pu atom ratios in Sellafield discharges, up to around 0.25, is even more unlikely as the Sellafield-derived Pu concentration in the Thule area is accepted to be less than global fallout.

Average isotope ratios for ^{240}Pu : ^{239}Pu atom ratios, and ^{238}Pu : ^{239,240}Pu and ^{241}Am : ^{239,240}Pu activity ratios for sediment samples containing >20 Bq ^{239,240}Pu/kg, i.e., at least an order of magnitude above the fallout background, are given in Table 3·22. The reference date is the sampling date, 1997. By comparing the ^{241}Am : ^{239,240}Pu activity ratios for sediments (Table 3·22) with those for benthic biota (Table 3·23), it is evident that some biota appear to have a higher uptake of americium (Am) than Pu. This appears to be the case for mollusks – bivalves as well as snails – and for some echinoderms, namely brittle stars (Ophiuroidea) and starfish (Asteroidea), but not sea urchins (Echinoidea) or sea cucumber (Holothurioidea). This greater affinity for Am than Pu is not new. The International Atomic Energy Authority reported higher concentration ratios for Am than Pu in mollusks (IAEA, 1985).

Table 3·22. Isotope ratios in Thule sediment samples with >20 Bq ^{239,240}Pu/kg in August 1997 (Dahlgaard *et al.*, 2001).

		Mean	SD, %	n
^{240}Pu : ^{239}Pu	atom ratio	0.045	15	30
^{238}Pu : ^{239,240}Pu	activity ratio	0.014	53	223
^{241}Am : ^{239,240}Pu	activity ratio	0.13	61	114

Table 3·23. ^{241}Am : ^{239,240}Pu activity ratios in sediment-dwelling benthic biota, Thule 1997 (Dahlgaard *et al.*, 2001).

		Mean	SD, %	n
Benthos	All	0.39	76	84
Mollusks	All	0.63	62	24
Polychaetes	All	0.28	77	16
Crustaceans	All	0.22	36	5
Echinoderms	Brittle stars	0.55	24	4
	Starfish	0.41	34	11
	Sea urchins	0.17	8	5
	Sea cucumber	0.13	67	2

3.8. Summary

Since the first AMAP assessment, monitoring of various man-made radionuclides in the Arctic environment has continued to a variable degree. In general, concentrations of radionuclides derived from global fallout, from the Chernobyl accident, and from earlier discharges from European reprocessing plants are slowly decreasing, as expected. This is especially evident for ^{90}Sr, where global fallout is still the dominant source. It is also the case for ^{137}Cs, although the contribution from the Chernobyl accident and reprocessing discharges during the 1970s and 1980s has added significantly to the fallout level and still constitutes a source to the Arctic marine environment owing to remobilization and relocation processes. More unexpectedly, evidence is mounting that the Pu being remobilized from Irish Sea sediments is now responsible for a major proportion of the Pu contamination in the Norwegian and Barents Seas.

Data for the Faroe Islands and Iceland were not adequately dealt with in the first AMAP assessment.They have therefore been addressed in significantly greater detail in the present assessment.

Chapter 4

Assessment of Human Exposure

The exposure of different Arctic populations to anthropogenic radionuclides was addressed during the first AMAP assessment (AMAP, 1998). However, several issues have since arisen which together justify further assessment: 1) data have become available for the Faroe Islands, 2) the Canadian population group selected to represent Arctic caribou herders has been criticized, 3) more detailed information has become available for some population groups in northwest Russia; and 4) owing to the increased releases of ^{99}Tc and ^{129}I from Sellafield and ^{129}I from Cap de la Hague, interest in the resulting doses has increased. No estimates of the uncertainties associated with the dose estimates are given, as the information needed for this was not available.

4.1. Atmospheric sources

The first AMAP assessment concluded that: *'The vulnerability of Arctic terrestrial ecosystems results in a five-fold higher exposure to radioactive contamination compared to that in temperate areas'*. Many post-Chernobyl studies have demonstrated that the highest exposures do not necessarily occur in the most contaminated areas, especially in the mid- to long-term after an accident. The reasons for this vary but can depend, for instance, on variable plant uptake from different soil types or on the application of countermeasures. An example of the effects of countermeasures is shown in Figure 4·1. This shows the dose contribution from global fallout and Chernobyl fallout to two Saami populations in mid-Norway and to Russians living in the Novozybkov district. The Chernobyl fallout in both areas was high and required the application of extensive countermeasures. The effect was to dramatically reduce the ratio between the wholebody ^{137}Cs content of people and ^{137}Cs deposition in areas where countermeasures were applied, compared to areas in which they were not.

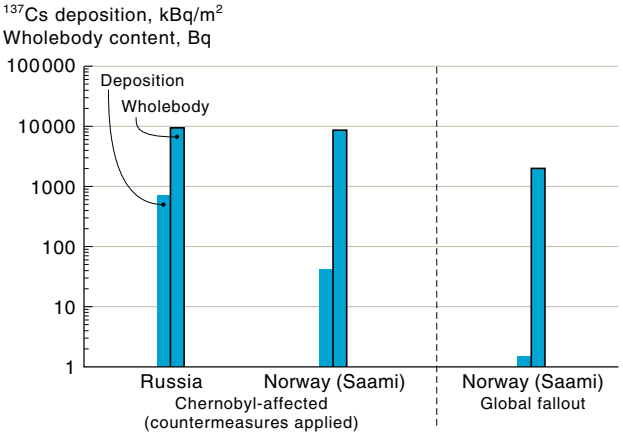

Figure 4·1. Wholebody content of ^{137}Cs for population groups in areas of different ^{137}Cs deposition in 1996 (Strand *et al.*, 2002).

Estimates of doses to the public, based on measurements or model predictions, frequently generalize variations in environmental conditions, either owing to the limited availability of data or to an inadequately detailed knowledge of conditions in the environment that influence exposure. Such generalizations mask considerable variability in the distribution of dose, both in space and time, even if the rate of input of radionuclides to the environment is essentially uniform. While this is of little consequence to the calculation of collective dose, variations in individual doses resulting from variations in vulnerability can be masked and locations and populations receiving comparatively high doses may not be identified. Allowing for these variations in the calculation and prediction of individual dose enables improved insights into the doses that would occur in the event of accidental release. This is particularly valuable for the Arctic because of the comparatively heavy reliance on locally produced foods and semi-natural foods in some populations.

4.2. Faroe Islands

Sufficient data are now available for the Faroe Islands to enable a similar analysis to that carried for the other Arctic countries in the first AMAP assessment (AMAP, 1998).

4.2.1. Food consumption

There have been two extensive nutritional investigations in the Faroe Islands; the first in 1936 (Knudsen, 1940) and the second in 1981 to 1982 (Vestergaard and Zachariassen, 1987; see Table 4·1). Most of the mutton is lamb meat; about 18 kg/yr/cap according to unpublished information (Joensen pers. comm., 2002). Reliable food consumption rates are not available for the Faroe Islands as much of the food is acquired privately, particularly mutton and fish (mainly cod and haddock).

It is likely that the relative proportions of the different food groups have changed over time. Milk has been produced locally for the last 15 to 20 years but most other dairy products are imported, mainly from Denmark. It is also likely that the relative proportions of the different food groups vary across the country, but the available data are insufficient to confirm this.

Table 4·1. Annual mean consumption of foodstuffs (kg/yr/cap) in the Faroe Islands, 1981 to 1982 (Vestergaard and Zachariassen, 1987).

Dairy products	Faroese mutton	Marine fish	Potatoes	Grain products	Vegetables	Whale meat	Whale blubber
142	25	26	70	78	12	4.4	2.6

4.2.2. Dose estimation

Internal doses resulting from the dietary intake of ^{137}Cs in the Faroe Islands since 1950 were calculated on the basis of activity concentrations in foodstuffs and consumption rates (Table 4·1). Activity concentrations in milk were used to represent dairy products, those in lamb to represent mutton, and those in white bread to represent grain. Activity concentrations of ^{137}Cs in whale meat and vegetables were not available. The calculation method was similar to that used in the first AMAP assessment. Annual values based on actual ^{137}Cs measurements were collated to yield 5-year means since the beginning of the 1960s. Owing to gaps in the data set, values for 1950 to 1960 were obtained by linear interpolation. Estimates of doses since 2000 were made using an effective ecological half-life (T_{eff}; Box 3.1) for ^{137}Cs of 10 yr. The modelled integrated calculated dose for the average Faroese population through the nuclear age was 3.5 mSv, which is consistent with the estimated value of 3.3 mSv in the first AMAP assessment. The foodstuffs contributing the major part of the ^{137}Cs dose (approximately 60%) were milk and lamb.

4.3. Canadian Arctic

Some reservations were expressed about the effective individual internal dose commitments due to ^{137}Cs intake calculated in the first AMAP assessment. These concerned the high rates of caribou meat consumption assumed for the Canadian selected (Gwich'in) population group, which appeared much higher than for selected high consumption groups in other Arctic countries. Since then, new dietary information for Canadian northern population groups has become available (Berti *et al.*, 1998; Kuhnlein *et al.*, 2000; Van Oostdam *et al.*, 1999). This indicates that the caribou consumption estimate applied to the Gwich'in selected group in the first AMAP assessment pertains to an extreme (high consumption) group in the population, and does not represent an 'average' consumption estimate for relatively large population groups, as was the case for the selected population groups for other Arctic countries. The Canadian selected group described in the first AMAP assessment was therefore not directly comparable with the selected groups for other Arctic countries. Consequently, the Canadian ^{137}Cs internal dose was re-evaluated in the present assessment using the new dietary data. The outcome is described in the rest of this section (Figure 4·2).

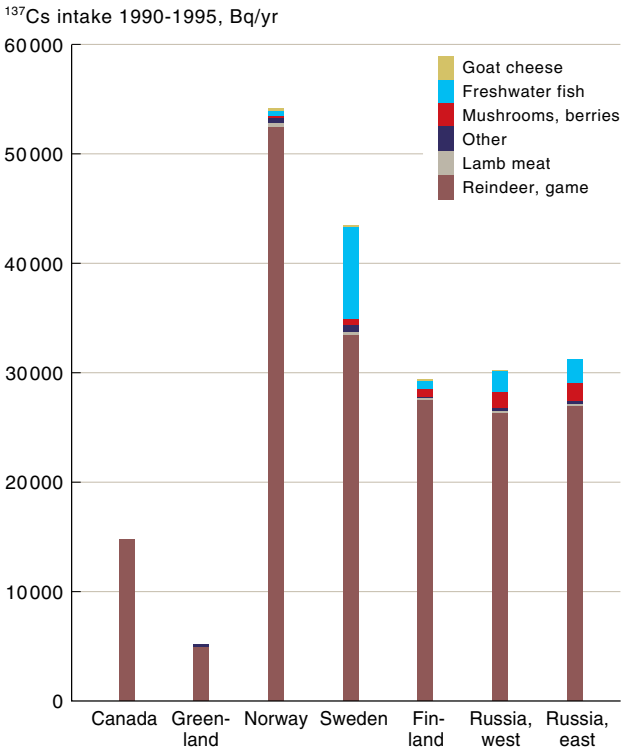

Figure 4·2. Intakes by the selected groups in the period 1990 to 1995 (AMAP Data Centre).

Berti *et al.* (1998) report on a dietary survey performed in five regions (16 communities) of Denendeh in 1994. During the study 1012 individuals completed a 24-hr dietary recall. Results were subdivided according to sex, age (20 to 40 yr, and 40+ yr), and (five) regions. The Gwich'in and Sahtú are groups with a high consumption of caribou meat (Table 4·2).

Table 4·2. Caribou consumption (g/d/cap) for the Gwich'in and Sahtú (three communities surveyed for each group) averaged over four seasons (Berti *et al.*, 1998).

	Female (mean ± SD)		Male (mean ± SD)	
Gwich'in	20-40 yr (n=19,32)*	40+ yr (n=8,35)	20-40 yr (n=17,29)	40+ yr (n=22,33)
Fresh meat	122 ± 73	178 ± 113	163 ± 81	224 ± 127
Dried meat**	15.8 ± 9.7	7.8 ± 4.6	34.8 ± 23.9	12.9 ± 7.3
Liver	0.8 ± 0.8	7.4 ± 7.2	4.4 ± 4.9	7.2 ± 7.0
Kidney	0.7 ± 0.3	1.5 ± 0.6	5.0 ± 1.5	1.8 ± 0.8
Sahtú	20-40 yr (n=24,22)	40+ yr (n=26,17)	20-40 yr (n=18,31)	40+ yr (n=29,13)
Fresh meat	126 ± 67	227 ± 129	135 ± 85	250 ± 132
Dried meat**	15.7 ± 8.7	63 ± 41.5	44.1 ± 28.8	44.2 ± 27.8
Liver	1.2 ± 1.2	12.8 ± 13.1	0.47 ± 0.49	11.2 ± 12.2
Kidney	0.8 ± 0.3	5.9 ± 2.3	2.8 ± 1.0	4.9 ± 2.4

*first number represents 24 hr diet recalls in the late winter and food frequency questionnaires for the winter, second number represents 24 hr diet recalls in the autumn and food frequency questionnaires for the summer;

**dry weight basis, all other values fresh weight.

Table 4·3. Average daily intakes (g ± SE) of caribou in Arctic Canada (after Kuhnlein *et al.*, 2000).

	15-19 yr	n	20-40 yr	n	41-60 yr	n	61+ yr	n
Inuvialuit – males								
Meat	88	24	132	100	114	32	72	14
Dried meat*	–	24	22	100	12	32	18	14
Ribs	–	24	7	100	27	32	16	14
Bone marrow	–	24	1	100	4	32	–	14
Liver	–	24	–	100	4	32	–	14
Heart	–	24	–	100	4	32	–	14
Kidney	–	24	–	100	4	32	–	14
Kivalliq – males								
Meat	428	7	259	87	365	33	440	15
Fat	–	7	12	87	–	33	56	15
Ribs	–	7	3	87	–	33	5	15
Dried meat*	–	7	4	87	–	33	7	15
Bone marrow	–	7	2	87	23	33	–	15
Tongue	–	7	–	87	–	33	14	15
Intestine	–	7	1	87	–	33	–	15
Stomach contents	–	7	2	87	–	33	–	15
Stomach	–	7	1	87	–	33	–	15
Baffin – males								
Meat	204	24	106	112	164	82	235	20
Fat	–	24	4	112	10	82	–	20
Dried meat*	2	24	3	112	6	82	–	20
Intestine	–	24	–	112	6	82	–	20
Stomach	–	24	–	112	10	82	–	20
Bone marrow	–	24	–	112	10	82	–	20
Liver	–	24	–	112	6	82	–	20
Kidney	–	24	2	112	0.1	82	–	20
Cartilage	–	24	2	112	–	82	–	20
Labrador – males								
Meat	142	18	148	80	135	68	74	21
Dried meat*	–	18	13	80	2	68	–	21
Heart	–	18	6	80	–	68	25	21
Ribs	–	18	6	80	–	68	–	21
Tongue	–	18	–	80	–	68	6	21
Bone marrow	–	18	–	80	0.5	68	–	21

n = number of 24 hr diet recalls and food frequency questionnaires in the autumn and late winter;
*dry weight basis, all other values fresh weight.

The Canadian Inuit also exhibit high intakes of caribou meat and other caribou products. Average daily consumption data for four Inuit regions are presented in Table 4·3. These data are based on 24 hr dietary recalls for individuals that had eaten caribou within the previous 24 hours (Kuhnlein *et al.*, 2000). The data reveal a high degree of variability in the frequency and amount of caribou consumed. Table 4·3 shows the average consumption for the groups as whole. Within each of these groups it is evident that the heaviest consumers had a fresh caribou meat intake of around 500 to 850 g/d on the days that they ate caribou during autumn and winter. Adding the consumption of other caribou products, especially dried meat, meant the total individual caribou product consumption rates could exceed 1 kg/d on certain days. Obviously, these high consumers represent a limited proportion of the population as a whole. Average consumption rates in the Canadian north are more typically around 100 to 400 g/d (Kuhnlein *et al.*, 2000; Table 4·3). These new data do not support the average consumption rate of 1000 g/d used in the first AMAP assessment, which would imply that individuals were consuming 1000 g of caribou meat per day seven days a week throughout the year. Although this new information indicates that the consumption rates for caribou meat in the selected Gwich'in population group, for which exposure calculations were performed in the first AMAP assessment, may have been substantially overestimated, it appears that there are individuals within the Canadian Arctic that periodically have consumption rates of a comparable magnitude.

The selected groups from the other Arctic countries represent 'average consumers' among larger groups involved in caribou/reindeer herding. The values for caribou/reindeer consumption were therefore lower than those used for Canada. The first AMAP assessment concluded that '*It cannot be ruled out that there are small numbers of individuals within other Arctic countries having similar dietary habits as the selected Canadian community. Accordingly, comparable or higher doses than those calculated for the Canadian selected group may exist within the Arctic*'.

Figure 4·2 shows a revised comparison of ^{137}Cs intake among selected groups using the new Canadian intake data (Berti *et al.*, 1998). The data for the selected Canadian group are now more comparable with those for the selected groups in other Arctic countries.

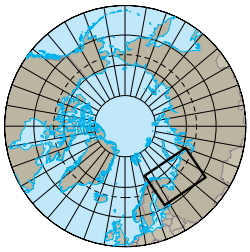

Figure 4·3. Settlements included in the 1998 to 2001 dietary study, and the larger cities.

4.4. Northwest Russia

Several datasets concerning radionuclide transfer to foodstuffs, dietary habit, and wholebody measurements associated with studies of ingested radiation dose in northwest Russia have become available since the first AMAP assessment. Data for the Kola Peninsula and the Nenets Autonomous Okrug (NAO) were collected under the European Union and the Russian–Norwegian bilateral project 'AVAIL', which involved five expeditions to northwest Arctic Russia between 1998 and 2001 (Borghuis *et al.*, 2002; see Annex Tables A4·1 and A4·2).

The purpose of the expeditions was to assess contemporary levels of environmental contamination by the long-lived radionuclides, ^{137}Cs and ^{90}Sr, and to determine site-specific characteristics for estimating current internal doses in the different population groups of northwest Russia.

Three population groups were considered: indigenous peoples, mainly reindeer herders and members of their families (Group I); rural populations and inhabitants of small villages and settlements having mixed diets (Group II); and the populations of big ports and cities whose inhabitants mainly consume imported foodstuffs (Group III).

The expeditions thus aimed to include a variety of different types of Arctic inhabitant. Surveys were undertaken in the areas shown in Figure 4·3, namely:

Lovozero village on the Kola Peninsula where Saami and Komi are occupied in reindeer herding. The Slavic population of the area is not directly connected with reindeer herding but does consume reindeer meat. Umba was also included although its inhabitants do not consume much reindeer meat;

Dolgoshelye village and Mezen town in the Mezen district of the Arkhangelsk Oblast. The Nenets nomad camp located by the mouth of the River Perechnaya was also included; and

Khongurey, Ust'-Kara, Nelmin Nos, and Krasnoye villages, and Nar'yan-Mar town in the NAO. Here, Nenets dominate the indigenous population and are largely occupied with reindeer herding. A nomad camp of reindeer herders, who have owned the land along the Kara Sea between Ust'-Kara and Amderma for many years, was also included.

A questionnaire was used to obtain information about the local population and the consumption and origin of the eight most important foodstuffs in the diet. Together with information on the levels of ^{137}Cs and ^{90}Sr in different foodstuffs, the dietary data were used to assess which foodstuffs contributed most to the radionuclide intake of each population group. The ^{137}Cs wholebody contents of individual local inhabitants were also measured.

4.4.1. Dietary preferences

4.4.1.1. Group I

The diet of reindeer herders in the Mezen district of the Arkhangelsk Oblast and the NAO was similar to that of the Saami and Komi reindeer herders on the Kola Peninsula, both in terms of the consumption of reindeer meat, and fish from local lakes and rivers, and the consumption of milk of local origin. Some differences occurred in the consumption of mushrooms and berries (Table 4·4).

The most significant differences occurred in the consumption of vegetables and fruit. This is primarily explained by 84% of the reindeer herders on the Kola Peninsula having kitchen gardens compared to only 16% in the NAO. However, the contribution of vegetables and fruit to internal dose is negligible. Since the

Table 4.4. Consumption of natural products by Group I inhabitants of northwest Russia (kg/d, mean ± SE) (Borghuis *et al.*, 2002).

	Mushrooms	Berries
Kola Peninsula	0.021 ± 0.005	0.045 ± 0.012
Mezen district	0.036 ± 0.009	0.042 ± 0.008
Nenets AO	0.014 ± 0.002	0.026 ± 0.003
Average	0.024 ± 0.011	0.038 ± 0.015

Table 4·5. Consumption of basic food products (kg/d; mean ± SE) by Group I inhabitants of the northern European part of Russia in 1998 to 2001 (Borghuis *et al.*, 2002).

	Milk	Reindeer meat	Other meat	Potato	Fish	Mushrooms	Berries
Local produce	0.086±0.013	0.310±0.011	0.007±0.001	0.110±0.015	0.130±0.009	0.024±0.011	0.038±0.015
Imported products	0.005±0.003	–	0.006±0.001	0.078±0.009	–	–	–
Total consumption	0.091±0.013	0.310±0.011	0.013±0.002	0.190±0.014	0.130±0.009	0.024±0.011	0.038±0.015

Table 4·6. Consumption of basic food products (kg/d; mean ± SE) by Group II inhabitants of northwest Russia (Borghuis *et al.*, 2002).

	Milk	Reindeer meat	Other meat	Potato	Fish	Bread
Kola Peninsula						
Lovozero (1998) n=25						
Local produce	0.206±0.052	0.083±0.011	0.004±0.004	0.323±0.040	0.109±0.024	–
Imported products	0.079±0.029	–	0.072±0.013	0.049±0.021	–	–
Total consumption	0.285±0.066	0.083±0.011	0.076±0.013	0.376±0.033	0.109±0.024	0.302±0.031
Nenets AO						
Ust'-Kara (2000) n=41						
Local produce	–	0.084±0.016	0.056±0.009	–	0.155±0.014	–
Imported products	0.014±0.008	–	0.022±0.004	0.173±0.012	–	–
Total consumption	0.014±0.008	0.084±0.016	0.078±0.011	0.173±0.012	0.155±0.014	0.382±0.045
Nar'yan-Mar (2000) n=37						
Local produce	0.206±0.056	0.169±0.028	0.040±0.014	0.239±0.042	0.087±0.011	–
Imported products	0.021±0.015	–	0.036±0.010	0.061±0.021	–	–
Total consumption	0.227±0.056	0.169±0.028	0.076±0.019	0.300±0.039	0.087±0.011	0.292±0.019
Mezen district						
Dolgoshelye (1999) n=13						
Local produce	0.497±0.162	0.032±0.009	0.051±0.005	0.333±0.040	0.125±0.020	–
Imported products	–	–	0.004±0.007	–	–	–
Total consumption	0.497±0.162	0.032±0.009	0.055±0.003	0.333±0.040	0.125±0.020	0.324±0.025

n = number in survey.

Table 4·7. Consumption of basic food products (kg/d; mean ±SE) by Group III inhabitants of northwest Russia (Borghuis *et al.*, 2002).

	Milk	Reindeer meat	Other meat	Potato	Fish	Bread
Kola Peninsula						
Umba (1998) n=58						
Local produce	0.128±0.041	0.002±0.002	0.021±0.005	0.334±0.026	0.113±0.011	–
Imported products	0.038±0.015	–	0.058±0.008	0.037±0.013	–	–
Total consumption	0.166±0.044	0.002±0.002	0.077±0.009	0.370±0.023	0.113±0.011	0.332±0.020
Mezen district						
Mezen (1999) n=22						
Local produce	0.414±0.110	0.012±0.005	0.024±0.007	0.420±0.055	0.073±0.014	–
Imported products	–	–	0.101±0.066	–	–	–
Total consumption	0.410±0.110	0.012±0.005	0.125±0.066	0.420±0.055	0.073±0.014	0.423±0.009

n = number in survey.

consumption of mushrooms and berries by this group makes a relatively small contribution to dose, the data for the reindeer herders for the entire northern European part of Russia can be combined (Table 4·5).

4.4.1.2. **Group II**

A comparison of reindeer consumption by reindeer herders (Table 4·5) and typical rural inhabitants of small towns and villages (Table 4·6) shows that, on average, rural inhabitants consume two to four times less reindeer meat than reindeer herders, with almost an order of magnitude lower consumption in Dolgoshelye in the Mezen district. This is explained by the gradual decline of reindeer herding in the Mezen district. In contrast, reindeer farms on the Kola Peninsula and in the NAO are comparatively stable, and Lovozero, Ust'-Kara and Nar'yan-Mar have shops selling reindeer meat.

Rural inhabitants, except inhabitants of Ust'-Kara, exceed the milk consumption rates of Group I by a factor of 2.5 to 5.5 and potatoes by a factor of 1.5 to 2. Ust'-Kara is on the shore of the Kara Sea, there is no agricultural production, and kitchen gardens are rare. Almost all foodstuffs, except reindeer meat and fish, are thus imported. Group II inhabitants have similar fish consumption rates to reindeer herders.

4.4.1.3. **Group III**

Inhabitants of large villages and towns in the European part of Arctic Russia do not consume significant amounts of reindeer meat (Table 4·7). However, milk

consumption exceeds that of Group I by a factor of 1.8 to 4.5 and potato consumption by a factor of 2.

4.4.1.4. All Groups

Average consumption rates of mushrooms and berries by Group II and Group III inhabitants of northwest Russia, i.e. individuals unconnected with reindeer herding, are shown in Table 4·8. Fish consumption rates were similar in all the population groups surveyed. Rates of bread consumption were also similar, at 290 to 350 g/d.

Table 4·8. Consumption of mushrooms and berries (kg/d; mean ± SE) by Group II and Group III inhabitants of northwest Russia (Borghuis *et al.*, 2002).

	Mushrooms	Berries
Kola Peninsula	0.025 ± 0.010	0.045 ± 0.012
Dolgoshelye	0.036 ± 0.010	0.052 ± 0.010
Mezen district	0.042 ± 0.007	0.071 ± 0.011
Ust'-Kara	0.012 ± 0.002	0.028 ± 0.004
Nar'yan-Mar	0.024 ± 0.004	0.026 ± 0.005
Average	0.028 ± 0.006	0.044 ± 0.009

4.4.2. Radionuclides in the diet

The highest ^{137}Cs activity concentrations currently occur in reindeer meat, mushrooms, freshwater fish, and berries (Figure 4·4). The ^{137}Cs level in reindeer meat is around two orders of magnitude higher than in locally produced agricultural foodstuffs. As expected, there are

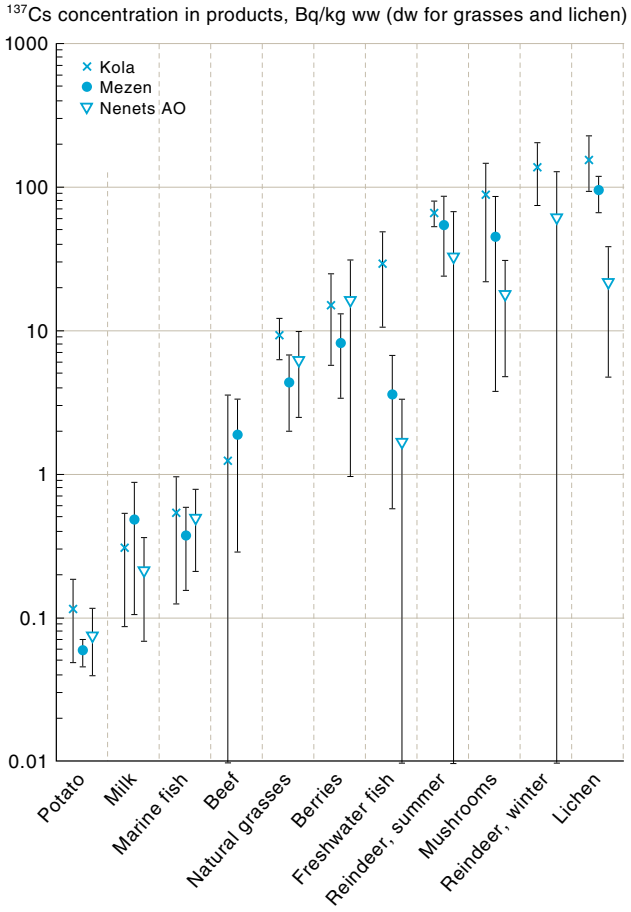

Figure 4·4. Activity concentrations of ^{137}Cs in products collected in northwest Arctic Russia 1998 to 2001 (Borghuis *et al.*, 2002).

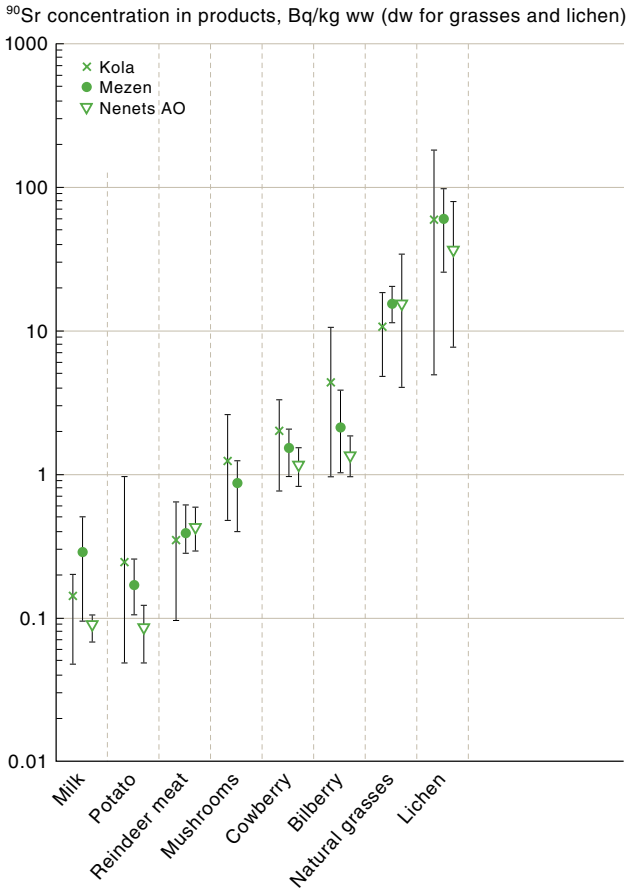

Figure 4·5. Activity concentrations of ^{90}Sr in products collected in northwest Arctic Russia 1998 to 2001 (Borghuis *et al.*, 2002).

significantly higher ^{137}Cs concentrations in lichen and fungi compared with grasses and agricultural products (potato, milk, and beef). Relatively high ^{137}Cs activity concentrations in Arctic freshwater fish are probably due to low mineralization (low dissolved potassium and other ions) of water and thus an elevated accumulation of ^{137}Cs. Activity concentrations of ^{137}Cs in marine fish are an order of magnitude lower than in freshwater species.

Activity concentrations of ^{137}Cs (Figure 4·4) and ^{90}Sr (Figure 4·5) in locally produced milk and potatoes and the corresponding T_{ag} values are similar to those observed in mid-latitudes.

For almost all foodstuffs, ^{137}Cs and ^{90}Sr activity concentrations were higher on the Kola Peninsula than in the other regions, with the differences more pronounced for ^{137}Cs.

4.4.3. Dose estimation

Ingestion doses (E_{int}) were estimated from the dietary intake data for the different population groups and the radionuclide activity concentrations in the various foodstuffs according to the formula:

$$E_{int} = 30 \cdot \sum (dk_i \cdot I_i), \ \mu Sv/month \qquad \text{Eqn. 4.1}$$

where dk_i is the dose factor for ingestion of the i^{th} radionuclide in the body of an adult; dk_i is equal to 1.3×10^{-2} and 2.8×10^{-2} μSv/Bq for ^{137}Cs and ^{90}Sr, respectively (ICRP, 1993); I_i is the daily intake of the i^{th} radionuclide in the body with food in Bq/d; and 30 is the number of days in a month.

Table 4·9. Average daily intake of ^{137}Cs and ^{90}Sr in summer 1998 for Group I inhabitants of Lovozero village (Borghuis *et al.*, 2002).

	Activity concentration, Bq/kg		Consump-, tion, kg/d	Food processing factor	Intake, Bq/d	
	^{137}Cs	^{90}Sr			^{137}Cs	^{90}Sr
Milk	0.39	0.15	0.10	1.0	0.04	0.015
Reindeer meat	70	–	0.30	1.0	20.9	–
Mushrooms	58	0.13	0.021	0.5	0.61	0.0014
Berries	19	3.4	0.045	1.0	0.86	0.153
Potatoes	0.12	0.25	0.31	0.8	0.03	0.062
Fish	20	–	0.10	1.0	2.1	–
Total					24.5	0.23

Table 4·10. Average daily intake of ^{137}Cs and ^{90}Sr in summer 1998 for Group II inhabitants of Lovozero village (Borghuis *et al.*, 2002).

	Activity concentration, Bq/kg		Consump-, tion, kg/d	Food processing factor	Intake, Bq/d	
	^{137}Cs	^{90}Sr			^{137}Cs	^{90}Sr
Milk	0.39	0.15	0.28	1.0	0.11	0.042
Reindeer meat	70	–	0.083	1.0	5.8	–
Mushrooms	58	0.13	0.025	0.5	0.73	0.002
Berries	19	3.4	0.045	1.0	0.86	0.153
Potatoes	0.12	0.08	0.38	0.8	0.04	0.024
Freshwater fish	20	–	0.085	1.0	1.7	–
Total					9.2	0.22

Table 4·11. Average daily intake of ^{137}Cs and ^{90}Sr in summer 1998 by Group III inhabitants of Umba (Borghuis *et al.*, 2002).

	Activity concentration, Bq/kg		Consump-, tion, kg/d	Food processing factor	Intake, Bq/d	
	^{137}Cs	^{90}Sr			^{137}Cs	^{90}Sr
Milk	0.49	0.17	0.128	1.0	0.063	0.022
Mushrooms	59	–	0.021*	0.5	0.62	–
Berries	3.7	–	0.045*	1.0	0.17	–
Potatoes	0.12	0.18	0.33	0.8	0.032	0.048
Fish	2-20**	–	0.11	1.0	0.22-2.2	
Total					1.1-3.1	0.07

* consumption assumed to be the same as at Lovozero village;
** first number corresponds to the average ^{137}Cs activity concentration in marine fish and the second to freshwater fish.

The daily radionuclide intake is estimated by combining the intake of different foodstuffs:

$$I_i = \sum_p (C_{ip} \cdot V_{ip} \cdot K_{ip}), \text{ Bq/d} \qquad \text{Eqn. 4.2}$$

where ($C_{ip}(t)$ is the concentration of the i^{th} radionuclide in the p^{th} foodstuff in Bq/kg; V_{ip} is the daily consumption rate of the p^{th} foodstuff in kg/d; and K_{ip} is the food processing factor accounting for the loss of the i^{th} radionuclide during cooking of the p^{th} foodstuff.

The internal dose to inhabitants from ^{137}Cs may also be assessed on the basis of measured activities in the human body. The mean monthly effective dose through internal exposure in adult inhabitants from ^{137}Cs (E_{137}), based on wholebody measurements, was calculated using the formula:

$$E_{137} = 30 \cdot kd_{137} \cdot Q/M, \text{ μSv/month} \qquad \text{Eqn. 4.3}$$

where Q is the ^{137}Cs activity in the body of a person under investigation in Bq; M is the mass of the body of the person in kg; kd_{137} is the dose rate coefficient converting the specific activity of ^{137}Cs in the body, Q/M, to the effective dose rate; $kd_{137} = 6.3 \times 10^{-3}$ (μSv/kg per Bq/d) (ICRP, 1993).

4.4.3.1. **Group I**

Table 4·9 shows the average daily ^{137}Cs and ^{90}Sr intakes from local foodstuffs in reindeer herders from Lovozero village. The consumption of reindeer meat in summer provides 83% of the ^{137}Cs internal dose. Fish and mushrooms and berries are also significant sources (at 8% and 9%, respectively). The average monthly internal dose from ^{137}Cs in summer 1998 was 10 μSv/month.

The diet of the reindeer herders was the same in spring as in summer. This is to be expected as the food supply of the indigenous population is generally stable throughout the year. Again, the major contributor to the internal dose was reindeer meat, at 88%, with the rest from local freshwater fish, and mushrooms and berries. As in summer, the contribution from milk was small. The average daily intake of ^{137}Cs by reindeer herders in

late winter was 39 Bq and the monthly internal dose was 15 μSv. This is 1.5 times higher than in summer and consistent with a factor of 1.6 between the ^{137}Cs activity concentration in reindeer meat after the winter slaughter and in summer. To estimate the annual internal dose, appropriate weightings were assigned to the winter (seven months with snow) and summer periods.

If wholebody measurements are used for dose estimation (the average ^{137}Cs content in the body of reindeer herders in winter 1999 was 3250 ± 250 Bq), the calculated dose is 8.8 μSv/month. This is only 10% higher than the estimate for the summer period and 1.7 times lower than that estimated on the basis of ^{137}Cs intake using food product data. The contribution of ^{90}Sr to the total internal dose to reindeer herders is about 1-3%.

4.4.3.2. **Group II**

Estimated average daily intakes of ^{137}Cs and ^{90}Sr in key foodstuffs for inhabitants of Lovozero village not directly connected with reindeer herding are given in Table 4·10. Reindeer meat still contributes the most to internal dose (63%), although the average daily consumption (0.08 kg) is 3.5 times lower than for reindeer herders. The next most important contributions are from fish (18%), and mushrooms and berries (17%). The average monthly internal dose from ^{137}Cs to Lovozero rural inhabitants is 3.6 μSv. The contribution of ^{90}Sr to the total internal dose of village inhabitants is about 5%.

4.4.3.3. **Group III**

In Umba, inhabitants rarely consume reindeer meat and so their intake of radionuclides is considerably lower than at Lovozero. Instead, the greatest contribution to internal dose is from local mushrooms and berries. Based on Table 4·11, the internal dose from ^{137}Cs and ^{90}Sr ranges from 0.5 to 1.2 μSv/month.

4.4.3.4. **All Groups**

The contribution of ^{90}Sr to total internal dose is lowest for Group I (reindeer herders) at 2.4 to 3.4% and highest for Group III (urban inhabitants) at 16 to 20%. This reflects the considerable difference in reindeer meat consumption by the two groups. Reindeer meat is the main dose-contributing foodstuff for Group I (through ^{137}Cs intake rather than ^{90}Sr). In Group II (rural inhabitants), the contribution of ^{90}Sr to total dose varies from 3.8 to 5.0%. Only for inhabitants of Dolgoshelye does it reach 15% and this reflects the comparatively low reindeer meat consumption (only 0.032 kg/d).

Table 4·12. Current average internal doses (μSv/yr) in northwest Russia (Borghuis *et al.*, 2002).

	Group (n)	Monthly dose summer	winter	Annual dose
Kola Peninsula (Lovozero, Umba)				
	I (43)	10	19	183
	II (25)	4.5	6.9	71
	III (58)	1.2	1.2	14
Mezen district (Dolgoshelye, Mezen)				
	I (8)	5.3	8.6	87
	II (13)	1.5	1.8	20
	III (22)	1.2	1.4	16
Nenets AO (Ust'-Kara, Nar'yan Mar)				
	I (63)	4.9	8.2	82
	II (41)	1.6	2.5	26
	II (37)	2.9	4.7	47

Table 4·13. Current average internal doses due to the intake of ^{137}Cs and ^{90}Sr normalized to the specific activity per unit soil surface in northwest Russia (Borghuis *et al.*, 2002).

	Annual dose, μSv/yr per kBq/m^2 ^{137}Cs	^{90}Sr
Kola Peninsula		
Group I	100	2.8
Group II	37	2.6
Group III	6.6	2.2
Mezen district		
Group I	31	2.6
Group II	6.3	3.1
Group III	4.6	3.2
Nenets AO		
Group I	30	1.9
Group II	13.0 (9.2-17)	1.1 (0.7-1.5)

Similar dose assessments were performed for inhabitants of the Mezen district in the Arkhangelsk Oblast and the NAO. Estimated doses are summarized in Table 4·12. The calculations were based on the intake of ^{137}Cs and ^{90}Sr with food.

Current doses to inhabitants of the Russian Arctic are <200 μSv/yr, much lower than during the 1960s when global fallout was at its highest. Doses on the Kola Peninsula are higher than in the Mezen district and the NAO. The highest doses occur among reindeer herders. Doses to reindeer herders on the Kola Peninsula are twice those in other regions.

A direct comparison of different areas is achieved by normalizing the internal doses due to ^{137}Cs and ^{90}Sr intake to the ^{137}Cs and ^{90}Sr activity concentrations per unit soil surface (Table 4·13). This shows that doses to Groups I, II, and III per unit deposition are higher on the Kola Peninsula than in the other areas. This presumably reflects the higher transfer to foodstuffs in the Kola region. Table 4·14 shows the contributions of different foods to effective internal dose in northwest Russia.

Table 4·14. Percentage contribution of different foodstuffs to effective internal dose in northwest Russia (Borghuis *et al.*, 2002).

	Kola Peninsula			Mezen district			Nenets AO	
	Group I	Group II	Group III	Group I	Group II	Group III	Group I	Group II
Dairy products	0.14	0.88	2.7	0.54	13	14	0.21	0.86
Beef	0.02	0.03	0.88	0.07	2.3	1.4	0.08	1.3
Reindeer meat	89	63	7.4	90	40	19	95	86
Mushrooms	2.5	7.9	39	4.8	21	31	0.84	2.4
Berries	2.5	6.6	32	2.8	15	26	2.8	7.4
Potato	0.15	1.1	5.7	0.22	2.8	4.5	0.14	0.52
Freshwater fish	5.1	20	11	1.3	5.5	4.1	0.67	1.7
Marine fish	0.09	0.04	1.8	0.14	0.57	0.42	0.19	0.47

Currently, the annual effective dose from external exposure to anthropogenic gamma radiation (^{137}Cs of global and Chernobyl fallout origin) varies from 1 to 10 μSv/yr. The highest values correspond to reindeer herders and the lowest to urban inhabitants.

Reindeer meat consumption is important to all groups, with the exception of the coastal community at Umba, and dominates intake for reindeer herders. For rural inhabitants, fish, mushrooms, berries, and dairy products are also important. For urban dwellers, dairy products, mushrooms, berries, and freshwater fish can be important contributors to internal dose.

This assessment of communities in northwest Russia has shown that some of the current variations in food product contribution and total dose are explained by the following factors:

- the rate of reindeer meat consumption;
- the higher transfer of radiocesium to foodstuffs on the Kola Peninsula than in areas further east;
- the location of the community – coastal communities receive the lowest doses;
- land use – communities on the Kola Peninsula cultivate their own crops and buy local produce in shops to a greater extent than communities further east; and
- the rate of mushroom and berry consumption.

4.4.4. Comparison of past and present estimates of internal dose

Table 4·15 compares the internal doses estimated for the 1990s in the first AMAP assessment (AMAP, 1998) with those of the present assessment (based on the data in Sections 4.4.1. to 4.4.3.). Because the recent data were obtained after those considered in the first AMAP assessment, the annual values of internal dose for 'average' and 'selected groups' (reindeer herders) for the period 1990 to 1994 (Tables 8·13 to 8·16 in AMAP, 1998) were corrected to allow for the expected decline in activity with time. For reindeer meat and freshwater fish the T_{eff} for ^{137}Cs was taken as 12 yr. For other foodstuffs, the corresponding physical half-lives of ^{137}Cs and ^{90}Sr were used as the effective ecological half-lives.

The two dose estimates for the average population agree well. However, the effective internal dose estimated for the reindeer herders in the first AMAP assessment is approximately twice that of the present estimate. This is mainly due to the lower average radionuclide activity concentrations in the data used in the present assessment, and is despite the slightly higher intakes of reindeer meat used in the present assessment.

Table 4·15. A comparison of annual internal effective doses (μSv) due to the intake of ^{137}Cs and ^{90}Sr for the average population and reindeer herders in western Arctic Russia in the late 1990s estimated during the first AMAP assessment (AMAP, 1998) and the present assessment.

	AMAP (1998)		Present assessment	
	Average population	Reindeer herders	Average population	Reindeer herders
^{137}Cs	14	280	12	80-180
^{90}Sr	1	5.6	1.5-2.2	1.3-2.0

4.5. European spent nuclear fuel reprocessing plants

The first AMAP assessment addressed discharges from the Sellafield and Cap de la Hague spent nuclear fuel reprocessing plants in Western Europe, the transport of released radionuclides into the Arctic, and the associated doses to Arctic inhabitants. This assessment focuses on ^{99}Tc and ^{129}I, two radionuclides released from these sources which were not discussed in detail in the previous assessment and about which concern has recently been raised.

4.5.1. Technetium-99

Technetium-99 has two isomers, ^{99}Tc and ^{99m}Tc. The latter has a short half-life ($T_{1/2}$ = 6 hr) and is used in medical applications. It is of little concern outside the area of medical health physics. The former, however, is a long-lived fission product ($T_{1/2}$ = 212 000 yr) that has generated increased interest since the first AMAP assessment. The main sources of ^{99}Tc are nuclear weapons tests and the nuclear fuel cycle. The most important sources for western Scandinavia and Arctic environments are the European nuclear fuel reprocessing plants at Sellafield and Cap de la Hague (Dahlgaard *et al.*, 1997). Releases of ^{99}Tc from Sellafield have increased substantially in recent years (Section 2.2.1). As a result, a steep increase has been observed in the levels of ^{99}Tc in some marine biota, particularly crustaceans and seaweeds (Section 3.3.1). Owing to its long half-life and conservative behavior in seawater, ^{99}Tc is now of concern to some European nations (especially Ireland and the Nordic countries) as evidenced by discussions at the Ministerial Meeting of the OSPAR Commission in 1998 (OSPAR, 1998).

The contribution of ^{99}Tc to average individual doses to members of the local critical group of seafood consumers for Sellafield discharges during the period 1994 to 1996 are estimated at 18 to 42 μSv/yr (Uranium Institute, 1998).

An assessment of ^{99}Tc in the marine environment, around Ireland which included an estimate of doses to members of the Irish population, was developed in preparation for the 1998 Ministerial Meeting of the OSPAR Commission (Pollard *et al.*, 1998). This concluded that the individual (committed effective) doses to average seafood consumers were 0.053 μSv in 1996 and 0.068 μSv in 1997, reflecting a moderate increase associated with the increased concentrations in seafoods resulting from the increased discharges of ^{99}Tc from Sellafield that began in 1993. For heavy seafood consumers (presumably corresponding to a critical group), the corresponding values were 0.21 μSv in 1996 and 0.27 μSv in 1997. However, it should be noted that there are no estimates of the uncertainties associated with these dose estimates. In a follow-up to this work, Smith *et al.* (2001) extended the calculations to 1998, a period when ^{99}Tc activity concentrations in the marine biota of Western Europe were still increasing. The values quoted for average and heavy consumers of seafood for the years 1996 and 1997 were identical to those quoted by Pollard *et al.* (1998). The doses for 1998 were estimated at 0.062 μSv for average consumers and 0.25 μSv for heavy consumers of seafood. As stated by Smith *et al.*

(2001), these doses are of negligible radiological significance when compared to the annual dose limit of 1000 μSv for members of the public from practices involving controllable sources of radiation (ICRP, 1991). The authors also noted that doses to the same seafood consumer groups associated with ^{210}Po, a natural radionuclide for which doses through marine pathways are usually dominant, were 32 μSv and 148 μSv, respectively, for average and heavy seafood consumers.

Since conservative radionuclides discharged from Sellafield can be transported with the prevailing marine currents into Arctic waters it is appropriate to consider the biogeochemical behavior of ^{99}Tc, and associated impact in terms of human doses, in northern marine environments. Although human doses in Arctic regions are likely to be significantly below those observed in areas close to discharge points, differences in biological uptake and human dietary patterns may offset lower ambient contamination levels caused by dilution. In other words, uncertainties associated with the fate of ^{99}Tc in Arctic marine environments require attention.

Within the context of estimating human exposure in northern marine environments based on analyses of ^{99}Tc activity concentrations in foodstuffs derived from the sea, the Norwegian Radiation Protection Authority (Brown *et al.*, 1998) observed that: '*Individual radiation doses from human consumption of seafood from Norwegian waters are probably low due to the present low levels of contamination and the low dose conversion factor of ^{99}Tc*'.

The Uranium Institute (1998) has estimated the dose in relation to the quantities of specific seafoods consumed in Norway on the basis of reported ^{99}Tc activity concentrations in seafood (Brown *et al.*, 1998). The Uranium Institute calculated that the consumption of Norway lobster (*Nephrops norvegicus*), containing ^{99}Tc in the range 11.2 to 42 Bq/kg, would give rise to an individual dose to the consumer of <0.03 μSv/kg ingested. The corresponding value for mussels and shrimps, with ^{99}Tc in the range 0.54 to 0.68 Bq/kg, is 0.5 nSv/kg. These are very low dose/mass consumption ratios. Even an extreme seafood consumer eating 1 kg/d of Norway lobster would receive an annual dose of <11 μSv.

Thus, it can be argued that discharges of ^{99}Tc from Sellafield, even at rates close to the authorized release rate of 200 TBq/yr, as in 1995, lead to levels in Arctic marine waters that are of no radiological significance for human populations. However, there is much uncertainty regarding the biogeochemical behavior of ^{99}Tc in the marine environment, which undermines the ability to make prospective human impact assessments with any great conviction. It is notable that uptake levels under field conditions for some crustaceans (Brown *et al.* 1999; Busby *et al.* 1997) were largely unstudied before the recent discharges from the Enhanced Actinide Removal Plant (EARP) caused environmental levels to increase dramatically (Section 2.2.1), although laboratory studies had indicated that uptake rates could be high in these groups and that large inter-species variability existed. Technetium uptake by many Arctic species remains largely unstudied and thus an additional area of uncertainty with respect to human dose assessment. Finally, although technetium forms the highly soluble pertechnetate ion in oxygenated seawater and can therefore be modelled using hydrodynamic models, little attention has been given to the field behavior of ^{99}Tc under anoxic conditions, as occur in some Norwegian fjords, for example. In reality, little is known about the environmental transfer and biological uptake of reduced forms of this radionuclide.

4.5.2. Iodine-129

Iodine-129 is a long-lived fission product ($T_{1/2}$ = 17 000 000 yr) released from the nuclear fuel cycle. The release of ^{129}I to the European marine environment has increased in recent years primarily due to increased release rates from the Cap de la Hague fuel reprocessing plant (Yiou *et al.*, 1995). Owing to its long physical and environmental half-life ^{129}I is globally dispersed. The total global release from the civilian nuclear fuel cycle (reactors and reprocessing operations) is 14.8 TBq. This corresponds to a collective dose commitment, truncated at 10000 years, of 295 manSv (UNSCEAR, 2000). The maximum individual dose from globally dispersed ^{129}I is 0.005 μSv/yr (UNSCEAR, 2000). Thus, current individual dose rates are of little significance and a virtually insignificant source of risk to human health.

4.6. Conclusions

This chapter presents some dose assessments based on newly available information. The main outcome is as follows.

Doses to the Faroe Islands' population were consistent with the outcome of the preliminary assessment during the first AMAP assessment.

The consumption of caribou by the indigenous peoples in Arctic Canada has been reassessed and the doses to critical groups are now shown to be lower.

There is good agreement between the present dose rates estimated for the average population in northwest Russia in this assessment and those of the first AMAP assessment. However, the effective internal dose estimated for reindeer herders in the first AMAP assessment has now been halved. This is mainly due to data reporting lower average radionuclide activity concentrations in reindeer meat, and is despite slightly higher intakes of reindeer meat.

Although doses to the Arctic population from the release of ^{99}Tc and ^{129}I are very low, because these radionuclides behave conservatively in seawater, have very long half-lives, and because ^{99}Tc accumulates strongly in certain species, more work should be done to assess their effects on marine biota.

The doses to populations from a number of nuclear power plants within or near the Arctic were assessed during the first AMAP assessment. As this situation is not known to have changed significantly since then, that assessment has not been updated.

Further work should be carried out to assess the uncertainties for calculated internal doses.

Annex. Tables

Table A4·1. Activity concentrations of ^{137}Cs (Bq/kg ww) in products from northwest Arctic Russia 1998-2001 (Borghuis *et al.*, 2002).

	Kola Peninsula		Mezen district		Nenets AO	
	n	mean ± SD	n	mean ± SD	n	mean ± SD
Reindeer meat						
summer	10	70 ± 14	3	58 ± 33	21	34 ± 38
winter	44	146 ± 68	–	–	30	64 ± 73
Fish marine	8	0.56 ± 0.43	4	0.39 ± 0.23	8	0.51 ± 0.29
Fish freshwater	19	31 ± 20	9	3.8 ± 3.2	20	1.8 ± 1.9
Mushrooms	57	93 ± 70	35	47 ± 43	26	19 ± 14
Berries	22	16 ± 10	7	8.7 ± 5.2	34	17 ± 16
Milk	34	0.32 ± 0.23	9	0.51 ± 0.40	9	0.22 ± 0.15
Beef	18	1.3 ± 2.3	5	1.9 ± 1.6	–	–
Potatoes	12	0.12 ± 0.07	3	0.060 ± 0.013	4	0.08 ± 0.04
Lichen*	20	164 ± 80	8	100 ± 28	23	23 ± 18
Natural grasses*	3	9.7 ± 3.2	4	4.6 ± 2.5	8	6.6 ± 4.0

*dry matter.

Table A4·2. Activity concentrations of ^{90}Sr (Bq/kg ww) in products from northwest Arctic Russia 1998-2001 (Borghuis *et al.*, 2002).

	Kola Peninsula		Mezen district		Nenets AO	
	n	mean ± SD (range)	n	mean ± SD (range)	n	mean ± SD (range)
Reindeer meat	10	0.36 ± 0.18 (0.10-0.65)	3	0.42 ± 0.1 (0.30-0.63)	4	0.44 ± 0.13 (0.31-0.61)
Mushrooms	19	1.30 ± 0.80 (0.50-2.70)	–	–	4	0.90 ± 0.38 (0.42-1.30)
Berries	8	2.80 ± 2.40 (0.80-11.0)	8	1.61 ± 0.44 (0.70-4.00)	7	1.00 ± 0.51 (0.40-1.90)
bilberry	3	4.61 ± 5.20 (1.0-11.0)	3	2.20 ± 1.62 (1.10-4.00)	2	1.40 ± 0.64 (1.00-1.90)
cowberry	5	2.12 ± 1.10 (0.80-3.40)	3	1.62 ± 0.61 (1.00-2.10)	2	1.20 ± 0.45 (0.86-1.50)
Milk	5	0.15 ± 0.09 (0.05-0.21)	4	0.30 ± 0.21 (0.10-0.54)	5	0.094 ± 0.018 (0.07-0.11)
Potatoes	12	0.25 ± 0.11 (0.05-1.00)	9	0.18 ± 0.08 (0.11-0.27)	5	0.090 ± 0.034 (0.05-0.13)
Lichen*	14	62 ± 56 (5.2-187)	9	61 ± 27 (27-99)	16	38 ± 22 (8-83)
Natural grasses*	3	11 ± 7 (5-19)	6	16 ± 3.7 (12-21)	6	16 ± 12 (4.2-35)

*dry matter.

Chapter 5

Protection of the Environment from the Effects of Radiation

5.1. Introduction

There is a growing awareness that radiation risk management needs to address the question of effects on the environment. Radiological protection has traditionally been based on the protection of man. This is because the international advisory body on such matters, the International Commission on Radiological Protection (ICRP), has maintained a strong bias toward human health. The ICRP has stated that: *'The Commission therefore believes that if man is adequately protected then other living things are also likely to be sufficiently protected'* (ICRP, 1977). More recently, a caveat has been added (ICRP, 1991) stating that *'individual members of non-human species might be harmed, but not to the extent of endangering whole species or creating imbalance between species'*.

The inadequacies of applying this approach to environmental protection are increasingly recognized, from both scientific and ethical perspectives (Strand, 2002; Strand *et al.*, 2000). One problem is that no evidence is given to support the ICRP statements, with the result that regulatory bodies in many countries are not in a position to demonstrate explicitly that the environment is being protected for a given situation. Laboratory studies and accidents have shown that radiation can have a number of detrimental effects on biota, including mortality, and reproductive and genetic damage. Nevertheless, current knowledge about the effects of radiation on wild plants and animals is limited and subject to large uncertainties, and there is little consensus on the relevance and acceptability of these effects within the context of risk management. The ICRP statements are potentially invalid in certain situations, for example when pathways to man do not exist or are long and tenuous, or when accidents contaminate sparsely populated areas. Hence, there are likely to be situations where the resident biota are exposed to harmful doses but doses to man are maintained at levels well below the recommended dose limits (Pentreath, 1998). It could be anticipated that the Arctic, where human population densities are very low and exposure pathways to humans can be relatively long, is a prime example.

For these reasons, there has been increasing pressure to explicitly demonstrate environmental protection from radiation and to incorporate environmental considerations into the system of radiological protection. AMAP activities, focusing on radioactivity and other hazardous substances, have played an important role in driving the debate, particularly by highlighting inconsistencies between the approaches taken for radioactivity and other environmental pollutants (Strand *et al.*, 2002). Widespread international consensus has been reached over the last couple of years on the need to develop a rationale for the protection of the environment from ionizing radiation and to demonstrate explicitly that the ecosystem and its components are not being harmed by exposure to radionuclides (Strand and Oughton, 2002).

The subject is specifically addressed within some agreements, for example the Joint Convention on the Safety of Spent Fuel Management and on the Safety of Radioactive Waste Management. Furthermore, the second principle of the International Atomic Energy Agency (IAEA) Safety Fundamentals for the Management of Radioactive Waste states that: *'Radioactive waste shall be managed in such a way as to provide an acceptable level of protection of the environment.'* In addition, several relevant agreements were made at the 1992 UNCED Earth Summit in which a number of general principles for environmental protection were laid down. An example is 'The Rio Declaration' (UNCED, 1992) which emphasizes the issue of sustainable development in Principle 4, by stating that *'Environmental protection shall constitute an integral part of the development process and cannot be considered in isolation from it.'*

5.2. Frameworks for environmental protection

Developing and defending a practical and coherent system of protection for flora and fauna raises a number of dilemmas and conflicts, including those relating to scientific, ethical, and legal issues. A better understanding of ecological effects and their uncertainties primarily requires a framework for risk and impact assessment that can incorporate the sensitivities of various species and ecosystems. Factors influencing sensitivity include exposure pathways, uptake to biota, and dose-effect relationships. These can be ecosystem-dependent (for example, nutrient status or biological activity) and species-dependent (such as high bioaccumulation of ^{99}Tc by lobster or the radiosensitivity of pine compared to other tree species). Acute lethal doses can vary by several orders of magnitude among and within species. Moreover, reproductive and population health effects may occur at much lower doses than would kill an organism and there is little information about the effects of low chronic exposure.

Ethical issues include whether animals have moral status and why, the definition of harm in relation to the exposed population or individuals, the balance between the interests of humans and non-human species, and the fundamental issue of why the environment should be protected anyway. In common with many risk management policies, the answers will need to reflect both scientific knowledge and ethical values. Interestingly, many of the groups concerned with the protection of the environment from radiation, including the IAEA and ICRP, have identified a need to address the ethical and philosophical questions. AMAP has collaborated on work with the International Union of Radioecologists (IUR), which was one of the first international organizations to

actively promote the need to focus on non-human biota and to propose a system for impact assessment.

Any framework for the protection of the environment from radiation should be compatible with protection systems for other environmental stressors. However, it is important to be aware that this area of law is under continuing development. There is general worldwide consensus on the issue of human rights (although not total agreement on how those principles might be applied in practice), which simplifies the management of human radiation exposure in some respects. Nothing like the same level of agreement has been reached on environmental principles however, although progress is being made and is pertinent to the present assessment. There are three major points to bear in mind when addressing the development of frameworks for protection of the environment from radiation. First, legislation for environmental protection is relatively new and still undergoing development. Second, the issue is global, is deemed important by governments and the public alike, and has stimulated action on an international scale. Third, practical solutions are not without conflict and controversy. Not withstanding these difficulties, examples of environmental law can be found in the national laws of every country. Although their scope and detail vary considerably, progress during the last 30 years has led to a certain amount of agreement on what is meant by the 'environment' and its 'protection' and which principles should guide that protection (see Box 5·1).

5.2.1. General legal and ethical principles

A two-stage approach is useful when assessing the legal and ethical basis for the development of a framework for environmental protection, namely: to consider some general and/or common legal and ethical principles used in environmental protection; and then to derive some policy or management principles on the basis of these being specifically relevant and pertinent to protection from radiation.

The following principles are drawn from international and national environmental policy (i.e., the Rio Declaration; and policy arising from the European Union, the U.S. Environmental Protection Agency, and the U.K. Environment Agency) or from environmental ethics. The list is not exclusive and reflects the broad issues and spirit of international and national law.

1. It is the responsibility of all humankind, where possible, to prevent detriment to the environment and to preserve and protect the health and integrity of the Earth's ecosystem. (*Principle of responsibility*).

2. The use and exploitation of natural resources must be sustainable and should equitably meet the developmental and environmental needs of present and future generations. (*Principle of sustainable development*).

3. Society must recognize the serious impact of humans in causing extinction and a loss in species and actively promote conservation measures to preserve the Earth's biodiversity. (*Conservation/biodiversity principle*).

4. Humans should avoid causing suffering to other living organisms. (*Welfare principle*).

5. Humans should respect the inherent and intrinsic worth of nature, recognizing that the environment has a value beyond its direct impact on human interest. (*Principle of respect*).

6. Environmental management needs to be combined with concerns for economic and social justice (particularly in developing countries) and with the informed participation of affected citizens. (*Principle of environmental justice*).

7. Decisions on environmental issues should reflect scientific understanding, acknowledge uncertainties, and recognize the identity, role, culture, and specific knowledge of indigenous peoples, traditional practices, and local communities. (*Transparency and participation principle*).

8. In order to protect the environment, a precautionary approach should be encouraged. Where there are threats of serious or irreversible damage, lack of full scientific certainty should not be used as a reason for postponing cost-effective measures to prevent environmental degradation. (*Precautionary principle*).

9. Authorities should promote the internalization of environmental costs taking into account that the polluter should bear the costs of pollution, including those connected to liability and compensation. (*Polluter pays principle*).

10. The need to prevent environmental damage at source requires that environmental impact assessments should be carried out for all new developments, proposals, and technologies. (*Environmental impact; justification principle*).

Box 5·1. *Definitions*

The term *environment* has been defined in a number of national and international laws. Common to most definitions is the notion that the environment consists of man, biota (e.g., microorganisms, plants, and animals), abiota (e.g., soil, water, and air), physical surroundings (e.g., climate, and light), and their interactions. Some definitions extend to both natural and man-made features of the environment (i.e., cultural heritage, and buildings); some limit the definition to those external factors having a direct effect on living organisms.

Protection of the environment is predominantly perceived as the prevention of detriment to the environment and its living components. But the term can also encompass restoration, enhancement, and maintenance of environmental quality. While recognizing natural environmental stressors, most legislation and international conventions deal specifically with anthropogenic effects.

In the broadest sense *protection of the environment from ionizing radiation* might include all biotic and abiotic components of the Earth's biosphere. In a more practical sense, the abiotic component of the biosphere is known to be unaffected directly by the effects of radiation under all but the most extreme of conditions. In some instances, an interpretation of 'damage' might reflect that an environment is contaminated *per se*, particularly for ecosystems perceived as 'pristine' such as the Arctic. However, in most cases, efforts to quantify systematically the consequences of radiation exposure and to develop a system for protection might be more constructively focused on the most sensitive components of the biosphere, i.e., living organisms, but not totally excluding the abiotic environment.

At present, the only part of the environment explicitly considered for protection from ionizing radiation is man.

Depending on the context, some principles may be deemed more relevant than others and some more fundamental than others. In practice, the principles may even conflict (e.g., 5 and 6). Principles 1 to 5 concern the question of *why* it is necessary to protect the environment, and 6 to 10 *how* to achieve this protection in practice.

The IAEA recently concluded that despite the apparent diversity of values in the different ethical outlooks, consensus on principles of environmental protection was sufficient to identify five common principles, namely: conservation of habitat and species; maintenance of biodiversity; sustainability; environmental justice; and human dignity (IAEA, 2002). Clear support for these five principles was obtained at the IUR consensus conference in 2001, which was attended by participants representing a wide range of disciplines connected to radiation protection and environmental protection. Participants identified a need for 'development of policy in an open, transparent, and participatory manner', considered that 'the best available technology, including consideration of economic costs and environmental benefits, should be applied to control any release of radionuclides into the environment', and supported a precautionary approach to risk management (Strand and Oughton, 2002).

5.2.2. Management of environmental risk

In general, programs addressing the management of environmental risk can be grouped (although somewhat arbitrarily) into three categories:

- management through pathway-based analysis of exposure, often involving environmental standards (e.g., radiation dose to certain organisms or concentrations of radionuclides in environmental media);
- management through process standards relevant to specific source(s) based on *best available technology* (BAT) and similar criteria of technical status and performance; and
- pure management standards, which may include certification schemes or schemes that ensure that positive action is taken to protect the environment and where continuous performance improvement is sought. An example is the EC Eco-Management and Audit Scheme (EMAS).

Pathway-based schemes are generally considered most relevant to the development of assessment frameworks for the environment, but aspects of other schemes may be incorporated when appropriate.

5.2.3. System for environmental impact assessment

A coherent and logical environmental impact assessment methodology for ionizing radiation is essential (Pentreath, 1999). Components that could form the basis for such a system include:

- a set of reference organisms – not all organisms can be studied, necessitating a selection procedure;
- a set of quantities and units to express doses to biota. Currently, doses are expressed in Grays per unit time, which does not reflect the variable biological effects arising from equal absorbed doses of differing radiation types;
- a defined set of dose models for a number of reference flora and fauna. Methodologies exist which allow the calculation of doses to organisms with varying geometries (e.g., consensus is required in adapting these algorithms for use within a protection framework); and
- a set of dose–effect relationships for reference organisms that could include data from low-exposure (e.g., cytogenetic effects) to high-exposure (e.g., lethal effects) situations.

Discussion within the scientific community has led to the adoption of these points into a proposed strategy comprising three key components (IUR, 2000), namely: exposure pathways and retention of radionuclides by biota; dose calculations; and dose-effect relationships.

5.2.3.1. Exposure pathways and retention of radionuclides by biota

The outcome of the work on exposure pathways will be based on the acquisition and synthesis of information concerning ecological characteristics and radionuclide uptake within selected ecosystems. Simple reference models could be developed for the simulation of radionuclide migration and uptake to the whole organism (and organs if applicable) for those reference species living in representative terrestrial and aquatic ecosystems.

5.2.3.2. Dose calculations

Radiation dosimetry models will be developed for the reference organisms. These will be designed to estimate the actual or potential absorbed dose rates to the organisms from internal and external sources of α-, β-, and γ-radiation. The final output will be a tabulation of absorbed dose rate coefficients (Gy/hr per unit radionuclide activity concentration in the relevant environmental compartment) for each reference organism for the radionuclides of concern. It is likely that the reproductive organs will be important targets for inclusion in the dosimetry models.

5.2.3.3. Dose–effect relationships

Endpoints of concern in individual generic organisms could be defined and dose rate/response relationships for the chosen endpoints tabulated. This would involve the integration of data from earlier reviews, and assessments of the potential impacts of radiation in the environment, assessments of the wider radiobiological literature, and assessments of newly available information from the Kyshtym (see Section 7.5.1) and Chernobyl accidents. Relevant effects of radiation will probably include, but not necessarily be limited to, changes in morbidity, mortality, fertility, fecundity, and mutation rate. Information will be organized so as to indicate the approximate dose rate/response relationships. An attempt should be made to quantify the intrinsic uncertainty in these threshold dose rates (e.g., through the extrapolation of laboratory data to natural conditions) and to indicate possible modifying influences (e.g., the influence of other environmental variables).

5.2.4. Target level of biological hierarchy

It is generally recognized that protective action should be taken in such a way as to ensure that populations of organisms receive an adequate level of protection (IAEA,

1992, 2000) and that the functioning of their associated ecosystems is unaffected by the presence of a contaminant. A practical approach to ensure that unacceptable effects on populations are avoided is to target protective action at the organizational level below populations, i.e., individuals. This is justified on the basis of a number of precepts, including:

- population effects are unlikely to be manifested if individuals are unaffected;
- population effects are more complex to assess than effects on individuals and more likely to be masked by the normal range of spatial and temporal ecosystem variability;
- scientific information on population effects is comparatively scarce; and
- in protecting threatened or endangered species, consideration of individuals is necessary.

However, the reasoning is not straightforward in all cases, bearing in mind that:

- for a variety of species (e.g., with asexual or vegetative propagation), individuals and populations in the conventional sense may be difficult to differentiate;
- there are cases where individuals may be affected (e.g., in the case of endocrine disrupters) while populations remain unaffected; and
- in the case of stochastic effects, effects may be observed in individuals while not affecting the viability of the population.

5.2.5. Dose or dose rate as an indicator of actual or potential impact

One approach to environmental assessment involves the calculation of the dose or dose rate to reference organisms. The rationale being that biological effects of radiation are mediated through the absorbed dose and much information is available linking the severity of effects to the dose or dose rate.

Alternatively, assessments could be based on radionuclide activity concentrations. However, the dose is further modified by the type of radionuclide and external and internal geometry, as well as other factors such as lifespan and size. Activity concentrations could be of relevance in compliance discussions, e.g., by comparing expected/observed concentration data with data from dose standards (e.g., U.S. DOE, 2002). However, for assessing effects, including the radiation dose or dose rate adds transparency.

Several dosimetry models are available for aquatic and terrestrial environments, although these are not necessarily sufficiently comprehensive for developing a framework for environmental protection. For the aquatic environment, the generic models relate to: small and large phytoplankton; pelagic and benthic crustaceans; benthic molluscs; and pelagic and benthic fish. These have been developed to the point at which dose rate factors have been tabulated for a range of radionuclides in environmental media (Amiro, 1997; Pentreath and Woodhead, 1988). It is envisaged that future work will focus on the development of the dosimetry models, and the associated dose conversion factors that relate directly to the reference organisms (and their local environment).

5.2.6. Practical and ethical advantages of the framework

A number of practical and ethically-relevant advantages of this framework can be highlighted; the framework is site- and case-specific, transparent, involves stakeholder participation, enables comparison with other environmental contaminants, is 'bottom-up'; is applicable to individuals and populations; and is compatible with anthropocentric and ecocentric environmental philosophies.

5.2.6.1. Site- and case-specific

That the framework is site- and case-specific promotes the notion that there may be a number of different reasons for protecting the environment. For example, the case may depend on available alternatives, the ecosystem itself (e.g., a protected habitat or common resource), and/or the organisms it contains (e.g., endangered species). There is also uncertainty in going from a measurement of concentrations in abiotic compartments (e.g., soil, water), to calculations of accumulation and doses in organisms, and to estimates of cellular up to ecosystem effects. Source-specific, site-specific, species-specific, and individual-specific variability all contribute to such uncertainty. This complexity has the disadvantage of introducing difficulties and there may be cases where a simple approach is sufficient. Until better scientific evidence is available to support such judgments, oversimplification should be avoided.

5.2.6.2. Transparency

The framework is transparent in that it indicates the potential consequences of actions and how these were derived. It also provides information relevant to the issue of 'risk', for example, uncertainties as to outcome, probabilities of harmful effect, errors in dose-risk calculations, and model sensitivity. Honesty about the level of scientific knowledge (meaning some distinction between what is widely acknowledged as fact, generally accepted, disputed, difficult to predict, unknown, etc.) is fundamental to building public trust; short-sightedness or dishonesty is one of the fastest ways to lose this trust.

5.2.6.3. Stakeholder participation

The framework promotes a more open debate on the acceptability of the consequences of radiation exposure to biota, and encourages public and stakeholder participation in such debates. A simple statement that 'releases are below dose limits' tends to beg the question as to where the limits came from and whether they are appropriate. It is also questionable whether the public is sufficiently competent to participate in such debates, and whether the perceptions influencing attitudes to their *own* risk – for example, whether voluntary or imposed – are equally relevant to the question of what is acceptable to animals and *other* living organisms. The public is not always 'rational' and consistent in the way it values animals (dogs and pandas being more important than mosquitoes and worms). Who decides which factors are relevant?

5.2.6.4. Comparison with other environmental contaminants

Because the framework provides information on effects and uncertainties for a range of endpoints, it should be possible to use that information to compare the environmental effects of other practices or alternative actions. Effects from radiation exposure may be compared directly with effects of other environmental stressors, many of which result in the same biological endpoints. This is an important step towards 'holistic' environmental management, and promotes coherence with other methods.

5.2.6.5. Bottom-up

In ecotoxicology, there is often talk of a distinction between 'bottom-up' and 'top-down' systems. This paradigm has attracted increasing attention, largely owing to scientific developments in the analytical techniques used to study the mechanisms and processes of environmental effect (e.g., molecular biology, population studies, and vulnerable species). A 'bottom-up' system means that the framework first acknowledges that actions can have a variety of effects on the environment (from DNA to ecosystems), and considers a range of biological endpoints, changes, and causes. From a risk management point of view, the question is: What might we do and how can we avoid doing it? A 'top-down' system focuses on constraints, standards, and compliance, usually derived from 'no observed effect level' or 'critical load' criteria. In this case, the question is: How much can we do?

5.2.6.6. Applicable to individuals and populations

The main area of focus for the framework is individual organisms. This is sometimes necessary from the point of protection, as in the case of protected species. But evaluation of possible population effects can also be derived from individual effects. Also, the individual is often the highest level at which scientific experiment and hypothesis testing can be directed. Observed biological or physiological effects on an individual organism (or its cells, DNA, etc.) may be reduced causally to the radiation exposure; subsequent effects at a population or ecosystem level require more complicated ecological modelling.

5.2.6.7. Compatible with anthropocentric and ecocentric environmental philosophies

Lastly, the framework is compatible with anthropocentric and non-anthropocentric (i.e., ecocentric) environmental philosophies and can be incorporated into national environmental legislation.

5.2.7. Conclusions

A system for assessing the consequence of radiation exposure on Arctic flora and fauna should have high priority. This requires collaboration at the international level and, with this in mind, joint activities are planned between AMAP and IUR. The European Commission has also initiated further scientific developments through the research projects FASSET (Framework for Assessment of Environmental Impact) and EPIC (Environmental Protection from Ionising Contaminants in the Arctic).

There is a need for the development of a framework for the protection of the environment from ionizing radiation. This is also required to structure the information derived from earlier studies in order to direct future scientific research. Such a system will include environmental transfer models, environmental dosimetry models, and tabulated dose-effect relationships. The system will also require 'reference organisms' (i.e., a group of organisms that are selected from a number of criteria such as radiosensitivity, accumulation potential, ubiquity, and importance to ecosystem functioning) and the derivation of relevant quantities and units. The final system should allow regulators to explicitly and transparently demonstrate a commitment to environmental protection and should provide a basis for developing standards against which to test for compliance of current and future practices.

5.3. Arctic-specific issues

The Arctic requires special attention in the selection of reference organisms owing to its greater vulnerability and lesser abundance of species. The project EPIC – an EC Inco-Copernicus funded research project coordinated by the Norwegian Radiation Protection Authority – aims to develop a methodology for the protection of natural populations of organisms in Arctic ecosystems from radiation. One component has been the development of a list of Arctic-specific reference organisms (Beresford *et al.*, 2001). These were proposed on the basis of their ecological niche, radiosensitivity, likely internal and/or external exposure to radionuclides, and their suitability for monitoring and/or future research.

5.3.1. Identification of reference organisms

5.3.1.1. Biological endpoints

The four 'umbrella' types of biological effect are morbidity (the general well-being of the organism), mortality, reproductive success, and cytogenetic effects.

The choice of endpoints will be facilitated by the development of a database for biological effects on a number of groups of terrestrial and aquatic fauna and flora. The effects of radiation on plants and animals have been reviewed many times from the perspective of assessing the potential impacts of radioactive waste disposal (IAEA, 1976, 1988, 1992; NCRP, 1991; UNSCEAR, 1996). The present need is to structure this information so as to identify the levels of dose rate at which different degrees of damage might be produced in the endpoints of interest. This will also identify gaps in scientific knowledge that could lead to further research to improve the level of understanding of these topics.

5.3.1.2. Identification based on exposure

For a suite of radionuclides, expert judgment and transfer models can be applied in order to identify which organisms assimilate and retain radionuclides to a high degree and which organisms occupy habitats that are likely to concentrate enhanced levels of radioactivity.

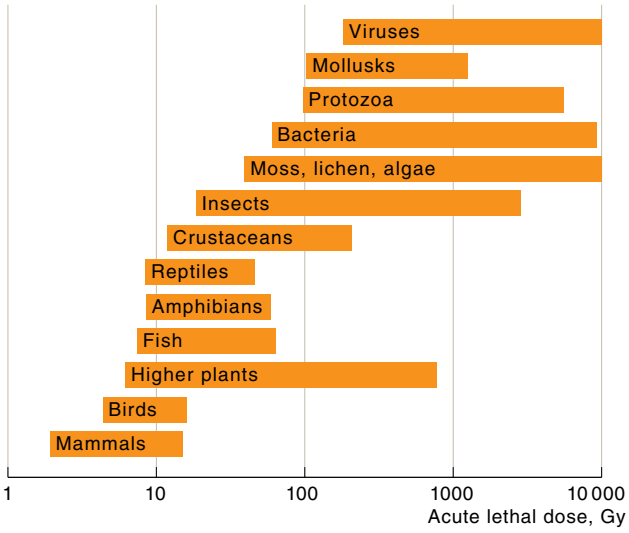

Figure 5·1. Comparative radiosensitivity of different organisms represented by the acute lethal dose ranges (UNSCEAR, 1996).

The habits and habitat of different life stages of some organisms may vary considerably (e.g., bird and egg, larval and adult insects) and this may lead to different exposure pathways. This should be considered when selecting reference organisms. A selection of candidate reference organisms for European ecosystems based purely on radioecological criteria were drawn up by Strand *et al.* (2001).

5.3.1.3. Identification based on ecological relevance

Ecological sensitivity is defined in terms of the role of the organism in the ecosystem concerned. A number of factors are relevant, e.g., population size; trophic level; reproductive strategy, including generation time; size; habitat; seasonal variations; physiological features; and biological complexity.

The *simple* approach, as used in EPIC, is to assess the requirements for representation of each trophic level. Dominant organisms at each trophic level are responsible for the major energy and nutrient flows in the ecosystems; therefore, it could be argued that protection of these organisms (by their selection as reference organisms) will ensure the protection of the ecosystem as a whole.

5.3.1.4. Identification based on radiosensitivity

The effects of ionizing radiation on living organisms have been reviewed extensively (Rose, 1992; UNSCEAR, 1996). The comparative sensitivity of different organisms to radiation in terms of acute lethal dose is shown in Figure 5·1. Although other radiation-induced effects (e.g., morbidity, fertility, and fecundity) may also be important; as a thorough review of these 'other' factors has not yet been conducted the comparative lethal dose (mortality) was used to aid the selection of reference organisms.

Available data on acute lethal dose exposures indicate that mammals and birds are the most radiosensitive groups, although the radiosensitivity ranges are large and sensitivities for different groups overlap considerably. These criteria indicate that mammals and birds should be included in any suite of reference organisms.

5.3.1.5. Distribution and practicality for research and monitoring

There is little point selecting reference organisms that are not widely distributed through at least one of the three Arctic zones (High-, Low-, and subarctic). Species known to occur in these zones, for those groups for which there is sufficient information, are listed in the EPIC report (Beresford *et al.*, 2001). The practicality of collecting the organisms for monitoring purposes (to determine the radionuclide content or to assess effects due to exposure) or to enable further radiosensitivity and radioecological studies is a further consideration. For some groups, this would be difficult owing to their protected status (e.g., raptors, marine mammals) or their perceived public sentiment (e.g., marine mammals, large terrestrial carnivores). Also, some potential reference organisms are of commercial importance, for example, macroalgae (in the Norwegian, Barents, and White Seas), benthic fish (haddock, Greenland halibut, European plaice) and pelagic carnivorous fish (Atlantic cod). Taking these factors into account, a selection of appropriate organism groups are listed in Table 5·1.

5.3.1.6. Examples of reference organisms

A search for candidate reference organisms occurred during the EPIC project. In this respect, it must be emphasized that the term 'reference organism' does not imply *a* particular species, but serves as a surrogate. Thus, in principle, it should be possible to identify specific plants and animals that are listed under the heading 'reference organism' (Table 5.1). In the practical application of the system, 'secondary reference organisms' may need to be defined at the species level. For example, in the case of a carnivorous terrestrial mammal, the Arctic fox (*Alopex lagopus*) might be selected and in the case of a marine benthos-eating bird, the common eider (*Somateria mollissima*). The selection process is driven by factors such as ubiquity and practicability for monitoring.

Table 5·1. Groups from which aquatic and terrestrial reference organisms should be selected (Beresford *et al.*, 2001).

Aquatic reference organisms	Terrestrial reference organisms
Benthic bacteria	Lichens and bryophytes
Macroalgae (marine)	Gymnosperms
Aquatic plants (freshwater)	Monocotyledons
Phytoplankton	Dicotyledons
Zooplankton	Soil microorganisms
Molluscs	Soil invertebrates
Polychaetes (marine)	Herbivorous mammals
Insect larvae (freshwater – benthos)	Carnivorous mammals
Pelagic fish (planktotrophic)	Bird eggs
Pelagic fish (carnivorous)	
Benthic fish	
Carnivorous mammals	
Benthos-eating birds	
Fish eggs	

Chapter 6

Nuclear Safety Initiatives

6.1. Introduction

This chapter considers nuclear safety initiatives relating to the eight Arctic countries. However, as many of the practices that impact upon or present a hazard to the Arctic environment are sited in northwest Russia, the emphasis of this chapter is on that region. Safety initiatives mostly relate to safety assessments of nuclear installations, particularly nuclear power plants (NPPs); other initiatives address regulatory improvements, arrangements for physical protection, and nuclear safeguards.

Production of weapons-grade nuclear materials, operation of NPPs, nuclear fuel cycle facilities, nuclear-powered ships, and other activities involving the use of nuclear energy and radioactive materials in the territory of the Russian Federation have resulted in the accumulation of significant amounts of radioactive waste and spent nuclear fuel in Arctic Russia. Their management presents a major challenge.

Nuclear safety support programs are designed to contribute to the prevention of serious nuclear accidents at nuclear facilities. Their purpose is to provide assistance to the operators of nuclear facilities and the national safety bodies that regulate these facilities. Other international programs address risks associated with nuclear waste, illicit trafficking, and terrorism involving nuclear materials. While terrorism has always been of concern to bodies such as the International Atomic Energy Agency (IAEA), interest in the wider community has been renewed following the 11 September 2001 attacks (Lubenau and Strom, 2002). The initiatives are not specific to the Arctic; however, the break-up of the former Soviet Union has meant that administrative controls and competence require strengthening to prevent terrorists obtaining nuclear material (Webb, 2002).

6.2. The purpose of risk management

Risk management is a process designed to assess, prioritize, and control risks with the specific goal of reducing risks in a manner that optimizes the use of resources and achieves the greatest reductions in risk for a given resource investment. A major fundamental underlying risk management is to ensure that planned activities, including monitoring and assessment, are formulated within the context of comparative risk. Thus, resource investments are justified on the basis of their relevance to the predominant risks or to improving the characterization of risks. The characterization of absolute and relative risks should consider both the risks posed by exposures from the planned operation of existing sources and practices and the hazards associated with proposed future sources and practices.

Owing to the stochastic nature of effects associated with low-level exposures, risk management within the context of radiological protection must deal with a number of categories of risk. The basis for radiological protection in low dose regimes is the assumption that the *risk* of effects on humans is proportional to radiation dose without the assumption of any threshold. While there is ongoing debate on the validity of this assumption (Koblinger, 2000) and on the approaches to its practical application in situations of very low level exposure (Clarke, 1999), this is the primary type of risk addressed and is of immediate relevance to existing exposures to radiation from anthropogenic sources and activities, i.e., *practices* in International Commission on Radiological Protection (ICRP) terminology (ICRP, 1991). The principles of radiological protection require that sources and practices are optimized to reduce doses to the extent achievable under the prevailing technical, social, and economic climate. Thus, optimization addresses the reduction of risks associated with operational and accidental exposures. Risk management involves the assessment of potential consequences of events at nuclear facilities that could result in additional exposure to radiation and the probability that any such event occurs. Here, the emphasis is on potential risks of exposure associated with exceptional events such as accidents at existing nuclear facilities within, or near, the Arctic.

While the risk management approach outlined in Section 6.3 concerns radioactivity from nuclear operations and activities, this approach can be used for all types of contaminants.

6.3. The approach to risk management

The first AMAP assessment identified known sources of radioactivity in the Arctic. These range from atmospheric fallout from nuclear weapons tests, past and present nuclear power reactor operations, nuclear-powered vessels, spent nuclear fuel management, and the Chernobyl accident. The presence of radionuclides in the Arctic from some of these sources will diminish with time. Nevertheless, spent nuclear fuel management and potential nuclear accidents present risks of additional exposure to Arctic populations and the environment.

In its most basic form, the risk management process consists of a sequence of steps. Namely:

- identification of hazards (in this case, current or proposed sources and practices);
- initial assessment of the risks presented by these hazards;
- identification and analysis of options for risk reduction through the imposition of preventive measures to abate risks;
- design and application of preparedness and response measures to reduce the consequences to society; and
- refinement of the selection of associated performance evaluation measures and the corresponding risk assessment.

Initial estimates of risk can be based on simple assumptions and relatively simple analyses. These warrant fur-

ther refinement through more detailed assessments if the scoping approach ranks a given risk as a major one among the various risks considered in relation to existing and potential sources and practices. Thus, substantial risks (from the various sources and practices) may require improved assessments, especially if the outstanding uncertainties are large or the scoping assessment suggests that a specific source or practice exceeds risk targets and/or regulatory protection objectives. More importantly, they may warrant intervention, or direct action, to reduce risks (either the probability of accidents or the magnitude of consequences), or other measures, such as monitoring, to provide early warning or detection of unplanned releases.

Estimation of overall risk is a convenient way of identifying those sources and activities deserving priority consideration from the perspective of risk reduction. However, risk reduction measures can never obviate the entire risk associated with a given source or practice. Commonly available options merely reduce the risk rather than removing it entirely. Accordingly, a more appropriate measure of the benefit of risk reduction measures is not the overall risk but the proportion of risk that is potentially averted by the action (i.e., the averted risk). It follows that, in setting priorities among risk reduction options, it is necessary to consider the degree to which they avert or reduce risk.

Environmental impact assessment (EIA) is also an important tool for evaluating the options for reducing risk. EIAs of the 'no action' scenario as well as options for risk reduction should be conducted prior to any decision to implement risk reduction measures. This provides a means of determining that there is an overall net benefit associated with any measure adopted and also of determining that the measure, when implemented, has the desired consequences by helping to identify and select measures of performance. EIA within the context of nuclear facilities in Norway and Russia is discussed by JNREG (2001).

6.3.1. Risk analysis

The risk management process represents an analysis of the probability and consequences of events associated with sources and practices. The elements of a risk analysis are:

- defining the facility and operation;
- identifying the hazards and determining the associated levels of risk (screening);
- characterizing the hazards that present the greatest risks;
- postulating and analyzing possible event scenarios; and
- estimating the consequences of the postulated scenarios.

A risk analysis leads to a plan for the development of risk management programs that are commensurate with each specific activity. The results of the risk analysis process are used to consider and analyze options for prevention, preparedness, and response strategies to minimize the consequences of releases of radionuclides.

6.3.2. Identification of hazards

The potential sources of radionuclides in the Arctic were identified in the first AMAP assessment. The following hazard prioritization is a ranking based on the magnitude of the potential consequences that could ensue from accidents at nuclear facilities. Namely, accidents resulting from the operation of:

- NPPs in the Arctic;
- NPPs within 1000 km of the Arctic;
- nuclear-powered vessels in the Arctic; and
- interim storage of spent nuclear fuel including improperly stored fuel elements and decommissioned vessels containing spent fuel.

For context, it should be noted that global fallout from atmospheric nuclear weapons tests, fallout from the Chernobyl accident, and previous underground nuclear device detonations continue to pose minor risks to man, plants, and animals in the Arctic through continuing exposure to radiation but that these risks are diminishing. Risks related to storage and handling of nuclear weapons have not been assessed, as no information on these issues has been made available.

Measurable, but in practice insignificant, releases of radionuclides to the environment occur during normal operation of NPPs, nuclear-fuel reprocessing plants, and nuclear-powered vessels.

6.3.3. Need for closer links between risk assessment and risk reduction activities

Risk management can only be effective when risk reduction measures are based on risk assessments. Prevention, preparedness, emergency response, and contingency strategies and plans, when based on a well-developed and well-considered risk management program, provide a basis for the optimization of risk reduction measures and options for intervention, if these are deemed necessary. Furthermore, risk management ensures that the consequences of contemplated actions are fully assessed and validated independently and against other impact assessments to provide the most appropriate measures of benefit and options for averting risk (see Figure 6·1). Communication and interaction between existing risk and impact assessment programs and programs leading

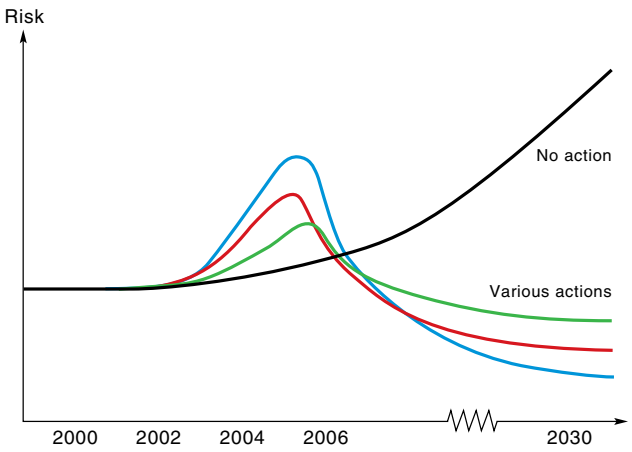

Figure 6·1. Potential risks and benefits connected with risk reduction actions.

to the formulation of actions and/or interventions to prevent accidental releases and/or to minimize their consequences is essential for decision makers in scoping and implementing risk reduction measures. This is vital to ensuring that risk reduction actions and/or interventions provide overall net benefits in terms of protection of the health and safety of workers, the public and the environment.

6.4. Nuclear power plants

Although challenges remain, especially related to the age and basic construction principles of some of the reactors, considerable progress has been made since the first AMAP assessment was completed in 1997 in improving safety assessments and introducing additional safety measures for nuclear power reactors, especially those in Russia and other eastern countries such as Lithuania (Ignalina NPP). This progress is, in large part, due to cooperation between the Russian Federation and the other Arctic countries (particularly Finland, Sweden, and the United States). This section reports progress in safety assessments and additional safety measures for NPP operations relevant to the Arctic; with links made to section 7.2 dealing with accident scenarios at land-based NPPs.

Tables 6.1 and 6.2 present an overview of the training and equipment improvements that have been made at the Bilibino, Kola, and Leningrad NPPs since the first AMAP assessment.

6.4.1. Bilibino

Bilibino NPP is located in the Chukotka region of Russia, and consists of four small (12 MW) light-water cooled, graphite-moderated reactors. Efforts at Bilibino have focused on improving the safety of day-to-day operations. This has been achieved through specific training events such as a workshop for plant engineers on the unique aspects of corrosion in cold weather environments; a training course on testing and repairing circuit boards; training on the use of ultrasonic, x-ray, and eddy-current equipment; training on the software packages SCALE and MCNP/Visual Editor (the former being a suite of criticality, neutronics, and heat-transfer codes used by the nuclear industry to support licensing submittals and the latter involving codes for criticality and shielding calculations); and provision of safety maintenance equipment, including thermography, vibration analysis, and alignment equipment.

6.4.2. Kola

The Kola NPP, in Murmansk, consists of four VVER-440 pressurized water reactors that produce 411 MW(e) each. Efforts at the Kola plant are directed primarily toward improving the safety of day-to-day operations in addition to upgrading critical plant safety systems. Projects focus on developing emergency operating instructions, upgrading the confinement system and improving other engineered safety systems. Projects are also in place to perform safety assessments, transfer capabilities for performing plant safety analyses, and provide a full-scope simulator to enhance staff training. There have also been a number of engineering upgrades specific to the plant, their purpose being to limit the spread of radioactive material in the event of an accident in Unit 2, to reduce leaks in the Unit 2 confinement system, and the installation of post-accident confinement radiation monitors. Plant safety evaluations were also carried out for internal events as well as probabilistic risk assessments and design basis accident analysis (NRPA, 2002). Safety improvements are planned until 2005.

Table 6·1. Training improvements.

	Bilibino	Kola	Leningrad
Completion of operator exchanges to train plant personnel to develop improved operating safety procedures and practices.	×	×	×
Plant instructors now trained in the 'systematic approach to training methodology' and in instructor skills.	×	×	×
A full set of emergency operating instructions that promote safety through improved accident mitigation strategies now available.		×	×
Transfer of the systematic approach to training methodology and training material developed at the Balakovo Training center to the NPPs.	×	×	×

Table 6·2. Equipment improvements.

	Bilibino	Kola	Leningrad
Analytical simulator.	×	×	
Inmarsat satellite phones.	×		
Safety maintenance equipment.	×		
Non-destructive examination equipment for evaluating pipes.	×	×	
Basic equipment such as computers, video and overhead projector facilities.	×		×
Valve-seat resurfacing equipment, a pipe lathe/welding preparation machine, and a vibration monitoring and shaft alignment system for improving safety maintenance activities			×

6.4.3. Leningrad

The Leningrad NPP is located just outside St. Petersburg and consists of four RBMK-1000 reactors of 925 MW output. At the Leningrad NPP, the focus is on improving the safety of day-to-day operations and upgrading critical plant safety systems. Specific projects include developing emergency operating instructions, providing modern safety maintenance tools and techniques, and performing in-depth safety assessments. In addition, projects are underway to provide fire detection and alarm systems in Units 1 and 2 (NRPA, 2002). Plant safety evaluations have been carried out to support the probabilistic safety assessment and full-scope in-depth safety assessment with a view to meeting Russian regulatory requirements.

6.5. Regulatory cooperation

Responsibility for nuclear safety in the Russian Federation is with the Russian regulators and operators. However, support from other Arctic countries is welcome to ensure application of best international practice and the continuous development of safety culture, as well as to satisfy international obligations, such as those resulting from the London Convention 1972 (Smith and Amundsen, 2002). Norway, Sweden, Finland, and the United States are the main contributors to regulatory improvement projects initiated by Russia.

Each of these countries has framework agreements with the Russian Federation concerning the development of protocols for regulatory and industrial projects. These help to reduce the time taken for projects to gain approval. The Joint Russian–Norwegian Working Group on Environmental Impact Assessment, the Murmansk Initiative trilateral agreement between Russia, the United States, and Norway, and the Collaboration Agreement between the Norwegian Radiation Protection Authority and Gosatomnadzor, have all been particularly prolific. Such regulatory cooperation encourages interaction between different regulatory bodies, and between the regulatory bodies and the operators; both Russian and western European (Sneve *et al.*, 2001).

A major step forward occurred with the adoption of the program 'Nuclear and Radiation Safety of Russia' for the period 2000 to 2006 (Government of the Russian Federation, 2000). This was commissioned and is coordinated by the Ministry of Atomic Energy of the Russian Federation.

The program aims at ensuring nuclear and radiation safety in an integrated manner. The primary objectives of the program include:

- dealing with the management of radioactive waste and spent nuclear material in an integrated manner;
- ensuring nuclear and radiation safety of nuclear fuel cycle facilities;
- ensuring safety in the operation and decommissioning of NPPs;
- ensuring nuclear and radiation safety during the construction, repair, and dismantling of nuclear-powered naval vessels, as well as nuclear-powered vessels and ships of the nuclear technical servicing infrastructure of the Ministry of Transport of the Russian Federation; and
- improving state radiation monitoring in the territory of the Russian Federation.

The program comprises 20 sub-programs, and includes protection of the public and the environment from the consequences of potential radiation accidents. The program will be implemented through the following activities:

- development and application of state-of-the-art technology for the safe handling of radioactive waste and spent nuclear fuel, their storage, and disposal;
- development and adoption of nuclear, radiation, explosion, and fire safety technology;
- preparation of design documentation and procedures to ensure nuclear and radiation safety during the dismantling of reactor compartments of submarines and ships, as well as in the handling of spent nuclear fuel and radioactive wastes at ship-building facilities; and
- design and establishment of a state-of-the-art and automated national system for radiation monitoring.

Social and economic benefits from the implementation of this program will arise from the improved radiation and environmental situation in and around nuclear facilities, minimization of direct and indirect economic losses caused by severe radiation accidents, and the prevention and minimization of economic losses from environmental and human exposures to radiation by taking prompt action to contain and mitigate contamination and its consequence.

6.6. Emergency preparedness

A national Emergency Response Center has been developed in St. Petersburg in addition to the Situation and Crisis Center at the headquarters of Minatom (the Ministry of Atomic Energy of the Russian Federation). All Russian NPPs, with the exception of Bilibino NPP, have direct emergency communication links to these crisis centers (see section 6.8).

NRPA has reported on the emergency response procedures in place in the Nordic and Baltic countries (NRPA, 1996). An updated report is currently in preparation. Several of the Arctic countries have well-developed regulations and emergency preparedness procedures that can be implemented should an accident or incident occur. These include methods to disseminate information, monitoring systems, and training exercises.

6.7. Waste management and risk reduction measures

There are a large number of risk reduction measures currently in place, or due to be implemented, in relation to sources of radioactive material in the Arctic. They have all been justified or supported, to a greater or lesser degree, by the type of risk analyses referred to in Section 6.3.

As a consequence of monitoring and assessments on the state of the environment in northwest Russia in 1995, five major projects relating to the prevention of radioactive contamination and a number of actions to address existing problems have been identified (NEFCO, 1996).

Since 1996, several sub-projects have resulted in significant risk reductions to the population and the environment. Some have been undertaken through the Nordic Environment Finance Corporation, while others have been addressed and funded by other international bodies and collaborations. The Contact Expert Group, set up under sponsorship by the IAEA, has facilitated international collaboration (CEG, 2002). The remainder of section 6.7 details some of the major projects that involve facilities other than NPPs.

6.7.1. Rehabilitation of the Murmansk RADON center

The Russian RADON interim storage for low and intermediate level radioactive waste located in the Murmansk area ceased operation in 1993 because it did not meet Russian quality requirements. Decommissioning of this facility with European Union assistance is now being considered. Recently, a proposal for a regional interim storage facility sited at the NERPA dockyard with the capacity to store all conditioned low and intermediate level waste from the Murmansk region, including that from the RADON facility, has been completed.

6.7.2. Submarine spent fuel management in northwest Russia

Under a bilateral assistance program to help tackle nuclear related clean-up in northwest Russia announced by the United Kingdom in February 1999, assistance is being provided for the management and interim storage on land of spent nuclear fuel from decommissioned nuclear submarines. This involves the creation of an interim storage facility for spent nuclear fuel comprising a storage pad for up to 50 casks and a number of certified Russian 40 t dual-purpose casks at either NERPA or Polyarnyi, two Russian shipyards.

6.7.3. Improved reprocessing facilities at Mayak

All reprocessable naval spent fuel should be sent to the Mayak reprocessing facility. However, current storage facilities are full and the lack of interim facilities has created a bottleneck in the decommissioning program. The European Commission, France, Norway, Sweden, Russia, and the United Kingdom collaborated in a study to investigate three possible interim storage options. The chosen option was a new dry store and additional interim storage for the spent nuclear fuel casks on-site. The project is due to be funded solely by the United States as part of the Co-operative Threat Reduction program. Other projects relating to improvements at Mayak are being funded by European countries and the European Commission.

6.7.4. Treatment of liquid radioactive waste

This project involves the construction and deployment of mobile processing facilities to decontaminate and reduce the volume of liquid radioactive wastes. The intention is to site treatment plants at Severodvinsk and in Snezhnogorsk (NERPA). These plants are based on a cementation process and are intended to be mobile and transportable by sea. Particular emphasis is placed on the processing of liquid wastes from the decommissioning of nuclear-powered submarines.

6.7.5. Atomflot

There are three consortium projects with Atomflot and the Russian Northern Fleet for the treatment of liquid radioactive waste with permanently-sited and moveable equipment. Trilateral collaboration between Norway, the United States, and Russia has been particularly successful in the expansion and upgrading of facilities at Atomflot. A notable success is the inclusion of quality assurance procedures in Russian methods and the use of environmentally friendly technology during implementation. The start of operation of the purification plant, however, has been seriously delayed, and in 2003 it was still not operational.

In addition, the Finnish NURES system for purifying liquid radioactive wastes has been successfully used at Atomflot. It has been proposed for use in a Norwegian–U.S.–Russian project to deal with military wastes in Murmansk although progress has been delayed by access restrictions.

6.7.6. Repository at Novaya Zemlya

A Russian-lead project developed designs for a low to medium level waste repository in the permafrost of Novaya Zemlya. The technical designs were peer reviewed by several international organizations, under the coordination of the European Commission. There was widespread support for the facility although more detailed safety assessments were required. Early in 2002, Russian designs for the repository were approved by the Ecological Expert Commission and are currently awaiting approval from the State Committee for Environmental Protection (Goscomecology). Following approval, detailed design and construction plans can be made. Large-scale international finance is required to implement the project as the estimated cost of such a facility is US$ 70 to 90 million.

6.7.7. Andreyeva Bay

At Andreyeva Bay there are 21000 spent fuel elements from the Northern Fleet's decommissioned submarines stored in three concrete tanks. These tanks are in very poor condition and the spent fuel elements need to be recovered. In 2001, a Norwegian–Russian bilateral agreement resulted in the initiation of several projects. Engineering infrastructure improvements and feasibility studies have been established and the main tasks planned involve the stabilization of current spent nuclear fuel storage units, treatment or removal of liquid radioactive waste, conditioning of solid wastes and their removal to a regional store, and decontamination and final remediation of the site.

6.7.8. The *Lepse*

The *Lepse* is a decommissioned service vessel of the Russian icebreaker fleet that is docked in Murmansk and used as a storage facility for spent nuclear fuel and other

radioactive wastes. The ship is in a very poor state of repair and there has long been a desire to offload and transfer the radioactive wastes and damaged spent fuel to land-based storage.

Since the first AMAP assessment, there has been little progress in the work to decommission the *Lepse*. However, the Murmansk 80 t Cask Project, which will provide transport and interim storage for spent nuclear fuel from Russian nuclear submarines and icebreakers currently stored on barges and service vessels, and in a low-level radioactive waste treatment facility in Murmansk, is addressing the transfer to storage of the spent fuel that is not suitable for processing owing to its damaged state. A cooperative venture between Norway, Sweden, and Gosatomnadzor (Russia's State Committee for Supervision of the Safety of Work in Nuclear Power Engineering) is tasked with identifying means of dealing with the wastes stored on the *Lepse*.

The results of Phase 1 of the Lepse Regulatory Project were published in April 2001 (Sneve *et al.*, 2001). The main results were a set of three regulatory guidance documents and increased mutual understanding of the differences in the regulatory systems and processes for licensing nuclear activities in the Russian Federation compared to other western countries, notably Sweden, Norway, and the United Kingdom. The guidance documents provide specifications for:

- documentation to substantiate nuclear and radiation safety assurance measures for submission by operators when applying for a license from Gosatomnadzor to implement the *Lepse* Project, as described by the NRPA (2001);
- the quality assurance program for unloading spent fuel assemblies from the *Lepse*; and
- the safety analysis report required to support a license application for unloading spent fuel assemblies from the *Lepse*.

This regulatory guidance is intended to help focus on safe implementation. In addition, considerable emphasis is being given to EIAs and their role in determining the suitability of specific mechanisms for unloading spent fuel from the *Lepse*. Phase 2 of the Lepse Regulatory Project will comprise the review of license application documents submitted to the appropriate Russian authorities, primarily the Gosatomnadzor.

6.7.9. Environmental impact assessments of other hazardous Russian facilities

A working group under the Joint Norwegian–Russian Expert Group for the Investigation of Radioactive Contamination of Northern Areas compared EIA systems in Russia with those in Norway and other western countries (JNREG, 2001) and concluded that the principles and methods used in Norway and Russia are broadly similar. They are based on the common principles of prevention, openness, and obligation to conduct EIAs for all projects likely to significantly influence the environment. Concerns have been expressed however about the degree to which transboundary impacts are considered under Russian procedures and the lack of attention to the effects of ionizing radiation on fauna and flora.

The working group also noted that, in the planning phase of projects having potential radiation hazards, close contact between the developer and the government bodies responsible for health protection, environmental protection, and nuclear safety is essential. It is important that those undertaking EIAs are well informed about the information required and the system for approving planning activities. This ensures the overall aims of EIAs are met; namely selection of the optimum location, appropriate technology, and methods for the protection of human health and the environment.

6.8. Alarm, notification, and radiation measurement systems in northwest Russia

Radiation monitoring in the Arctic is of great importance because Russia is the largest country in the region and operates many relevant sources and practices. A major area of work for AMAP involves risk and impact assessment, including monitoring systems. Much of this occurs within the context of a general Barents region environmental and human health monitoring system. There are also plans for a risk and impact assessment for workers and members of the public that may be affected by military and civilian sources; development of a monitoring system for environmental releases of radioactivity from such sources; provision of an emergency and monitoring system in the Archangelsk Oblast; and construction of a regional laboratory for surveillance and early warning systems. The first AMAP assessment provided useful input to these developments.

In 1992, the Finnish Radiation and Nuclear Safety Authority (STUK), in cooperation with Gosatomnadzor, installed push button alarm panels and satellite communication systems in the site offices of Gosatomnadzor at the Leningrad and Kola NPPs and at the Atomflot Repair Technical Plant near Murmansk. These facilitate the prompt transmission by Gosatomnadzor local safety inspectors of a selected pre-programmed emergency or incident telex message. These can be transmitted to the 24-hour emergency response systems of STUK, other Nordic countries, and the Emergency Response Center in Moscow operated by the Federal Nuclear and Radiation Safety Authority of Russia. The notification system is independent of local ground communications and has battery back-up to ensure continuous operation. It is also tested automatically each week and manually each month to all Nordic receivers and to Moscow. There has been no actual emergency use of this system since its installation.

In 1994, eight environmental monitoring stations of Finnish origin were installed on the Kola Peninsula. These operate under local supervision and without automatic connections to the central system at Roshydromet in Murmansk for their data acquisition and alarm systems. Data collection is manual and the data are transmitted by telephone and telex. Reliable automatic operation of these stations would be difficult as the local telecommunications environment is prone to interference and other disturbances. In 1998, STUK and the NRPA signed a joint agreement on the development of the Roshydromet environmental radiation measuring system. Radiation monitoring stations would be up-

graded and the telecommunication connections enhanced. Progress on this project is conditional upon the conclusion and implementation of a general agreement on this work between Norway and Russia.

In 2000, the nuclear and radiation safety authorities in the Nordic countries signed a framework agreement concerning joint Nordic financing for upgrading alarm and notification systems.

6.9. Security (including physical security)

Safety and security of radiation sources has acquired a new significance since the terrorist attacks in the United States on 11 September 2001. Special security measures to protect against terrorism should be part of safety assessments (Lubenau and Strom, 2002). There are a number of orphaned sources (i.e., sources that are no longer under regular institutional control) in the Russian Federation that should be located and brought back under institutional control. The European Commission and the United States are funding programs to do this.

A 'safeguard' is generally understood to be a method for controlling fissile/fissionable material. Six of the eight Arctic countries have signed IAEA safeguard agreements to contribute to non-proliferation obligations. Safeguard support programs have constituted the primary means of bilateral Finnish assistance to the Ukraine, the Baltic States, and the Russian Federation. Their objectives are to assist in establishing and improving national systems for accounting and control of nuclear material. The relevant regulatory bodies are assisted in the development of regulations, guides, and inspection procedures. Training has also been extended to border-control authorities in the detection and control of radioactive and nuclear materials. Training courses were organized for the Russian border controls using experts from the Finnish Radiation and Nuclear Safety Authority (STUK) and from other institutes within the European Union (STUK, 2000).

A bilateral Russian–Norwegian project was started in 1998 to replace the radioisotopic power sources at four Russian lighthouses in Varanger Fjord by solar powered technology. The aim is to reduce the likelihood of radioactive contamination of the northern marine environment. When the project is complete, all radioisotopic power sources in the Russian parts of Varanger Fjord will have been replaced by solar panels. A Russian information video has been made in connection with this project. The radioisotope thermoelectric generators will be stored at Atomflot before transport to the Minatom Institute for Technical and Atomic Physics and then to Mayak for final treatment and storage.

6.10. Conclusions

The main criterion of success for a nuclear safety project is its net contribution to the improvement of nuclear safety (NRPA, 2002). Owing to the difficult economic situation in Russia, improvement initiatives in the region are often only possible through international collaboration. Lack of funds and/or difficulties in developing bilateral/multilateral agreements can delay the start of nuclear safety initiatives; nevertheless, the Arctic countries are committed to further improvements. Priorities for risk reduction are being identified through a process of risk analysis. In addition, projects are being supported only within the context of demonstrated compliance with Russian regulatory requirements. That context includes safety assessments and EIAs incorporating a variety of risk analyses to demonstrate compliance with risk objectives relating to environmental and human health protection. Risk assessments and EIAs should also be used to select and/or prioritize risk reduction projects, to optimize the use of resources. Resources and effort will continue to be focused on the areas of greatest risk and on the operations and facilities that pose the greatest potential threats.

Chapter 7

Potential Accident Scenarios

7.1. Introduction

This chapter considers the nature and scale of consequences arising from potential accidental releases of radioactivity into the Arctic environment from sources under human control. The sources considered to warrant accident assessment are those described in Chapter 2. All scenarios are described in more detail in the relevant literature. Relevant accidents and subsequent impact assessments are reviewed and, where possible, extrapolated to provide perspectives on the consequences of accidents associated with other sources. The chapter concludes with recommendations for further impact assessments.

Scenario analysis begins with a consideration of the different possibilities for sequences of events and processes (such as containment failure mechanisms) that can lead to radionuclide release. These depend upon the specific management and engineering features of the facilities under consideration. The release mechanisms and characteristics are important determinants of the environmental and human health impacts. Relevant variables might include: isotopic composition; amounts of each isotope released; physical-chemical form of release (gas, solution, aerosol, etc.); time development of the release; release point and plume height; and the energy content of the release.

The scenarios discussed here were not necessarily developed for the same purposes as the AMAP assessment. They are illustrative of selected aspects of the possible consequences of radionuclide release rather than representative of comprehensive risk or scenario analyses for the Arctic.

The radionuclide release information provides input to a radionuclide transport model that is used to predict the subsequent environmental distribution of contamination. For accident scenarios resulting in releases to the atmosphere, the assumed (or actual, if assessing the consequences of past accidents) meteorological data are very important as they can radically affect the degree of atmospheric dispersion. Similarly, for releases to the aquatic environment, the hydrodynamic characteristics of the receiving environment are equally important. Distribution following releases to the ground are strongly dependent upon surface geology and hydrology. The radiation characteristics of the radionuclides and their environmental mobility are also important determinants of the magnitude of the consequences following release. The receiving environments themselves also influence the scale of the consequences, since some are more susceptible to incorporating radionuclides into human exposure chains than others, as discussed in Chapter 4 (NRPA, 1999; Skuterud *et al.*, 1999). Radiation doses to humans and other biota are assessed using assumptions about the ways in which they interact with contaminated media. Finally, the impact on human and environmental health is assessed using assumptions relating radiation doses to health impacts. Such information is essential for risk management.

7.1.1. Risk management

Risk management includes the analysis of accident scenarios and consequences and, where possible, an assessment of the probabilities of accidents and their consequences. Sources may occur within the Arctic, in which case these 'point sources' require analysis, and outside the Arctic. The potential for accidents occurring outside the Arctic to contaminate the Arctic environment depends on the dispersal characteristics.

Generally, the larger the inventory of radionuclides, the greater the hazard. In most cases, the inventory in Becquerels (Bq) is well established owing to the application of well-proven technology and associated regulatory requirements. However, in some cases, the information may be less complete, as for example, in the case of old waste storage facilities for which information is limited.

The risks associated with hazardous sources may also be modified by measures to control the source term. That is, consideration of the risks must address both the scale of the consequences and the likelihood of their occurrence. Thus, while a very large source term may present the greatest hazard (potential for harm), measures to reduce the chances of release may reduce the risks to a tolerable level (HSE, 1988). Nuclear safety initiatives to reduce the likelihood of accidents are discussed in Chapter 6. Other ways of reducing risk include measures to reduce the consequences of potential accidents. Risk management must account for both. A simple example is the case of spent nuclear fuel, which is a significant radioactive source term. Left on the surface with limited containment, the chance of releases into the environment, before radioactive decay has reduced the hazard significantly, whether as a result of waste container degradation or by human sabotage, is relatively high. Deep disposal is considered to reduce the risks by reducing the likelihood of gross and acute environmental releases. See, for example, discussions concerning high level waste disposal in Japan (JNC, 2000).

Another aspect of risk management is the introduction of measures to mitigate the impact of accidental release. International guidance concerning countermeasures is provided by the International Commission on Radiological Protection (ICRP, 1992). Examples include evacuation, advice to remain indoors, and the distribution of iodine tablets. Interventions should be based upon evaluations of their benefit, expressed as averted doses, and disbenefits, especially those of an economic or social nature. Assessments of the consequences of accidents should take account of the planned emergency response in the early and latter phases of the accident; in the long-term the effects of clean-up measures may be important, see Brown *et al.* (2000).

7.1.2. First AMAP assessment

The first AMAP assessment concluded that of greatest concern were the possible accidents associated with: nuclear power plant (NPP) operation; nuclear weapons handling and storage; decommissioning of nuclear submarines; and the management of spent fuel from nuclear-powered vessels.

Consideration is given to the consequences of each of these types of accident. Also, to additional or modified potential accident sources, particularly reactors in sunken submarines such as the *Kursk* (Amundsen *et al.*, 2002a) and the management of damaged spent fuel, as stored on the *Lepse* (NRPA, 2001).

7.2. Land-based nuclear power plants

Operational land-based NPPs in the Arctic include the Kola and Bilibino NPPs. The Kola plant comprises four VVER-440 pressurized water reactors each with a design output of 1375 MW(th) and 411 MW(e). The Bilibino NPP is located in the Chukotka region in eastern Russia and comprises four light-water cooled, graphite-moderated reactors each of output 62 MW(th) and 12 MW(e).

Owing to design differences, a direct comparison of the risks posed by the Kola and Bilibino NPPs is not straightforward. Risk assessments need to include a consideration of the engineered features and management at the respective plants, but for assessing significant releases, there are obvious differences owing to the reactors at the Kola NPP being more than 20 times larger.

The power plants on the Kola Peninsula clearly represent the major potential reactor accident source within the Arctic.

7.2.1. Accident scenarios and consequences for the Kola NPP

The Kola NPP is located in Murmansk Oblast in northwest Russia and severe accidents at the site have the potential to substantially contaminate both northwest Russia and northern Fennoscandia. Studies by Stokke (1997), including a review of the Kola reactor safety systems, have provided detailed information on the Kola plant and its reactor inventories.

7.2.1.1. Initiating events

Initiating events that may lead to core melt sequences in pressurized water reactors are generally grouped into three classes: loss of coolant accidents (LOCAs), transients, and common cause initiators (CCIs).

LOCAs may be initiated by large leaks or breaks in the primary circuit, which in turn may be caused by mechanical failure (such as pipe breaks, fire, and corrosion) resulting from poor maintenance. The loss of cooling may take place early in the sequence or at a later stage. Early loss of cooling is potentially the most dangerous as it gives little time to re-establish cooling and because significant decay of short-lived radioisotopes will not have occurred.

Transients can be failures in power supply, reactivity transients (sudden increases in reactivity), failures in control systems (e.g., control rod ejection), and loss of flow.

CCIs (e.g., power transients and earthquakes) lead to multiple failures and may affect several components in the system.

According to Stokke (1997), loss of coolant in pressurized water reactors does not immediately signify a large radioactive release. The vessel should be able to contain an overheated core for a period that may be sufficient to allow restoration of adequate core cooling. If there is extensive core damage, it is unavoidable that radioactivity leaks occur. A core melt by itself, however, does not create an explosive situation unless reactor containment fails. A source term for the Kola NPP having a very high radioactive plume rise and thus exposure of core and fuel to the open air has therefore a low probability. Nevertheless, releases of noble gases and volatile radioactive compounds should be expected in a severe core damage accident.

There are differences between the two older and the two newer plants. For the older Model 230 reactors, the effectiveness of the confinement structure in containing the radioactive steam–gas mixture after a LOCA is uncertain. The airtightness of the confining structure is not assured and there may be considerable leakage even without open valves or other penetrations. Breaks in the largest coolant pipes may generate a steam pressure that could crack or rupture the confinement structure and create an open passage from the core to the environment, although the reactor vessel would still be intact (Stokke, 1997).

7.2.1.2. Probabilities

The probability that a severe accident may occur is dependent on many factors such as design features, construction quality, and human performance. The probability that an event may lead to an unintentional core melt can be assessed on the basis of engineering judgment or by performing a Probabilistic Safety Assessment where, in principle, all realistic chains of events leading to core melt are analyzed and their probabilities of occurrence calculated. The sum of all probabilities for all possible initiating events to cause a core melt is the Core Melt Frequency (CMF), given as the probability per reactor operating year. The CMF does not include the probability of human failure, sabotage, or terrorist attack (Stokke, 1997). For modern NPPs, the CMF is considered to be within the range 10^{-4} to 10^{-5}. At present, there is no CMF for the Kola NPP for use in accident consequence analysis. However, the International Atomic Energy Agency (IAEA) has reported a preliminary estimate of 5.5×10^{-3} per year for the oldest Kola reactors (Stokke, 1997).

7.2.1.3. Accident source terms

Accident source terms depend on the initiating events, which may result in different accident scenarios. Examples of scenario development are provided by Stokke (1997). Worst case scenarios concern situations in which the reactor core contains the maximum number of products and maximum activity concentrations of radionuclides at the end of the normal fuel burn-up cycle.

Table 7·1. Combined inventory of radionuclide groups with release potential in a VVER-440 Kola NPP (Stokke, 1997).

		Total activity in core, Bq
Noble gases	^{85m}Kr, ^{87}Kr, ^{88}Kr, ^{89}Kr, ^{90}Kr, ^{133}Xe, ^{135}Xe, ^{138}Xe	1.21×10^{19}
Halogens	^{84}Br, ^{87}Br, ^{131}I, ^{133}I, ^{134}I, ^{135}I	1.48×10^{19}
Alkaline metals	^{86}Rb, ^{88}Rb, ^{89}Rb, ^{90}Rb, ^{134}Cs, ^{136}Cs, ^{137}Cs, ^{138}Cs	7.53×10^{18}
Tellurium group	^{127m}Te, ^{127}Te, ^{129}Te, ^{129m}Te, ^{131m}Te, ^{132}Te, ^{127}Sb, ^{129}Sb	3.90×10^{18}
Alkaline earth metals	^{89}Sr, ^{90}Sr, ^{91}Sr, ^{140}Ba	6.42×10^{18}
Transition metals	^{90}Y, ^{91}Y, ^{95}Zr, ^{97}Zr, ^{95}Nb, ^{99}Mo, ^{99m}Tc, ^{103}Ru, ^{105}Ru, ^{106}Ru, ^{105}Rh	2.14×10^{19}
Lanthanides	^{140}La, ^{141}Ce, ^{143}Ce, ^{144}Ce	9.57×10^{18}

Table 7·1 summarizes information on the relatively mobile radionuclides present at the end of the fuel burn-up cycle that are most likely to be released in the event of an accident.

There is currently no information on the inventory of actinides for the Kola NPP and so the consequences of actinide release have not been assessed. Stokke (1997) has estimated the highest release fractions for various potential accidents from the Kola reactors (Table 7.2).

Table 7.2. Highest release fractions (%) from the core inventory for different initiating events (Stokke, 1997).

	VVER-440/213			VVER-440/230
	Transient	Small LOCA	Large LOCA	Large LOCA
Noble gases	100	20	50	100
Iodine	2.5	0.05	1	15
Cesium	2.5	0.05	1	12
Tellurium	0.1	0.05	0.2	10
Strontium	1	0.1	1	2
Barium	0.5	0.05	0.5	2

For a given event, the estimated release for a single nuclide is calculated by multiplying the amount of the nuclide in the core by the release fraction. The source term suggested for the VVER-440 230 model (Stokke, 1997) is based on source terms applied in earlier consequence assessments of accidents at the Kola NPP. The source term for the VVER-440 213 model is based on the IAEA Technical Co-operation Project on Evaluation of Safety Aspects for VVER-440 model 213. Because all aspects of a potential accident are not yet completely understood, a conservative approach should be taken so as not to underestimate the risk. The source terms for model 230 reflect a conservative approach that results in source terms that are larger than most other source terms previously applied for VVER-440 reactors.

Releases of noble gases, radioiodine, and radiocesium are the most important, as these are assigned the highest release level (level 7) on the International Nuclear Event Scale (INES) for this reactor type. Releases of radioiodine, radiocesium, and radiostrontium are important from a radiological hazard point of view, while the long-term consequences of much smaller releases of actinides are also significant. The inventory estimates in Table 7·1 and fractions released in Table 7·2 are assumed to be valid for both VVER reactor types. Various accidental release scenarios have been considered, examples of which are given in Sections 7.2.1.4 to 7.2.1.7, for an unintentional 'worst case' scenario (i.e., where the accident that is not a result of malicious intent, e.g. terrorism) with a large LOCA and a less severe transient scenario. Table 7·3 shows the inventory, fraction released, and consequent activity emitted to atmosphere for the two scenarios. A major fraction of the radionuclides, including Cs- and Sr-isotopes will be present as particles. Since the air dispersion and transfer model does not currently include radioactive particles, these are not considered in the estimated ecosystem transfer and doses to humans.

7.2.1.4. **Initial dispersion**

The dispersion of radionuclides from a source depends on the release height and meteorological conditions at the release site and along the transport route, in addition to the properties of the released material such as size distribution and the degree of volatilization. Buildings and other structures near the release point can also affect the initial dispersion, especially in the case of releases at low height. This effect, however, becomes insignificant at distances over a few kilometers from the source, and is negligible for releases with a high effective release height. Scenarios for a hypothetical release from the Kola NPP are based on data from the Norwegian and Danish Meteorological Institutes.

Table 7·3. Calculated release fractions for selected radionuclides under a large LOCA and a transient scenario (Larsen *et al.*, 1999; Stokke, 1997).

	Inventory, PBq	Fraction released, %		Activity released, PBq	
		LOCA	Transient	LOCA	Transient
^{137}Cs	117	12	2.5	14.0	2.9
^{134}Cs	156	12	2.5	18.7	3.9
^{90}Sr	85	2	1	1.7	0.9
^{132}Te	2240	10	0.1	224	2.2
^{132}I	2330	15	2.5	233	57.5
^{131}I	1570	15	2.5	236	39.3
^{103}Ru	2350	1	0.1	23.5	2.4
^{140}Ba	2790	2	0.5	55.8	55.8
^{140}La	2860	0.2	0.1	5.7	2.9

The information was supplemented by a study of the meteorology and transport of radioactive contamination from the Kola NPP (Bartnicki and Saltbones, 1997; Saltbones *et al.*, 1997). The Norwegian Meteorological Institute defined the meteorology in the area, calculated the transport times, and investigated the likelihood of nuclear contamination at specific locations. Three scenarios were selected as initial conditions for the dispersion model (Saltbones *et al.*, 1995). The scenarios used three sets of weather situations that would provide particularly unfavorable consequences for Norway. Two of the three situations were relevant to Arctic Norway.

Scenario A, with a rapid transit time to northern Norway, where one-eighth of the released material was deposited within 72 hr.

Scenario B, with precipitation during the passage of the plume over northern Norway, with wet deposition such that nine-tenths of the release was deposited over Norwegian territory.

7.2.1.5. **Consequences**

There have been recent assessments of both the short-term and long-term doses from hypothetical accidents at the Kola NPP. The analysis of short-term doses was confined to Norway and to external and inhalation doses. The short-term analysis did not include ingestion doses based on the assumption that the accident occurred outside the short growing period, when dairy animals would be housed; a situation which prevails for most of the year in the Arctic. The long-term doses were estimated for northern Norway and for various regions of northwest Russia and considered external and ingestion doses arising from the mobile, long-lived radionuclides, radiocesium and radiostrontium only. For both assessments, the unintentional worst-case accident was selected for an assessment of radiation levels and doses.

7.2.1.6. **Short-term assessment**

The assessment included more than 50 radionuclides. The dispersion model results were based on the release of particles with a given mathematical mass and must therefore be combined with information about the radioactive emission in order to calculate the atmospheric and ground levels of the various radionuclides. Activity concentrations and ground deposited activity were used as a starting point for calculating doses from the various exposure pathways associated with inhalation and external irradiation from radionuclide deposits on the ground and in the air.

Although foodstuffs are the major contributor to the total long-term dose, the focus of the short-term consequence assessment was on external exposure and inhalation. There is a very short growing season in the Arctic and the implicit assumption is that the accident occurred outside this period.

For an adult, the effective dose was calculated at about 1 mSv for Scenarios A and B. In both cases, the contribution from radionuclides deposited on the ground predominates, especially where precipitation is high and wet deposition considerable.

Throughout the first year, external irradiation from the ground is the most important exposure pathway: calculated effective doses are 3.5 mSv and 5.1 mSv for Scenarios A and B, respectively, with ^{134}Cs and ^{137}Cs the most important contributors (60 to 70%). High activity of deposited radiocesium is the main reason for the higher annual dose in Scenario B, the precipitation scenario.

External irradiation from airborne radionuclides is a relatively insignificant exposure pathway a week or more after the hypothetical accident. However, during cloud passage, airborne radionuclides, such as noble gases with very short half-lives (e.g., ^{133}Xe), could be of some significance for the dose rate.

Owing to large quantities of short-lived radionuclides emitted during the hypothetical accident, a considerable fraction of the external dose will occur during the first few days after contamination. In Scenario A, where the atmospheric transport of radioactivity occurs rapidly, about 20% of the effective external dose will result from irradiation during the first week after deposition. In Scenario B, where the transport is considerably slower, the corresponding value was estimated at about 10%. Furthermore, if radionuclide migration through soil profiles, and the subsequent radioactive shielding by overlying soil, is considered, then the external dose received from the first few days may be of even greater relative importance than that estimated.

Of the 50 radionuclides considered, only a few are significant contributors to the total external dose. Over a short time-scale (days or weeks) the dominant nuclides are ^{132}Te/^{132}I, ^{131}I, ^{103}Ru, and ^{140}Ba/^{140}La. After these decay the external dose is dominated by ^{134}Cs and ^{137}Cs.

The effective dose from inhaled radionuclides is predicted to be <1 mSv for both scenarios and is generally highest for young individuals. The dominant nuclide is ^{131}I, which contributes 50 to 70% to the total inhaled dose, depending on the scenario and age group. Equivalent doses to the thyroid gland were calculated at 4.6 to 10.6 mSv (Scenario A) and 1.6 to 3.6 mSv (Scenario B).

The doses calculated for the two scenarios are assumed to represent the worst possible consequences of a severe nuclear accident at the Kola NPP. Nevertheless, the received doses are much too low to result in any acute radiation injuries.

International guidance concerning countermeasures (e.g., ICRP, 1992), such as evacuation, staying indoors, or the distribution of iodine tablets is based on evaluations of the benefits and disadvantages of implementation, expressed as saved (averted) doses. This assessment indicates that the saving potential is too low to justify the direct implementation of countermeasures; however, this must be investigated further. The uncertainties in the calculations are large, and an evaluation of the pros and cons for an actual situation can result in the use of different countermeasures, e.g., for special groups.

7.2.1.7. **Long-term assessment**

The first AMAP assessment concluded that the vulnerability of the Arctic (defined as the relationship between dose and atmospheric deposition of nuclides) is higher than in most other areas of the world, particularly for ^{137}Cs. This reflects the transfer of radionuclides deposited from the atmosphere through terrestrial food chains to human radiation exposure. The long-term assessment estimated the long-term impact of radioactive

contamination from a hypothetical LOCA at the Kola NPP on the two northernmost counties of Norway (Troms and Finnmark), and on the Murmansk Oblast in Russia. The weather pattern for the Russian scenario was based on predicted ground deposition provided by the Danish Meteorological Institute, with most deposition occurring on the Kola Peninsula.

The study considered radionuclide deposition, transfer to and contamination of locally produced foodstuffs, and external and ingestion doses for reindeer herders and other inhabitants. A spatial model was developed within a geographical information system to predict the long-term consequences of radionuclide deposition on northern Norway and northwest Russia. As no site specific data were available, general transfer factors were used in the model (JNREG, 2002a,b,c).

External doses

The highest individual external γ-doses occur in those areas receiving most accident deposition, but are negligible compared to ingestion doses. Individual external γ-doses for reindeer herders are twice those of the other inhabitants owing to the tendency of the latter to occupy areas with higher shielding (i.e., buildings).

Internal doses

Radionuclide transfer to foodstuffs was modelled using aggregated transfer coefficients (T_{ag}; Box 3·2) and effective ecological half-lives (T_{eff}; Box 3·1). Long-term predictions were made for the spatial variation in activity concentrations in foodstuffs, individual external and ingestion doses for reindeer herders and other inhabitants, and radionuclide fluxes (total Bq output from contaminated land areas over specified time periods).

Data were collated for each study area to derive area-specific T_{ag} and T_{eff} values for radiocesium and ^{90}Sr (JNREG, 2002a). The biggest difference was the 3-fold higher T_{ag} value for ^{137}Cs transfer to reindeer meat for the Murmansk Oblast compared to Norway, and the longer associated half-life. Together, these were responsible for the greater intakes predicted for radiocesium in reindeer meat per unit deposition, and the greater persistence in reindeer meat and thus Russian reindeer herders. In addition, the T_{ag} value for ^{137}Cs transfer to potato and to a lesser extent berries, was lower for Norway than Russia. T_{eff} values for freshwater fish were lower for Norway than Russia. The T_{eff} value used for ^{90}Sr in milk was much greater for Russia than that assumed for Norway. For ^{90}Sr, T_{ag} values for Russian dairy products and potatoes were lower than for Norway, while those for most other products were higher.

The most obvious difference between the diets of the Norwegian and Russian inhabitants is in the consumption of dairy products; these are important in Norway but much less so in the Murmansk Oblast. Reindeer meat consumption is highest in the male reindeer herders in Lovozero in Russia. Sheep and goat meat is only consumed in Norway. Potato and freshwater fish consumption is also greater in Russia.

Under the scenarios considered, high activity concentrations persist in foodstuffs owing to the high T_{eff} values. Activity concentrations for ^{90}Sr in foodstuffs are much lower than for radiocesium. In the first year after accident deposition, the highest radiocesium activity concentrations were predicted to occur in reindeer meat, sheep meat, mushrooms, and berries, and the highest ^{90}Sr activity concentrations in berries and potatoes. After fifty years, the highest activity concentrations predicted for foodstuffs were for ^{137}Cs in mushrooms, reindeer meat, and berries.

As for foodstuffs, predicted annual individual ingestion doses for reindeer herders and other inhabitants vary spatially according to differences in deposition and land cover. Annual ingestion doses for all population groups in the first year after deposition were predicted to exceed 1 mSv. Annual individual radiocesium ingestion doses for reindeer herders are significantly greater than for other inhabitants. In the first year after deposition, the most significant contributor to annual individual radiocesium ingestion dose is reindeer meat for all population groups, with the exception of other Norwegian inhabitants for whom dairy products and mutton are important contributors. Potatoes and dairy products are the largest contributors to the much lower annual individual ^{90}Sr ingestion doses for all population groups. Berries are another important ^{90}Sr contributor to the two Russian population groups, while reindeer meat is also a source of ^{90}Sr for Russian reindeer herders.

Under all accident scenarios, reindeer herder annual ingestion doses are predicted to exceed 1 mSv for many decades after accident deposition (and are much higher in the first few years); for the other population group, ingestion doses exceed 1 mSv for a few years after accident deposition in northern Norway and for a decade in Murmansk Oblast. Fifty years after accident deposition, individual ^{137}Cs ingestion doses for reindeer herders are over two orders of magnitude lower than during the first year; those for the other population group are more than 30 times lower. The largest contributors to annual individual ^{137}Cs ingestion doses for Norwegian reindeer herders fifty years after accident deposition are reindeer meat, freshwater fish, and dairy products, with dairy products, freshwater fish, mushrooms, and reindeer meat the most important contributors to the other Norwegian population group. Reindeer meat and mushrooms are the largest contributors to annual individual ^{137}Cs ingestion doses to the Russian population groups 50 years after accident deposition.

Sr-90 is a much less important contributor to ingestion dose and the predicted consequences of the accident scenarios are much less certain owing to the paucity of relevant data for the Arctic, in particular for milk. For reindeer herders, freshwater fish, potatoes, berries, and reindeer meat, provide the largest contribution to annual individual ^{90}Sr doses, while potatoes, freshwater fish, and berries, are the most significant contributors for the other inhabitants.

The most significant contributor to total doses for all population groups is radiocesium ingestion. Vulnerability to ^{90}Sr contamination is much lower than to radiocesium for both reindeer herders and other inhabitants.

There are substantial differences in agricultural production within the various areas of northern Norway. Production of almost all agricultural products in Troms is 2- to 5-fold higher than in Finnmark, whereas reindeer production is 20-fold higher in Finnmark where most of reindeer herders live. Detailed production data were not available for Murmansk Oblast. Annual radionuclide

fluxes have been predicted for all locally grown foodstuffs (production of mushrooms, berries, and freshwater fish was estimated by multiplying diet and population). In the first year after deposition, the highest radionuclide fluxes are predicted to coincide with the areas receiving the highest accident deposition. The largest contributors to radiocesium fluxes are reindeer meat and dairy products, while dairy products and potatoes are the largest contributors to annual ^{90}Sr fluxes. The contribution of different foodstuffs to radionuclide fluxes changes with time. Fifty years after accident deposition, the highest radionuclide fluxes do not necessarily occur in those areas receiving the greatest accident deposition. High radionuclide fluxes can occur in areas with high food production. In general, reindeer meat and dairy products remain the significant contributors to ^{137}Cs fluxes in the fiftieth year, while berries, potatoes, and freshwater fish are the largest contributors to the lower annual ^{90}Sr fluxes.

This study confirms the outcome of the first AMAP assessment, i.e., that Arctic residents are particularly vulnerable to radiocesium contamination and that the vulnerability would persist for many years after deposition. Reindeer herders are particularly vulnerable due to their higher levels of reindeer meat consumption. Nevertheless, other inhabitants of northern Norway and Russia would also be potentially exposed to high doses, especially if consuming many local products. While reindeer production is the most vulnerable pathway, freshwater fish, lamb meat, dairy products, mushrooms, and berries are also vulnerable foodstuffs. Although game was not included in this study, post-Chernobyl studies show high and persistent contamination of some game animals.

The location of communities and their types of agricultural production are important variables determining vulnerability; if high deposition occurred in the major reindeer production areas (Finnmark in Norway and Lovozero in the Murmansk Oblast) the impact would be much higher than if deposition occurred in areas where other types of agriculture predominated. Conversely, because dairy cattle are inside for much of the year, vulnerability increases if an accident occurs during the short summer grazing period, especially for ^{90}Sr.

Major factors contributing to the uncertainties in the estimates of doses and fluxes are the limited number of nuclides being considered, as well as the use of general rather than site specific transfer factors. Also, the scenarios address releases of gaseous and aerosol components but potential releases of radioactive particles are not taken into account. The effects of countermeasures were not evaluated in this assessment. Doses and fluxes were predicted assuming no mitigating actions having been taken. However, the results clearly indicate the need for an effective emergency response, including the application of countermeasures, should an accident of the scale considered in this assessment ever occur at the Kola NPP.

7.2.2. Barents region environmental center study of atmospheric transport pathways from the Kola NPP

An assessment of atmospheric transport pathways from the Kola NPP was undertaken for four geographical regions: Scandinavia, Europe, the central former Soviet Union (CFSU), and the Taymir Peninsula. Several approaches were used to determine the probability that air would be transported from the Kola NPP to each of these regions, transport times, and seasonal variations in atmospheric transport.

The assessment indicated that Scandinavia would be affected by a release for 44.5% of days between 1991 and 1995, Europe for 8.1%, the CFSU for 43.2%, and the Taymir Peninsula for 55.5%. The airflow probability field had a similar pattern to that for the one available assessment of the consequences of hypothetical accidents at the Kola NPP, where the released materials were distributed in an almost circular pattern extending slightly in a northeasterly direction (Baklanov *et al.*, 2002). Seasonal variations influenced the transport pattern.

Two cases of rapid transport from the Kola NPP to Scandinavia were selected for more detailed study. In both scenarios, 60 PBq of ^{137}Cs were released over a period of 20 hr in a plume rising 400 to 600 m. The areas contaminated by ^{137}Cs to a level exceeding 30 kBq were 190 000 and 250 000 km^2.

Mean individual doses, collective doses, and collective risks were calculated for one of the two scenarios, based on assumptions of the relative importance of various nuclides and exposure pathways to the total dose resulting from the effects of the Chernobyl accident on Scandinavia. The highest mean individual doses, 1.15 mSv, occurred in northern Norway. The collective dose for the area affected was calculated as 1100 manSv, corresponding to a collective risk of 54 cases of additional cancer.

7.3. Nuclear-powered vessels

The reactors of nuclear-powered vessels located around the Kola Peninsula represent the greatest density of nuclear reactors in the world. Several types of release have been registered from these vessels, particularly from those operating at sea. However, releases have also occurred at bases on shore, for example in Andreyeva Bay and Gremikha Bay. Limited effort has been made regarding impact assessments for accident scenarios related to operating vessels, decommissioned vessels, or vessel components after dismantling, owing to the traditional secrecy surrounding these vessels, their reactors, and the composition of their fuel. There is a need to standardize the existing studies comparing Russian and western efforts and to complete the assessments. Nevertheless, some significant contributions have already been made, such as the IAEA assessment of the risks from the dumped reactors close to Novaya Zemlya (IAEA, 1998a). Other more recent efforts include the pilot study by the NATO Committee on the Challenges of the Modern Society concerning an environmental risk assessment for decommissioned Russian nuclear submarines still containing fuel, and an evaluation of the potential impact of large releases from the *Kursk* at the time of sinking and during subsequent recovery operations (Baklanov *et al.*, 2003).

The operation, maintenance, decommissioning, and dismantling of a nuclear vessel fleet is a complex process involving a large number of smaller operations. The activities include: different modes of operation (training, patrolling, tracking, etc.); assignments in port, changes

of crew; docking for maintenance and repair; refuelling and defuelling; storage onboard of fuelled reactors; on- and off-loading of fresh and spent fuel from vessels and transport ships; mode of fuel transport; and storage of damaged reactors/damaged fuel.

To date, some of these operations have been covered by risk assessments. Hopefully, the most serious scenarios involving potential releases have been covered; however, as this work and international efforts to assist Russia in these tasks are reaching new levels of advancement and maturity, new facts and scenarios are being identified. The most recent and relevant efforts toward comprehensive impact assessments are presented in the rest of this section. These have been subdivided into vessels in operation (Sections 7.3.1. and 7.3.2.), decommissioned vessels still containing spent fuel on board (Section 7.3.3.), and accident scenarios involving spent fuel and radioactive waste after dismantling of the vessel (Section 7.3.4.). The focus is on the presence of spent fuel because 90 to 99% of the radioactivity resides within the fuel. However, the reactor compartments, and the solid and liquid high-level, medium-level, and low-level radioactive waste also constitute formidable problems, mainly in the remediation of the bases and sites. The latter requires further evaluation as remedial work involving international participation is to begin shortly.

7.3.1. Military vessels

There are around 33 operative nuclear submarines within the Russian North Fleet. According to Ølgaard (2001), these comprise 12 ballistic missile submarines (Typhoon and Delta Classes), 4 cruise missile submarines (Oscar Class), 12 attack submarines (Akula, Sierra, Yankee, and Victor Classes), 1 cruiser (Kirov Class), and 4 other submarines (Yankee, Uniform, and X-ray Classes). These regularly patrol the nearby oceans as part of their contribution to the Russian defense force. During service, four Russian nuclear submarines have sunk, 36 accidents have occurred, and there have been 378 associated fatalities (Ølgaard, 2001).

The first AMAP assessment made reference to design and beyond-design accident scenarios prepared in relation to Russian nuclear-powered submarine refuelling. No new assessments of this type were available for the present assessment and so that in AMAP (1998) remains the most appropriate. A submarine incident in a ship repair yard in Chazhma Bay on the Russian Pacific coast on 10 August 1985 (Sarkisov, 1999; Sivintsev *et al.*, 1994) involved inadvertent criticality in a reactor core. This can be used to illustrate the potential circumstances and the nature, scale, and consequences of such accidents. The accident claimed ten lives and gave rise to 39 cases of acute radiation effects. Subsequent on-site observations and radioecological investigations showed that the accident did not have a measurable radiological impact on Vladivostok or the nearby Shkotovo-22 village. Residual long-lived radioactive contamination in the Chazhma Bay region is localized and does not give rise to serious radioecological concern.

Risk estimates of criticality events during refuelling have been performed by NATO (NATO, 1998). The probability of a severe accident in the Russian navy is estimated to be 2×10^{-3} per refuelling.

7.3.1.1. *Kursk*

The latest accident involving a Russian submarine was that of the *Kursk* in August 2000. The sinking, and subsequent recovery operation, raised considerable concern about possible consequences. The accident represented a significant challenge for the nuclear emergency preparedness organization; from the day of the accident until the larger part of the submarine was brought into dock at Roslyakovo in October 2001.

Owing to considerable concern in Norway, the Norwegian Radiation Protection Authority undertook an environmental risk assessment for four scenarios; combining two inventory calculations and two release scenarios. The *Kursk* inventory calculations were based on information for the Russian cargo ship *Sevmorput* with some adjustment of the technical input data. The hypothetical release rate for radionuclides depends strongly on release conditions. These range from instantaneous release owing to the explosion of torpedoes or cruise missiles within the submarine, to the slow long-term corrosion of fuel material. The latter may occur when seawater has penetrated the fuel cladding. If the cladding is zirconium, penetration may take several hundred years. However, if conditions for galvanic corrosion are present, the cladding could be fully corroded in less than a year.

Two radionuclide release scenarios were considered.

1. An abnormal event one year after the accident, i.e., during the salvage operation, in which 100% of the inventory in both reactors is released instantaneously.

2. The assumption that all barriers, for all practical purposes, have been removed after 100 years, and that 100% of the inventory of both reactors is then released.

Two versions of operational history, resulting in burn-ups of 12 000 (Version 1) and 24 000 (Version 2) MW-days respectively, were considered for each scenario. Both versions were based on the submarine being operational for an average of 50 days per year for each year since commissioning at the end of 1994. Version 2 includes extensive operation of the reactors for electrical power in port, as has been reported to occur by several sources in recent years. An estimated release of 100% of the inventory, a very pessimistic approach, was chosen to demonstrate the consequences of a simple scenario, even if not realistic, to the public concerned. There is a lack of comprehensive environmental assessments of accidents involving submarines in operation and the associated release mechanisms and source terms. Earlier studies concentrated on releases from sunken submarines to the marine environment (Eriksen, 1990; IAEA, 1997) or releases from decommissioned non-defuelled submarines (NATO, 1998) to sea and air. A consideration of submarines in operation, such as the *Kursk*, might indicate more severe consequences owing to the greater amount of short-lived radionuclides present.

Estimates of the radiological consequences for the marine environment of potential radionuclide releases from the *Kursk* were performed for Scenarios 1 and 2, using a box model to estimate radionuclide transport over large distances (>1000 km) and long time-scales

(up to centuries or millennia). The model included terms that describe the dispersion of radionuclides into the marine environment over time (Iosjpe and Strand, 1999; Iosjpe *et al.*, 1997, 2002).

Transport, transfer to fish, and collective doses to humans were modelled for a range of radionuclides present in the reactors. However, most attention was focused on ^{137}Cs because this has a relatively long physical half-life (30 years), readily dissolves in water, and accumulates in edible parts of fish and shellfish. For ^{137}Cs dispersion in oceanic surface water for the worst case potential accidental release, with immediate release of spent fuel and high burn-up (Scenario 1, Version 2), the model predicted that 0.5 years after a hypothetical accidental release of 100% of the inventory, the average activity concentration in Barents Sea water would be 160 to 210 Bq/m^3 in the vicinity of the submarine. Activity concentrations would decrease rapidly and after ten years the average incremental water activity concentration in the Barents Sea was estimated at 0.1 to 2.8 Bq/m^3.

For ^{137}Cs activity concentrations in fish from the Barents Sea region (also for Scenario 1, Version 2) the calculations indicate that during the first few years of potential dispersion, the activity concentrations would vary widely depending on the habitat of the fish. During the early stages of dispersion, the Barents Sea would contain areas with relatively high levels of contamination and areas that were completely unaffected. The calculated transfer to fish is subject to large uncertainties and other transfer pathways, such as particle ingestion, were not considered. The maximum ^{137}Cs activity concentration in fish was calculated as between 0 and 100 Bq/kg during the first year after a hypothetical leak from the *Kursk*. By comparison, the intervention level for ^{137}Cs in basic foodstuffs, as recommended by the EC and adopted by several countries, including Norway, is 600 Bq/kg.

For Scenario 1, the collective dose to man is dominated by the contribution from ^{137}Cs. Calculations show that a collective dose of 61 manSv would be attributable to the intake of ^{137}Cs from the Barents Sea alone, while the total collective dose from all radionuclides from the whole marine area would be 97 manSv. For the latter, contributions from ^{137}Cs and ^{239}Pu correspond to collective doses of 69 and 5.5 manSv, respectively. For comparison, collective doses from other radionuclides for Scenario 1 are estimated at 6.5, 4.3, 2.2, 0.37, and 0.27 manSv for ^{90}Sr, ^{134}Cs, ^{241}Am, ^{147}Pm, and ^{106}Ru, respectively. For Scenario 2, Version1, the total collective dose was estimated at 8.4 manSv. Furthermore, approximately 80% of the collective dose from the Barents Sea was attributable to ^{137}Cs exposure. There is no significant contribution from ^{239}Pu to the collective dose for Scenario 1. For Scenario 2, however, the contribution of ^{239}Pu is comparable to that of ^{137}Cs. This mainly results from the comparatively short radioactive half-life for ^{137}Cs of 30 years.

No indications of leakage from the *Kursk* submarine were observed during the expeditions to the site in August and October 2000. Elevated levels of radioactivity were not observed in any dose-rate readings or in any environmental samples from close to or inside the submarine, even after the submarine was taken ashore in Roslyakovo.

7.3.1.2. *Komsomolets*

The *Komsomolets* submarine sank in 1989 in the Norwegian Sea, south of Bear Island (Bjørnøya). The radioactive inventory at the time of the accident is estimated at 2.8×10^{15} Bq of ^{90}Sr and 3.1×10^{15} Bq of ^{137}Cs in the reactor, and 1.6×10^{13} Bq of plutonium in the warheads. Minor releases of radioactivity from the reactor compartment have been detected but large-scale releases are thought to be unlikely as the containment barriers will prevent corrosion of reactor fuel for at least a thousand years.

7.3.1.3. **Other nuclear submarines**

There are 70 decommissioned submarines moored around the Kola Peninsula at the bases from which they operated, some close to international borders. Fifty-two are waiting to be defuelled and are in various states of repair. Some have damaged cores due to accidents, which has prevented the removal of the fuel from the reactor compartment. Decommissioning submarines with damaged cores is a major problem requiring large investment and often significant radiation risk to workers.

In 1993, the International Arctic Seas Assessment Project (IAEA, 1997) began a study of the radiological and environmental hazard posed by the reactor compartments dumped in the Barents and Kara Seas in the 1960s and 1970s. Six were dumped with spent nuclear fuel onboard (two being complete submarines) and ten were dumped without fuel. An environmental survey of the disposal sites found limited evidence of contamination that could be attributed to the reactor compartments (Strand *et al.,* 1997). Transport and dispersion models using isotope release rates indicated that the maximum annual dose would be received by local populations, although this was <0.1 μSv/yr. However, military personnel that patrol Novaya Zemlya were projected to receive a potential annual dose of up to 700 μSv (comparable to natural background doses). The global collective dose over the next 1000 years from ^{14}C in the inventory was estimated at 8 manSv.

NATO has considered accident analysis in some detail (NATO, 1998). For an environmental release to occur, an event with sufficient energy to dislodge the radioactive material from its normal location and a failure of the containment boundary are required. Fuel within the reactor compartment is the most probable area for such an event to take place due to existing defects, mechanical damage, or overheating. Events can be internal or external (specific to the mooring location) and the main concern is core overheating or a LOCA. This is used as the reference event and indicates the upper limit for consequences arising from other events. The activity release from a core containing spent fuel is estimated at 100 TBq of ^{90}Sr and 600 TBq of ^{137}Cs immediately after the event. The NATO report does not attempt to estimate the quantity of radioactivity that could be released for each internal event analyzed.

The number of potentially hazardous radionuclides likely to be dispersed following a criticality accident on a decommissioned but non-defuelled submarine is relatively small, taking into account core activity, the release fraction, and exposure pathways for radiological effects.

Actinides and fission products provide the greatest potential hazard. Short-lived radionuclides may dominate immediately after an accident and their presence is important in the vicinity of the accident site, but they do not cause extensive spatial contamination. The majority of the dose from an atmospheric release is contributed by ^{137}Cs, ^{134}Cs, and ^{90}Sr, with source terms estimated at 350, 35, and 70 TBq, respectively.

A decommissioned, non-defuelled moored submarine can sink and release radioactivity to the sea as a result of lapsed maintenance. The consequences are not necessarily severe since reactors and submarines are designed to withstand considerable pressure. There may be some activity release from corrosion of the outer surfaces of the nuclear reactor. However, if the reactor compartment was breached, as for example in the event of a collision, corrosion of the fuel could occur rapidly and release fission products to the sea. Estimates based on models using data from Ara Bay, near Murmansk, suggest that the release in the year of the accident would be 1.6 PBq, with actinides providing <1 TBq of the release and fission products dominating. Over time, the predominant isotopes would change owing to differential decay and mobilization.

It is also possible for an undamaged submarine to sink and such a scenario was examined for Ara Bay. Sinking in such shallow water (a few tens of meters) is unlikely to damage a submarine's primary systems. Releases of the four major activation isotopes, ^{60}Co, ^{59}Ni, ^{63}Ni, and ^{14}C, were estimated at around 300 MBq one year after sinking, decreasing to 180 MBq after 20 years (IAEA, 1997). Except for ^{14}C, these isotopes would adsorb onto coastal sediments and ultimately settle to the sea floor or remain on the hull. Models indicate little activity in the waters of Ara Bay and even less 1.5 km from the release site.

Such studies indicate that recovery of sunken submarines or reactor compartments is not too difficult if the reactor is undamaged but that the effects of a criticality accident are difficult to predict, making the consequences difficult to estimate. Nevertheless, the risks of radionuclide release to the Arctic are considered to be negligible.

7.3.2. Civilian icebreakers

The Murmansk Shipping Company operates the Russian icebreaker fleet. According to Ølgaard (2001) the fleet currently comprises six operational icebreakers (*Arktika*, *Rossiya*, *Sovetskiy Soyuz*, *Yamal*, *Taymyr*, and *Vaigach*), and one icebreaking container ship (*Sevmorput*). These are stationed at the Atomflot Repair Technical Plant near Murmansk. Two icebreakers have been decommissioned and defuelled (*Lenin* and *Sibir*). A new icebreaker, *50 let Pobyedy* (50 Years of Victory), is currently under construction at the Baltiysky shipyard in St. Petersburg.

7.3.3. Decommissioned, currently-fuelled submarines

The decommissioning of Russian nuclear submarines in the Arctic has caused considerable concern since the end of the Cold War. In 1992 and under the auspices of the NATO Committee on the Challenges of the Modern Society, Norway initiated a study on cross-border defense-related environmental problems. At an early stage, the working group decided to focus on decommissioned but still fuelled submarines. Operational submarines and nuclear weapons were beyond the scope of the working group. A direct comparison of the risks was not undertaken.

Three scenarios were used to examine releases to the sea: sinking of an undamaged submarine; sinking of a damaged submarine; and a criticality accident followed by sinking. Using the release rate model established during the International Arctic Seas Assessment Project, the dose rate to an individual on a small craft in the harbor of Ara Bay, chosen as the location of the sunken submarine, was 100 μSv/hr. At 2 km north of the site, average dose rates from the water surface to personnel in a small craft decreased to about 10 μSv/hr. At the mouth of Ara Bay, the level decreased to 1 μSv/hr. This work did not include uptake by edible fish species; however, Klopkhin *et al.* (1997), based on model considerations of a radionuclide plume in water, suggest that fish swimming in the plume do not accumulate enough activity to justify restricting their consumption.

Of the many scenarios discussed, only criticality accidents, LOCAs, and hull damage due to sinking or ship collision were considered potential causes of cross-border contamination. Weather conditions during an incident may lead to contamination of foreign territory. For example, using Ara Bay as the accident venue a Gaussian puff model was used to calculate the dispersion of radioactivity for stable weather conditions with winds toward Kirkenes and the county of Finnmark in Norway. Kirkenes is an urban environment, whereas Finnmark represents a critical group with a high consumption of locally produced foodstuffs. For dry deposition only, the ^{137}Cs deposition at Kirkenes was about 10 kBq/m^2 and for ^{134}Cs and ^{90}Sr was typically a factor of ten lower. The maximum annual effective dose for adult members of the public for the two cross-border receptor areas assuming a 'worst-case' scenario (NATO, 1998) is shown in Table 7·4. With dry conditions during the passage of the radioactive cloud, the average individual effective radiation dose received in the first year is <1 mSv. Rainfall during cloud passage may lead to enhanced deposition of radioactivity, which would cause significantly higher long-term radiation doses.

7.3.4. Storage of spent fuel

Russian marine reactors and spent fuel are of international concern. One hundred and eighty-eight nuclear submarines have been decommissioned in Russia. Of these, 48 have been dismantled, 28 are being dismantled, and 112 are waiting for dismantling to start. Most still contain loaded reactors. While the focus on military nuclear issues and spent fuel began around 1990, the infamous service ship *Lepse*, containing more than 600 spent fuel assemblies, is still harbored near Murmansk. According to the CEG (2003), fuel arising from 130 submarine nuclear reactor cores is currently being stored in northwestern Russia, while fuel from an additional 20 cores is located in far eastern Russia. An average reactor core has approximately 455 fuel assemblies. In Decem-

Table 7·4. Maximum annual effective dose estimates for adult members of the public for two cross-border receptor areas assuming a 'plausible worst-case' accident scenario (NATO, 1998). 'Short-term' refers to the first 24 hr of the event (cloud passage) and 'long-term' to the first year excluding the first 24 hr. 'Wet' refers to the assumption of moderate rainfall during passage of the radioactive cloud.

	Kirkenes (urban)		Finnmark (rural)	
	dry	wet	dry	wet
Ground deposition of ^{137}Cs, kBq/m^2	10*	250*	1	25
Integrated air concentration of ^{137}Cs, MBq s/m^3	10	10	1	1
Effective dose from exposure pathway, mSv				
Short-term**				
Inhalation***	0.19	0.19	0.02	0.02
Cloud-shine	–	–	–	–
Ground-shine	–	0.02	–	–
Short-term subtotal	0.19	0.21	0.02	0.02
Long-term				
Ground-shine****	0.08	1.9	0.02	0.5
Ingestion***	0.03*	0.9*	0.19	4.5
Long-term subtotal	0.11	2.8	0.21	5.0
First-year				
Total annual dose	0.30	3.0	0.23	5.0

* average contamination of the wider surroundings of Kirkenes is set equal to 30% of the Kirkenes value;
** no protection assumed in the early phase of the incident;
*** effective dose commitment;
**** corrected for runoff (urban environment only) and shielding (rural area lower than urban environment).

ber 2000, there were reports of large quantities of fuel ready to be taken ashore (Moltz, pers. comm., 2000).

The scenarios for an accident or inappropriate use of a Russian marine reactor or its fuel are numerous, as evidenced by various incidents throughout the 1990s. For example, the sinking of *Komsomolets* and the *Kursk*, several thefts of fresh fuel from bases in northwest Russia, and an attempt to blow up the *Vepr*, an Akula Class submarine, by a distressed Russian sailor after a serious hostage situation at the Gadzhiyevo Naval Base on 11 September 1998. The scenarios include: releases to air, sea and/or the terrestrial environment; sabotage and other radiological incidents initiated deliberately; and thefts or other illegal, organized acquisitions of radiological or fissile material by terrorists.

Earlier impact assessments concentrated on releases from sunken submarines to the marine environment (Eriksen, 1990; IAEA, 1997) or releases from decommissioned, non-defuelled submarines to sea and air (NATO 1998). There is need for additional understanding of criticality issues related to remediation and clean-up activities; damaged cores; and the types of spent fuel configurations currently stored at naval bases such as those at Andrejeva Bay and Gremikha Bay.

7.4. International transport of spent nuclear fuel from commercial use

Between 1992 and 1999 there were six shipments of plutonium and vitrified high level radioactive waste from France to Japan and one shipment of mixed oxide reactor fuel from the United Kingdom to Japan. Such shipments, if carried out in a manner consistent with international guidance and existing IAEA Conventions paying specific attention to the prevention of criticality accidents, pose only minor risks to human health. The risk of accidents for such transport has been reviewed extensively over recent years in a comprehensive cooperation between the IMO and IAEA (IAEA, 2001). The doses to a maximally exposed individual that might be caused by the loss of a flask at sea were estimated to range from 5×10^{-12} Sv/yr for the loss of a vitrified high level waste flask to the deep ocean, to 2×10^{-6} Sv/yr for the loss of a high burn-up irradiated fuel flask to shallow coastal waters.

It is difficult to predict the long-term trend in such traffic. However, if mixed oxide fuel is increasingly used as a means of safeguarding surplus weapons-grade plutonium and if investment in nuclear power generation increases as a means of reducing dependence on fossil fuels and emissions of carbon dioxide to the atmosphere, the quantities and frequency of such shipments may increase substantially. A seminar on the transport of spent nuclear fuel in Norwegian coastal areas convened for Norwegian senior officials in March 2002 concluded that, even if the calculated risk is low, there is a need for consideration of possible release scenarios and for detailed impact assessments. The possible transfer of spent nuclear fuel through Arctic areas has caused controversy, for example in Norway, and will continue to do so if such concerns are not addressed properly.

In the case of transport of spent fuel within, for example, Russian territory, there are potential problems associated with Russian transport ships not adhering strictly to international transport regulations. Any foreign assistance, such as the provision of Norwegian transport ships for assisting Russian authorities in the dismantling of nuclear submarines, would probably demand and ensure adherence to international regulations and standards (IAEA, 2001).

7.5. Reprocessing and production plants

7.5.1. Mayak

Operations at the Mayak PA installation have resulted in serious nuclear environmental contamination. Two accidents have resulted in severe contamination outside the Mayak site boundary. In 1957, an explosion in a

high level waste storage tank caused severe ^{90}Sr contamination of a 1000 km^2 area within the Chelyabinsk, Sverdlovsk and Tyumen regions. This is referred as the 'Kyshtym accident'. In 1967, wind dispersal of contaminated sediment from the dried-out bed of Lake Karachay (a storage reservoir for liquid radioactive waste) resulted in ^{137}Cs deposition over 1800 km^2 surrounding the site. Between 1949 and 1956, authorized discharges of intermediate-level radioactive waste directly into the Techa River resulted in severe contamination downstream from the release point. Although operational procedures have been revised extensively since the late 1950s, as has also been the case at other nuclear installations, the possibility of accidents remains. The human population in the vicinity of Mayak is at most risk from an accident and has, together with the environment, suffered the adverse effects of previous accidents. However, since the Mayak installation is sited at the head of the Techa River, which is a tributary of the major Ob River, there is also the possibility of long-range transport of radionuclides to Arctic areas. AMAP has therefore recommended studies on the transport of radionuclides from land-based sources through river catchments (AMAP, 1998). The possible consequences of far-field transport of radionuclides released as a result of various hypothetical accidents at the Mayak installation have been assessed by the Joint Norwegian–Russian Expert Group on Radioactive Contamination (JNREG, 2003). The study focused on six accident scenarios.

1. *An explosion in a storage tank for high level waste*. This is a modern analogue of the Kyshtym accident. It results in radioactive contamination of the environment and subsequent washout of radionuclides into the river system.
2. *A tornado in the Lake Karachay area*. A tornado passing over Lake Karachay lifts and disperses contaminated water and sediment over the surrounding area in a similar manner to the events of 1967.
3. *Inflow of water from Reservoir 11 to the Techa River due to:*
   a. *a dam break*, which brings dissolved and particulate radionuclides as well as washout from the Techa riverbed and floodplain into the river system;
   b. *a controlled release* that results in a discharge of dissolved radionuclides from Reservoir 11 into the Techa River.
4. *Release of radionuclides from the Asanov Swamp*. This was heavily contaminated by early operational discharges of radionuclides into the Techa River, due to flooding.
5. *An accident at the reprocessing plant*. This is comparable to scenario 1, although on a smaller scale and with other radionuclides being involved.
6. *Groundwater contamination from Lake Karachay* reaches the river system.

The accidents vary in size, impact, and duration. Some allow time for the introduction of measures to reduce their severity; others represent serious, acute accidents (e.g., a dam failure) that allow little possibility of mitigation. All incidents have the potential to release radionuclides that could result in impacts on biota and humans in the surrounding area, both in the near and far field.

Because of the concern regarding long-range river transport, a major focus has been to model the transport of radionuclides through the Techa-Iset-Tobol-Irtysh-Ob River system to Ob Bay and the Kara Sea. In some cases,

Table 7·5. Consequences for Arctic areas of six hypothetical contamination scenarios at Mayak PA (JNREG, 2003).

	Total inventory	Release to environment	Discharge to the Techa River	Collective dose, manSv	Maximum dose per person, mSv/yr*
Current runoff		1.2 TBq/yr ^{90}Sr	0.6 TBq ^{90}Sr for 50 yr	0.01	0.009
Scenario 1. Waste tank explosion	370 PBq – single tank; 20000 PBq – total	15.2 PBq ^{90}Sr + ^{90}Y 20.4 PBq ^{137}Cs	180 TBq ^{90}Sr 59 TBq ^{137}Cs	0.39	1.9
Scenario 2. Tornado	4400 PBq	4.4 PBq ^{90}Sr + ^{137}Cs	5 TBq ^{90}Sr, 0.5 TBq ^{137}Cs	0.005	0.006
Scenario 3a. Dam burst	In water: 650 TBq	300 TBq ^{90}Sr 3.7 TBq ^{137}Cs	300 TBq ^{90}Sr 3.7 TBq ^{137}Cs	1.0	4.8
	In sediment: 1500 TBq	205 TBq ^{90}Sr 150 TBq ^{137}Cs	205 TBq ^{90}Sr 150 TBq ^{137}Cs		
Scenario 3b. Controlled release	650 TBq ^{90}Sr + ^{137}Cs	13 TBq ^{90}Sr 0.16 TBq ^{137}Cs	13 TBq ^{90}Sr 0.16 TBq ^{137}Cs	0.009	0.05
Scenario 4. Asanov Swamp	19-22 TBq ^{90}Sr 170-190 TBq ^{137}Cs	3.2 TBq ^{90}Sr	3.2 TBq ^{90}Sr	0.002	0.01
Scenario 5. Plant accident			1.1 TBq**	0.0007	0.004
Scenario 6. Groundwater contamination	4400 PBq	22 TBq/yr ^{90}Sr	0.6 TBq ^{90}Sr for 50 yr	0.00008	0.00007

*for a diet containing 28 kg fish per year; **estimate for all radionuclides.

other transport processes (e.g., atmospheric transport of radioactive debris) have also been considered. The models, developed by scientists at Mayak PA and SPA Typhoon, include radionuclide transport in river systems, and tornado, flood, and groundwater contamination. Where possible, the models are based on existing scientific knowledge concerning the transport and behavior of radionuclides in the area surrounding Mayak and on the outcome of previous accidents, both at Mayak and other installations. Information on the physico-chemical forms of radionuclides and the influence of speciation on transport processes and mechanisms was also included. Finally, major uncertainties, variability, and model sensitivity has been assessed.

The models required particular variables (release inventory, radionuclide composition, meteorological conditions, etc.) for each scenario. Each scenario will vary according to the course of events, particularly concerning the quantity of radionuclides released, which could be more or less than the hypothetical estimate. Therefore, the estimates derived using the models have large uncertainties. Worst-case conditions were generally considered for each scenario.

The outcome of the modelling exercise is compared with current run-off in Table 7·5. For each scenario, the table presents estimates for the total radionuclide inventory and estimates of radioactive releases to the environment outside the Mayak PA area. Radioactive discharges to the upper Techa-Iset-Tobol-Irtysh-Ob River system (mainly into the Techa River) can be much lower than those to the environment, if for example, contamination is due to washout from the water catchment area. Doses were estimated for Ob Bay and the Kara Sea for a 50-year period after the hypothetical accidents.

Scenarios 1 and 3a result in a very high radioactive discharge to the Techa River. The other scenarios result in much lower radioactive contamination of the Techa River.

The models indicate that ^{90}Sr transport through the river system will lead to a significant increase in contamination in the lower reaches of the Ob River compared to current levels. For example, the additional ^{90}Sr activity concentration for the first year after the dam break is estimated to be five times higher than background. Contamination of Ob Bay and the Kara Sea by other radionuclides is much lower. The longer-lived radionuclides released, ^{137}Cs and Pu, are less mobile in river systems than ^{90}Sr. For all six scenarios, the estimated activity concentration is much lower than the norms regulated by modern radiation safety standards. Overall, it was concluded that the potential doses to Arctic biota and human populations from hypothetical accidents at the Mayak PA installation are very low. However, for the local population, the consequences may be severe.

7.5.2. **Sellafield**

The U.K. Health and Safety Executive have produced safety assessment principles for nuclear plants (HSE, 1992) that address safety issues, including accident scenarios for the Sellafield site. There are a number of principles to ensure that safety is maintained throughout operations and in the event of design or beyond-design based accidents. During operation there are a number of design features that can mitigate an accident before it reaches a critical state. In the event of a design-based accident, the safety assessment principles state that 'there is no release of radioactivity except in the most severe of cases, and even then, no person will receive an effective dose of 100 mSv or more'.

The storage of Highly Active Liquor (HAL) is an important source of concern for severe accidents at the Sellafield site. HAL is a waste product from the reprocessing of irradiated nuclear fuel and is currently stored on-site in water-cooled storage tanks. It is converted into solid form via the process of vitrification (incorporation into borosilicate glass) at a rate limited by the capacity of the vitrification plant. The Nuclear Installations Inspectorate has instructed British Nuclear Fuels (BNFL) to vitrificate HAL from a current volume of about 1300 m^3 (1999) to a buffer volume of 200 m^3 (to feed the vitrification process) by 2015 in response to the potential hazard associated with these wastes (HSE, 2000). The main part of the activity in a typical HAL tank is due to ^{137}Cs and ^{90}Sr. There are in total 21 tanks containing about 7×10^{18} Bq of ^{137}Cs and 4.8×10^{18} Bq of ^{90}Sr (Turvey and Hone, 2000). Vitrified wastes are also stored on-site at Sellafield and are generally thought to be safer than HAL because the fission products are immobilized in a solid matrix and cooled by the circulation of air. They are thus not dependent on an active cooling system.

A major BNFL safety case for HAL stores was completed in 1994. This was followed by a Nuclear Installations Inspectorate assessment (HSE, 2000). The assessment concluded that the BNFL approach to accident analysis was incomplete and not best practice. In 1999, BNFL completed the Continued Operation Safety Report (COSR), which is the latest safety analysis associated with the HAL stores. Although this report is not publicly available, the Radiation Protection Institute of Ireland (RPII) was given access to the BNFL safety documentation and has published an evaluation report of the COSR (Turvey and Hone, 2000). The objectives of the RPII examination of the safety material were to determine whether the COSR includes all significant hazards; to evaluate the conclusions of the COSR on the probability of occurrence of a number of accident scenarios; to determine whether confidence can be placed in the database used in the COSR and to assess the significance of any shortcomings; and to assess the need for further improvements in safety. Turvey and Hone (2000) conclude that the risks of a severe accident associated with the HAL stores are low but identify some areas where the risks could be reduced further. The report also states that the risk of damage from a severe earthquake has not been fully analyzed. According to Turvey and Hone (2000), all other major accident scenarios appear to have been considered in the COSR. Despite the probability of an accident involving a significant release of radioactivity being considered low, Turvey and Hone (2000) identify certain safety weaknesses, e.g., that the water supplies for cooling the tanks are not fully independent of each other, that there is no instrumentation for detecting possible hydrogen build-up in the storage tanks, and that the consequences of very severe accidents have not been adequately assessed.

Low probability but high consequence events appear to pose the greatest environmental risk at Sellafield, and

could even impact upon the Arctic. These include: seismic events; fire or explosion due to hydrogen generation as a result of radiolysis of HAL or red oil reactions (hot organic liquid and aqueous nitrate solution); extreme weather conditions; aircraft crashes; other man-made hazards (toxic gases); criticality; beyond-design basis accidents; and accidents as a result of human factors.

These issues have been considered by BNFL and the U.K. Health and Safety Executive (HSE, 1992, 2000) but estimates of the radiological consequences (i.e., radiation doses) of each accident scenario have not been assessed.

After 11 September 2001, the possibility of a terrorist attack on, or an airplane crash into, nuclear plants has received much attention. A report prepared for the European Parliament by an external contractor, WISE-Paris (WISE-Paris, 2001a), mentions this briefly and addresses the subject in greater detail in a later report (WISE-Paris, 2001b). The HAL stores are identified as the major risk for radioactivity releases. An assessment is made based on an estimated release of half the total ^{137}Cs content in the HAL tank, and then compared with consequences from the Chernobyl accident. However, these estimations are controversial and have received some criticism.

7.6. Conclusions

Risk assessments are important for establishing priorities. Even though the absolute results from these assessments have large uncertainties, their relative magnitudes may be compared in order to help identify where to focus efforts for risk reduction. The outcome of risk assessments and actual accidents indicate that the consequences of releases to the atmosphere, and subsequent fallout to the terrestrial environment, are greater than for releases to the marine environment.

This assessment has addressed the unintentional potential releases from reprocessing plants in central and southern Russia in detail. The first AMAP assessment concluded that possible consequences of accidents at these plants should be assessed for the Arctic population and environments, owing to the possible transport of radionuclides through the river systems. The present assessment shows that the consequences of such accidents for the Arctic are likely to be much less than previously expected.

That many of the sources to be evaluated in risk assessments are within the military domain, e.g., naval reactors and nuclear weapons, is a problem. Necessary information is often restricted. Openness regarding military sources should be promoted, such that risks to society as a whole can be compared and resources for risk reduction programs used optimally.

The increased awareness of terrorist activities since 2001 has also forced the nuclear industry to reassess the probability and consequences of a terrorist event. Although AMAP does not address security issues, and this matter has therefore not been discussed further, it should be noted that, with negative intentions, the results of an 'accident' could be worse than those estimated in the present scenarios.

Chapter 8

Summary

New information on actual and potential sources of radioactive contamination in the Arctic environment has been provided for this assessment. More detailed knowledge for several sources has enabled new impact assessments; however the major sources of radioactive contamination of the Arctic environment are still fallout from atmospheric nuclear weapons tests conducted between 1945 and 1980, discharges from European spent nuclear fuel reprocessing plants, and fallout from the accident at the Chernobyl nuclear power plant in the Ukraine in 1986. Doses to humans are derived mainly from global fallout and fallout from the Chernobyl accident.

A topic new to this assessment is the loss of the submarine *Kursk* of the Russian Northern Fleet off Murmansk in August 2000 after an explosion on board. The *Kursk* has now been recovered and monitoring shows that the accident did not result in any measurable releases of radionuclides to the Arctic environment.

In general, levels of radionuclides in the Arctic are declining. The exceptions are seawater concentrations of the long-lived water-soluble fission products ^{99}Tc and ^{129}I. This is due to increased releases from nuclear fuel reprocessing in Western Europe and supports the recommendation by AMAP in 2000 that the Arctic Council encourage the United Kingdom to reduce the releases from Sellafield to the marine environment of technetium, by implementing best available technology.

There is evidence that sediments are now a source of Pu and ^{137}Cs to the Arctic. Previous releases, such as those from Sellafield that have deposited in Irish Sea sediments, are being remobilized such that these deposits now act as sources to the Arctic. Thus, even if operational releases from reprocessing plants are reduced, radionuclides remobilized from contaminated sediments in the Irish Sea and the Baltic Sea will continue to be observed in the Arctic. Nevertheless, present doses to Arctic peoples from radionuclides originating from spent nuclear fuel reprocessing plants are small, although the uncertainty surrounding the pathways to and effects of such radionuclides in the Arctic indicates the need for further assessment. Impacts on the Arctic should be considered when evaluating discharge reduction measures, and it is recommended that the Arctic Council support a more detailed study on the remobilization of radionuclides from sediment and its potential long-term effects on the Arctic.

Despite the decline in current levels, there is continuing uncertainty about the amount of radionuclides present at a number of sources and potential sources in the Arctic. Access to information about civilian and military sources continues to be a problem. It is recommended that the Arctic Council promotes more openness for restricted information.

New work has been done on doses to populations in the Faroe Islands, Canada, and northwest Russia. Doses to the non-indigenous populations in Russia and to the Faroe Islands' population were the same as estimated during the first AMAP assessment, whereas the new estimates for doses to the indigenous populations in Canada and northwest Russia were lower and higher, respectively. In both cases, the difference was due to revised estimates of the intake of reindeer/caribou meat.

Previously, the focus of radiation protection has been the protection of human health. A new initiative, highlighted in this assessment, is an attempt to develop a basis for protecting the environment from the effects of radiation. An international consensus has emerged for the rapid development of a system and framework for the protection of the environment. The International Union of Radioecology, with support from AMAP, was one of the first international organizations to promote this. It is recommended that AMAP be asked to take an active part in continued efforts to address environmental protection, taking special responsibility for the Arctic. This should focus on the scientific needs associated with protection of the environment, and the development of associated monitoring strategies and assessment tools.

The major concern regarding potential environmental contamination relates to accidents involving nuclear material, especially accidents at nuclear power plants. Models show that a major accident at the Kola nuclear power plant in Russia resulting in substantial release of radioactive material to the atmosphere would require countermeasures to avoid high radiation doses to the population, which may then need to be applied for several years. Vulnerability, expressed as dose from a given fallout, can vary considerably, even over small areas. Owing to high transfer rates and long ecological half-lives, previous deposition must be quantified when estimating the consequences of potential accidents. It is recommended that AMAP be asked to clarify the vulnerability and impact of radioactivity on the Arctic environment and the consequences for emergency preparedness planning.

Major efforts are underway to reduce radiation risks associated with nuclear reactors and radioactive waste handling. Nevertheless, further improvements are warranted. The main criterion of success for a nuclear safety project is its net contribution to the improvement of nuclear safety. Projects must be undertaken within a context that includes safety assessments and environmental impact assessments that incorporate a variety of risk analyses, to demonstrate compliance with risk objectives relating to environmental and human health protection. Future effort will continue to be concentrated on the areas of greatest risk and the operations and facilities that pose the greatest potential threat. To reduce risk, to mitigate the consequences of possible future accidents, and to optimize the use of resources, work has been undertaken on risk management and risk analysis of nuclear activities and on assessments of the vulnerability of Arctic areas. This provides a basis for improved emergency prevention, preparedness, and response to nuclear incidents, with the optimal use of resources.

It is recommended that risk and impact assessment programs, including uncertainty estimates, be performed before action is taken to reduce risk. Risk and impact assessments, including accident scenarios, should be undertaken for the transport of nuclear waste and fuel within the Arctic and nearby areas, and with regard to storage and reprocessing within the Arctic and nearby areas. Since the first AMAP assessment, nuclear safety programs have been undertaken in Russia at nuclear power plants and other nuclear installations relevant to the Arctic. It is recommended that the Arctic Council continue its cooperation with Russia to improve the safety and safeguarding of nuclear installations and waste sites.

Co-operation is required between relevant authorities on the development of initiatives concerning health and safety, and preparedness. Of particular interest are health and safety risks immediately before, during, and after a risk reducing initiative. Although such cooperation has not been prioritized to date, current cooperation between the authorities responsible for radiation protection, environmental protection, and nuclear safety is working well and contributing to effective international programs. Such efforts are not costly and contribute significantly to the development of large multilateral internationally funded projects. At the same time, they have wider significance – one consequence is that Russian management practices and demands relating to radiation protection are now becoming more transparent and more compatible with international guidelines. A further strengthening of the Russian authorities responsible for nuclear protection would increase their ability to effectively implement these improved management practices.

Personal communications

Eikelmann, I-M., 2002. Norwegian Radiation Protection Authority, Østerås, Norway

Joensen, H.P., 2002. University of The Faroe Islands, Argir, Faroe Islands.

Moltz, C., 2000. Institute of International Studies, Monterey, CA, USA.

Rissanen, K., 2003. Finnish Radiation and Nuclear Safety Authority.

Salbu, B., 2001. Agricultural University of Norway, Ås, Norway.

References

Aaltonen V., J. Paatero, J. Hatakka and Y. Viisanen, 2002a. Airborne Cesium-137 in Northern Finland in the early 1960's based on the measurement of archived air filter samples. NKS-70/April 2002. *In:* E. Ilus (ed.). Proceedings of the eighth Nordic Seminar on Radioecology. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki. pp.149-151.

Aaltonen V., J. Paatero, J. Hatakka and Y. Viisanen, 2002b. Airborne Cesium-137 in Finland after the Chernobyl Accident based on the measurements of archived air filter samples. STUK-A 195/June 2003. *In:* V. Paile (ed.). Radiological Protection in the 2000s – Theory and Practice. Turku, Finland, August 25-29, 2002. Nordic Society for Radiation Protection, Roskilde. pp. 465-468

Aarkrog, A. 1971. Radioecological investigation of plutonium in an arctic marine environment. *Health Physics*, 20:31-47.

Aarkrog, A. 1977. Environmental behaviour of plutonium accidentally released at Thule, Greenland. *Health Physics*, 32:271-284.

Aarkrog, A., H. Dahlgaard and K. Nilsson, 1984. Further studies of plutonium and americium at Thule, Greenland. *Health Physics*, 46:29-44.

Aarkrog, A., S. Boelskifte, H. Dahlgaard, S. Duniec, E. Holm and J.N. Smith, 1987. Studies of transuranics in an arctic marine environment. *Journal of Radioanalytical and Nuclear Chemistry-Articles*, 115:39-50.

Aarkrog, A., E. Buch, Q.J. Chen, G.C. Christensen, H. Dahlgaard, H. Hansen, E. Holm, S.P. Nielsen and M. Strandberg, 1994. *Environmental Radioactivity in the North Atlantic Region including the Faroe Islands and Greenland, 1990 and 1991*. Risø National Laboratory, Roskilde, Denmark, 88p.

Åhman, B., and T. Nylén. 1998. Radiocesium i ren i relation till markbeläggning och halter i växter samt till årstid och diet (Radiocaesium in reindeer in relation to ground deposition, levels in vegetation, season and diet) – Progress report to the Swedish Radiation Protection Institute, Reindeer Husbandry Unit, Swedish University of Agricultural sciences, Uppsala. In Swedish.

Akhunov, V.D., V.N. Kovalenko, A.D. Ruzahkin, S.G. Testov, A.P. Vasiliev, A.V. Grigorev, V.G. Aden, S.U. Bulkin, V.K. Bulygin, V.V. Vazinger, U.A. Ivanov and N.G. Sandler, 2001. Spent nuclear fuel and waste management at Andreeva Bay. Workshop on Russian Federation reports, conditions, priorities and response options. Hosted by the U.S. DoE, U.S. DoD, U.S. DoS. Idaho Falls, October 23-25 2001.

AMAP, 1997. *Arctic Pollution Issues: A State of the Arctic Environment Report*. Arctic Monitoring and Assessment Programme (AMAP), Oslo, 188p.

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

Amiro, B.D., 1997. Radiological dose conversion factors for generic non-human biota used for screening potential ecological impacts. *Journal of Environmental Radioactivity*, 35(1):37-51.

Amundsen, I., B. Lind and A. Nikitin, 2002a. Environmental monitoring of radionuclides before, during and after the raising of the nuclear submarine the Kursk. *In:* Strand P. and T. Jølle (eds.). Proceedings of the fifth International Conference on Radioactive Contamination in the Arctic and Antarctic, St Petersburg, June 16-20, 2002, pp. 31-34.

Amundsen, I., B. Lind, O. Reistad, K. Gussgaard, M. Iosjpe and M. Sickel, 2002b. The accidental sinking of the nuclear submarine, the Kursk; monitoring of radioactivity and the preliminary assessment of the potential impact of radioactive releases. *Marine Pollution*, 44:459-468.

Anon, 2001. Monitoring of radioactive contamination of marine environment when preparing for the Kursk raising, during the raising operation and after it. Report on works carried out in pursuit of the Directive of Roshydromet No. 140-26 of 03.08.2001. On participation of the staff of Roshydromet in the special purpose expedition. SPA Typhoon (Obninsk).

Arnalds, Ó., 1999. Soils and soil erosion in Iceland. *In*: Ármannsson H. (ed.). Proceedings of the fifth International Conference on the Geochemistry of the Earth's Surface, pp. 135-138. Balkema, Rotterdam.

Baklanov, A., A. Mahura, D. Jaffe, L. Thaning, R. Bergman and R. Andres, 2002. Atmospheric transport patterns and possible consequences for the European north after a nuclear accident. *Journal of Environmental Radioactivity*, 60(1-2):23-48.

Baklanov, A., A. Mahura and J.H. Sørensen, 2003. Methodology for prediction and estimation of consequences of possible atmospheric releases of hazardous matter: 'Kursk' submarine study. *Atmospheric Chemistry and Physics Discussions*, 3:1515-1556.

Bartnicki, J. and J. Saltbones, 1997. Analysis of atmospheric transport and deposition of radioactive material released during a potential accident at Kola Nuclear Power Plant. Research Report No. 43. Det Norske Meteorologiske Institutt, Oslo.

Beasley, T.M., L.W. Cooper, J.M. Grebmeier, L.R. Kilius and H.A. Synal, 1997. ^{36}Cl and ^{129}I in the Yenisei, Kolyma, and Mackenzie Rivers. *Environmental Science & Technology*, 31:1834-1836.

Beresford, N.A., S.M. Wright and T. Sazykina (eds.), 2001. *Reference Arctic Organisms. A deliverable report for EPIC*. EC Inco-Copernicus project, ICA2-CT-2000-10032, Centre for Ecology and Hydrology, UK.

Berti, P.R., H.M. Chan, O. Receveur, C.R. MacDonald and H.V. Kuhnlein, 1998. Population exposure to radioactivity from consumption of caribou among Dene/Métis of Denendeh (Western Northwest Territories, Canada). *Journal of Exposure Analysis and Environmental Epidemiology*, 8(2)145-158.

Blaylock, B.G., 1982. Radionuclide data bases available for bioaccumulation factors for freshwater biota. *Nuclear Safety*, 23(4): 427-438.

BNFL, 2000. *Annual Report on Discharges and Monitoring of the Environment in the United Kingdom, 1999*. British Nuclear Fuels, UK.

Borghuis, A.M., A. Liland and P. Strand, (eds.), 2002. Arctic vulnerability to radioactive contamination. EU contract No. ERB IC15-CT98-0201. Final report for the period 1998.09.01-2001.08.31. Norwegian Radiation Protection Authority, Østerås.

Brown, J.E., A.K. Kostad, A.L. Brungot, B. Lind, A.L. Rudjord and P. Strand, 1999. Levels of ^{99}Tc in biota and seawater samples from Norwegian coastal waters and adjacent sea. *Marine Pollution Bulletin*, 38:560-571.

Brown, J., A.K. Kolstad, B. Lind, A.L. Rudjord and P. Strand, 1998. Technetium-99 contamination in the North Sea and in Norwegian coastal areas 1996 and 1997. StrålevernRapport 1998:3, Norwegian Radiation Protection Authority, Østerås.

Brown, J., J.R. Cooper, J.A. Jones, L. Flaws, R. McGeary and J. Spooner, 2000. *Review of Decontamination and Clean-up Techniques for use in the UK following Accidental Releases of Radioactivity to the Environment*. DOE/RAS/96.009.

Busby, R., M. McCartney and P. McDonald, 1997. Technetium-99 concentration factors in Cumbrian seafood. *Radioprotection - Colloques*, 32:311-316.

Cappelen, J. and E.V. Laursen, 1998. *The Climate of the Faroe Islands – with Climatological Standard Normals, 1961-1990*. Technical Report No. 98-14, Danish Meteorological Institute, Copenhagen, 62p.

CEC, 1990. The radiological exposure of the population of the European Community from radioactivity in North European ma-

rine waters. Project 'Marina' CEC Radiation protection – 47 EUR 12483. Commission of the European Communities, 571p.

CEG, 2002. Working material for the 14th meeting of the Contact Expert Group for International Radwaste Projects in the Russian Federation. Volume I, Information on cooperation projects. International Atomic Energy Agency, Vienna.

CEG, 2003. Main problems of spent nuclear fuel and radioactive waste management in the Russian Federation. Contact Expert Group for International Radwaste Projects in the Russian Federation. International Atomic Energy Agency, Vienna.

Claassen, H.C., 1978. Hydrologic processes and radionuclide distribution in a cavity and chimney produced by the Cannikin nuclear explosion, Amchitka Island, Alaska. Geological Survey Professional Paper 712-D, U.S. Government Print Office, Washington DC.

Clarke, R., 1999. Control of low-level radiation exposure: time for a change? *Journal of Radiological Protection*, 19:107-115.

Cochran, J.K., S.B. Moran, N.S. Fisher, T.M. Beasley and J.M. Kelley, 2000. Sources and transport of anthropogenic radionuclides in the Ob River system, Siberia. *Earth and Planetary Science Letters*, 179:125-137.

Cook, G.T., A.B. Mackenzie, P. McDonald and S.R. Jones, 1997. Remobilization of Sellafield-derived radionuclides and transport from the north-east Irish Sea. *Journal of Environmental Radioactivity*, 35:227-241.

Cooper, L.W., T.M. Beasley, X.L. Zhao, C. Soto, K.L. Vinogradova and K.H. Dunton, 1998. Iodine-129 and plutonium isotopes in Arctic kelp as historical indicators of transport of nuclear fuel-reprocessing wastes from mid-to-high latitudes in the Atlantic Ocean. *Marine Biology*, 131:391-399.

Cooper, L.W., T. Beasley, K. Aagaard, J.M. Kelley, I.L. Larsen and J.M. Grebmeier, 1999. Distributions of nuclear fuel-reprocessing tracers in the Arctic Ocean: Indications of Russian river influence. *Journal of Marine Research*, 57:715-738.

Cooper, L.W., J.M. Kelley, L.A. Bond, K.A. Orlandini and J.M. Grebmeier, 2000. Sources of the transuranic elements plutonium and neptunium in Arctic marine sediments. *Marine Chemistry*, 69:253-276.

Crochet, P., 2002. A linear model for mapping precipitation in Iceland. Report No. 02028. Icelandic Meteorological Office, Reykjavik.

Dahlgaard, H., 1994. Sources of Cs-137, Sr-90 and Tc-99 in the East Greenland Current. *Journal of Environmental Radioactivity*, 25:37-55.

Dahlgaard, H., 2002. Baltic ^{137}Cs outflow through the Danish Straits indicates remobilisation. *In*: Børretzen P, T. Jølle and P. Strand (eds.). Proceedings of the International Conference on Radioactivity in the Environment, Monaco, September 1-5, 2002, pp. 516-520.

Dahlgaard, H., Q. Chen, J. Herrmann, H. Nies, R.D. Ibbett and P.J. Kershaw, 1995. On the background level of Tc-99, Sr-90 and Cs-137 in the North Atlantic. *Journal of Marine Systems*, 6:571-578.

Dahlgaard, H., T.D.S. Bergan and G.C. Christensen, 1997. Technetium-99 and caesium-137 time series at the Norwegian coast monitored by the brown alga *Fucus vesiculosus*. *Radioprotection – Colloques*, 32, 353-358.

Dahlgaard, H., M. Eriksson, E. Ilus, T. Ryan, C.A. McMahon and S.P. Nielsen, 2001. Plutonium in the marine environment at Thule, NW-Greenland after a nuclear weapons accident. *In*: Kudo, A. (ed.). *Plutonium in the Environment*, pp. 15-30. Elsevier Science, UK.

Dasher, D., W. Hanson, S. Read, S. Faller, D. Farmer, W. Efurd, J. Kelley and R. Patrick, 2002. An assessment of the reported leakage of anthropogenic radionuclides from the underground nuclear test sites at Amchitka Island, Alaska, USA to the surface environment. *Journal of Environmental Radioactivity*, 60:165-187.

Dudley, W.W. Jr., W.C. Balance and V.M. Glanzman, 1977. Hydrology. *In*: Merritt, M.L. and G.R. Fuller (eds.). *The Environment of Amchitka Island, Alaska*, pp. 35-51. Report TID-26712, U.S. Energy Research and Development Administration.

Eichelberger, J., J. Freymueller, G. Hill and M. Patrick, 2002. Nuclear stewardship: Lessons from a not-so-remote island. *Geotimes*, 47(3):20-23.

Energgoatomizdat, 1998. Norms of radiation safety (NRB76/87). Basic sanitary regulations. OSP-72/87 3rd edition. Energgoatomizdat, Moscow.

Eriksen, V.O., 1990. Sunken nuclear submarines: a threat to the environment. Norwegian University Press, Oslo, 176p.

Eriksson, M., 2002. On weapons plutonium in the Arctic environment (Thule, Greenland). Risø-R-1321, Risø National Laboratory, Roskilde, 150p.

Eriksson, M., H. Dahlgaard, E. Ilus, T. Ryan, Q.J. Chen, E. Holm and S.P. Nielsen, 1999. Plutonium in the marine environment off Thule Air Base, N.W. Greenland. Inventories and distribution in sediments 29 years after the accident. *In*: Strand, P. and T. Jølle (eds.). Proceedings of the fourth International Conference on Environmental Radioactivity in the Arctic, Edinburgh, 20-23 September, 1999, pp. 60-62.

Fadeev, S.L., V.A. Balakin, V.S. Koltunov, N.N. Krasnov, A.A Purstovalov and V.V. Fedorov, 1980. Development and production of radionuclides, radioisotope sources of heat and radioactive sources of ionizing radiation. Isotopes in the USSR. Special issue: Materials of the science and technical conference: 30 years of production and application of isotopes in the USSR, Obninsk, October 1978, No 59. State Committee of the USSR on application of atomic energy (Atomizdat), Moscow. pp. 9-19. In Russian.

Faller, S.H. and D.E. Farmer, 1998. Long Term Hydrological Monitoring Program, Amchitka Island, Alaska 1997. EPA-402-R-98-002. U.S. Environmental Protection Agency. 25p.

Fehn, U. and G. Snyder, 2000. ^{129}I in the Southern Hemisphere: Global redistribution of an anthropogenic isotope. *Nuclear Instruments & Methods in Physics Research*, 172:366-371.

Fehn, U., G.R. Holdren, D. Elmore, T. Brunelle, R. Teng and P.W. Kubik, 1986. Determination of natural and anthropogenic ^{129}I in marine sediments. *Geophysical Research Letters*, 13:137-139.

Fylkesmannen i Finnmark, 2001. Miljøprosjekt 2001, Avfallsbehandling av Radioisotopiske Termoelektriske Generatorer på Kolahalvøya. Miljøvernavdelingen, Fylkesmannen i Finnmark, Kirkenes. In Norwegian and Russian.

Gedeonov, A.D., E.R. Petrov, V.G. Alexeev, I.N. Kuleshova, M.L. Savopulo, I.S. Burtsev, V.Yu. Shkroev and V.I. Arkhipov, 1999. Residual radioactive contamination at the peaceful nuclear explosion sites 'Craton-3' and 'Crystal' in the Republic of Sakha (Yakutia). Strand, P. and T. Jølle (eds.) Proceedings of the fourth International Conference on Radioactive Contamination in the Arctic, Edinburgh, September 20-23, 1999, pp. 43-46.

Goscomsanepidnadzor, 1996. Norms of radiation safety (NRB-96) Hygienic regulations -M. Goscomsanepidnadzor publishers, Russia, 127p.

Government of the Russian Federation, 2000. Resolution No. 149 of 22 February 2000, Moscow, on the federal targeted program 'Nuclear and radiation safety of Russia' for the period 2000-2006.

Gray, J., S.R. Jones and A.D. Smith, 1995. Discharges to the environment from the Sellafield site. *Journal of Radiological Protection*, 15:99-131.

GRNC, 1999. Inventaire des rejets radioactifs des installations nucléaires. *Groupe Radioecologie Nord Cotentin*, 1: juillet. In French.

Grøttheim, S., 2000. Artificial radionuclides in the northern European marine environment in 1995. Distribution of radiocaesium, plutonium and americium in sea water and sediments. StrålevernRapport 2000:1, Norwegian Radiation Protection Authority, Østerås. 102p.

Hameedi, M.J., D.W. Efurd, M.R. Harmon, N.J. Valette-Silver and A. Robertson, 1999. Levels and sources of radionuclides in sediment and biota of the eastern Bering Sea (US Arctic). *In*: Strand, P. and T. Jølle (eds.). Proceedings of the fourth International Conference on Radioactive Contamination in the Arctic, Edinburgh, September 20-23, 1999, pp. 16-19.

Herrmann, J., H. Nies and I. Goroncy, 1998. Plutonium in the deep layers of the Norwegian and Greenland Seas. *Radiation Protection Dosimetry*, 75:237-245.

Hou, X.L., H. Dahlgaard and S.P. Nielsen, 2000a. Iodine-129 time series in Danish, Norwegian and northwest Greenland coasts and the Baltic Sea by seaweed. *Estuarine, Coastal and Shelf Science*, 51:571-578.

Hou, X.L., H. Dahlgaard, S.P. Nielsen and W.J. Ding, 2000b. Io-

dine-129 in human thyroid and seaweed in China. *Science of the Total Environment*, 246:285-291.

Hove, K., H. Lönsjö, I. Andersson, R. Sormunen-Cristian, H. Solheim Hansen, K. Indridason, H.P. Joensen, V. Kossila, A. Liken, S. Magnússon, S.P. Nielsen, A. Paasikallio, S.E. Pálsson, K. Rosén, T. Selnes, P. Strand and T. Vestergaard, 1994. Radiocaesium transfer to grazing sheep in Nordic environments. *In*: H. Dahlgaard (ed.). *Nordic Radioecology, the Transfer of Radionuclides through Nordic Ecosystems to Man*, pp. 211-227. Elsevier Science, Amsterdam.

Howard, B.J., 2000. The concept of radioecological sensitivity. *Radiation Protection Dosimetry*, 92:29-34.

HSE, 1988. *The Tolerability of Risk from Nuclear Power Stations*. U.K. Health and Safety Executive.

HSE, 1992. *Safety Assessment Principles for Nuclear Plants*. U.K. Health and Safety Executive.

HSE, 2000. The storage of liquid high level waste at BNFL, Sellafield - an updated review of safety. U.K. Health and Safety Executive.

Huizenga, D.G., 2001. Letter to Doug Dasher of ADEC Re: Joint US Department of Energy and US Air Force environmental assessment of the disposition of radioisotope thermoelectric generators (DOE/EA-1351). March 28, 2001, 4p.

Hunt, G.J. and P.J. Kershaw, 1990. Remobilisation of artificial radionuclides from the sediment of the Irish Sea. *Journal of Radiological Protection*, 10:147-151.

IAEA, 1976. *Effects of Ionizing Radiation on Aquatic Organisms and Ecosystems*. IAEA Technical Report No. 172. International Atomic Energy Agency, Vienna.

IAEA, 1985. *Sediment Kds and Concentration Factors for Radionuclides in the Marine Environment*. IAEA Technical Report No. 247. International Atomic Energy Agency, Vienna. 73p.

IAEA, 1988. *Assessing the Impact of Deep Sea Disposal of Low Level Radioactive Waste on Living Marine Resources*. IAEA Technical Report No. 288. International Atomic Energy Agency, Vienna.

IAEA, 1992. *Effects of Ionising Radiation on Plants and Animals at Levels implied by Current Radiation Protection Standards*. IAEA Technical Report No. 332. International Atomic Energy Agency, Vienna.

IAEA, 1997. *Radionuclide release from marine reactors dumped in the Kara Sea*. International Arctic Seas Assessment Project. IAEA Technical Report No. 837, International Atomic Energy Agency, Vienna.

IAEA, 1998a. *Radiological conditions of the western Kara Sea*. International Arctic Seas Assessment Project. International Atomic Energy Agency, Vienna.

IAEA, 1998b. *The Radiological Situation at the Atolls of Mururoa and Fangataufa*. International Atomic Energy Agency, Vienna.

IAEA, 2000. Protection of the environment from the effects of ionizing radiation: international perspectives, 29 August – 1 September 2000, Vienna. Report of specialists meeting (reference: 723-J9-SP-1114.2). International Atomic Energy Agency, Vienna.

IAEA, 2001. *Severity, Probability and Risk of Accidents during Maritime Transport of Radioactive Material: Final report of a co-ordinated research project, 1995-1999*. IAEA Technical Report No. 1231. International Atomic Energy Agency, Vienna.

IAEA, 2002. *Ethical Considerations in Protection of the Environment from the Effects of Ionizing Radiation*. IAEA Technical report No. 1270. International Atomic Energy Agency, Vienna.

ICRP, 1977. Recommendations of the International Commission on Radiological Protection. ICRP Publication No. 26. *Annals of the ICRP*, 1(3).

ICRP, 1991. Recommendations of the International Commission on Radiological Protection. ICRP Publication No. 60. *Annals of the ICRP*, 21(1-3).

ICRP, 1992. Principles for intervention for protection of the public in a radiological emergency. International Commission on Radiological Protection, ICRP Publication No. 63. *Annals of the ICRP*, 22(4).

ICRP, 1993. Age-dependent doses to members of the public from intake of radionuclides: Part 2. Ingestion Dose Coefficients. ICRP Publication No. 67. *Annals of the ICRP*, 23(3-4).

Iosjpe, M., 2002. Box modeling approach for evaluation of influence of ice transport of radionuclides for doses to man. *In:* E. Ilus (ed.). Proceedings of the eighth Nordic Seminar on Radioecology. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

Iosjpe, M. and A. Borghuis, 2000. Box model improving. *In*: M. Iosjpe (ed.). *Transport and Fate of Contaminants in the Northern Seas*. Norwegian Radiation Protection Authority, Østerås. pp. 2-31.

Iosjpe, M. and P. Strand, 1999. New approach for box modelling for marine environment. *In*: Strand, P. and T. Jølle (eds.). Proceedings of the fourth International Conference on Environmental Radioactivity in the Arctic. Edinburgh, 20-23 September, 1999, pp. 63-66.

Iosjpe, M. and P. Strand (eds.), 2002. Radiological assessment of consequences from radioactive contamination of Arctic marine areas (ARCTICMAR), Final report. Contract IC 15-CT98-0209. Norwegian Radiation Protection Authority, Østerås.

Iosjpe, M., P. Strand and B. Salbu, 1997. Estimations of significance of some processes for modelling of consequences from releases in the Arctic Ocean. *In*: P. Strand (ed.). Proceedings of the third International Conference on Environmental Radioactivity in the Arctic. Tromsø, June 1997, pp 74-75.

Iosjpe, M., J. Brown and P. Strand, 2002. Modified approach for box modelling of radiological consequences from releases into the marine environment. *Journal of Environmental Radioactivity*, 60(1-2):91-103.

IUR, 2000. Doses and effects in non-human systems. International Union of Radioecology, Report No. 1.

Ivanov, A.V., G.A. Krasilov, V.A. Logachev, A.M. Matushchenko and V.G. Safranov, 1997. The north testing site Novaya Zemlya. Radioecological consequences of nuclear testing. State Institute of Applied Ecology, Moscow. In Russian.

Izrael, Yu.A., E.D. Stukin, S.V. Konstantinov, V.N. Soifer and A.V. Smetankin, 2000. Previous results of the radioecological expedition in the region of loss of the nuclear submarine 'Kursk'. *In*: Proceedings of the International Conference on Radioactivity after Nuclear Explosions and Accidents, pp. 24-25. Moscow, April 24-26, 2000.

JNC, 2000. H12: Project to establish the scientific and technical basis for HLW disposal in Japan. Project overview report. Japan Nuclear Cycle Development Institute (JNC), TN1410 2000-001.

Joensen, H.P., 1999. Long-term variation of radiocaesium in the food chain of lamb in the Faroe Islands. *Journal of Environmental Radioactivity*, 46:345-360.

JNREG, 2001. EIA in Norway and Russia – report from the working group on EIA systems in Norway and Russia. The Joint Norwegian–Russian Commission on Environmental Protection and the Ministry of Environment of Norway.

JNREG, 2002a. Long-term consequences of potential radioactive contamination in the Northern areas: Northern Norway. Joint Norwegian–Russian Expert Group on Radioactive Contamination of the Northern Areas, Norwegian Radiation Protection Authority, Østerås.

JNREG, 2002b. Long-term consequences of potential radioactive contamination in the Northern areas: Murmansk region. Joint Norwegian–Russian Expert Group on Radioactive Contamination of the Northern Areas, Norwegian Radiation Protection Authority, Østerås.

JNREG, 2002c. Long-term consequences of potential radioactive contamination in the Northern areas: Summary report. Joint Norwegian–Russian Expert Group on Radioactive Contamination of the Northern Areas, Norwegian Radiation Protection Authority, Østerås.

JNREG, 2003. Impacts on man and the environment in Ob Bay and the Kara Sea resulting from hypothetical accidents at the 'Mayak' production association, Urals, Russia. Joint Norwegian–Russian Expert Group on Radioactive Contamination of the Northern Areas, Norwegian Radiation Protection Authority, Østerås.

Karcher, M.J., S. Gerland, I.H. Harms, M. Iosjpe, H.E. Heldal and P.J. Kershaw, 2002. The dispersion of technetium-99 in the Nordic Seas and the Arctic Ocean in the 1990s according to model results and observations. *In:* Børretzen, P., T. Jølle and P. Strand (eds.). Proceedings of the International Conference on Radioactivity in the Environment. Monaco, 1-5 September, 2002, pp. 511-515.

Karcher, M.J., R..Gerdes, F. Kauker and C. Koeberle, 2003. Arctic warming – evolution and spreading of the 1990s warm event in the Nordic Seas and the Arctic Ocean. *Journal of Geophysical Research*, 108:C2,10.1029/2001JC001265.

Kershaw, P.J., D.S. Woodhead, M.B. Lovett and K.S. Leonard, 1995. Plutonium from European reprocessing operations – Its behaviour in the marine environment. *Applied Radiation Isotopes*, 46:1121-1134.

Kershaw, P.J., D.C. Denoon and D.S. Woodhead, 1999. Observations on the redistribution of plutonium and americium in the Irish Sea sediments, 1978 to 1996: concentrations and inventories. *Journal of Environmental Radioactivity*, 44:191-221.

Khlopkin, N.S., V.V. Elatomtsev, B.G. Lystsov, B.G. Pologikh, Y.V. Sivintsev, A.S. Timonin and A.P. Zotov, 1997. Nuclear safety assessment of stored afloat non-defuelled decommissioned nuclear submarines. Report No. 31/8010, RRC Kurchatov Institute. Norwegian Defence Research Establishment, Kjeller.

Knudsen, Sk.V., 1940. Ærnæringsekspedition til Færøerne 1936-37 (Nutritional expedition to the Faroe Islands 1936-37). Nyt Nordisk Forlag. In Danish.

Koblinger, L., 2000. Can we put aside the LNT dilemma by the introduction of the controllable dose? *Journal of Radiological Protection*, 20:5-8.

Kodyukov, V.M., N.A. Gorshkov, A.A. Pustovalov, Yu.A. Grebenshchikov, Yu.D. Gubanov, S.P. Lalykin, B.G. Maslennikov, A.D. Dolnik, V.I. Peresypkin and S.M. Mansurov, 1980. Application of radioisotope thermoelectric generators on the base of Sr-90 for power supply of the systems of navigation, self-contained meteorological and magnetic variation stations. Isotopes in the USSR. Special issue: Materials of the science and technical conference: 30 years of production and application of isotopes in the USSR, Obninsk, October 1978, No 59. State Committee of the USSR on application of atomic energy (Atomizdat), Moscow. pp. 80-89.

Kolehmainen, S., E. Häsänen and J.K. Miettinen, 1966. ^{137}Cs levels in fish of different limnological types of lake in Finland during 1963. *Health Physics*, 12:917-922.

Kolehmainen, S., E. Häsänen and J.K. Miettinen, 1967. ^{137}Cs in fish, plankton and plants in Finnish lakes. *In:* Åberg B. and F.P. Hungate (eds.). Proceedings of the International Symposium on Radioecological Concentration Processes, pp. 913-919. Stockholm, April 25-29, 1966.

Kolstad, A. and B. Lind, 2002. Radioactivity in the marine environment 2000 and 2001. Technetium-99 concentrations in Norwegian coastal waters and biota. StrålevernRapport 2002:6. Norwegian Radiation Protection Authority, Østerås. 19p.

Kostiainen, E. and K. Rissanen, 2003. ^{137}Cs and ^{90}Sr in dairy and farm milk in Finnish Lapland 1960-2000. *In:* V. Paile (ed.). Radiological Protection in the 2000s – Theory and Practice. Turku, Finland, August 25-29, 2002. Nordic Society for Radiation Protection, Roskilde. pp. 493-497

Krey, P.W., E.P. Hardy, C. Pachucki, F. Rourke, J. Coluzza and W.K. Benson, 1976. Mass isotopic composition of global fall-out plutonium in soil. *In: Transuranium Nuclides in the Environment*, pp. 671-678. International Atomic Energy Agency, Vienna.

Kryshev, I.I. and E.P. Riazantsev, 2000. Ecological Safety of Nuclear-energy Complex in Russia. IzdAT, Moscow.

Kudelsky, A.V., J.T. Smith, S.V. Ovsiannikova and J. Hilton, 1996. Mobility of Chernobyl-derived ^{137}Cs in a peat bog system within the catchment of the Pripyat River, Belarus. *Science of the Total Environment*, 188(2-3):101-113.

Kuhnlein, H.V., O. Receveur, H.M. Chan and E. Loring, 2000. Assessment of dietary benefit/risk in Inuit communities. Centre for Indigenous Peoples Nutrition and Environment (CINE), McGill University, and Inuit Tapirisat of Canada, Ottawa. 377p + App.

Kuznetsov, Yu.V., 1995. An assessment of the contribution by the Enisey and Ob into total radioactive contamination of the Kara Sea. *In*: Strand P. and A. Cooke (eds.). Proceedings of the second International Conference on Environmental Radioactivity in the Arctic. Oslo, 21-25 August, 1995, pp. 72-74.

Lamp T. 1994. Power system assessment for the Burnt Mountain seismic observatory. Report #W1-TR-94-2026. Aeropropulsion and Power Directorate, Wright Patterson Air Force Base, OH, 81p.

Larsen, E., Holo, E.N., Saltbones, J. and E. Stokke, 1999. Kola konsekvensanalyse, Vurdering av dosemessige konsekvenser av en eventuell ulykke ved kjernekraftverk, StrålevernRapport 1999. No 10. Norwegian Radiation Protection Authority, Østerås. In Norwegian.

Lavkovsky, S.A., 1998. Project 101B-96. Review of the project and its preliminary results. *In*: Afterword to the 'White Book' (Yablokov Commission), pp. 8-18. Proceedings of the International Seminar, January 19-21, 1998. International Science and Technology Center, CDB 'Lazurit' – Nizhni Novgorod.

Leonard, K.S., D. McCubbin, P. Blowers and B.R. Taylor, 1999. Dissolved plutonium and americium in surface waters of the Irish Sea, 1973-1996. *Journal of Environmental Radioactivity*, 44:129-158.

Logachev, V.A. (ed.), 2000. Nuclear tests of the USSR: the Novaya Zemlya testing site. Ensuring general and radiation safety of nuclear tests. Facts, evidence, recollections. Federal Department of Medical and Biological and Extreme Problems at the Ministry of Health of the Russian Federation, Moscow. 487p.

Lubenau, J.O. and D.J. Strom, 2002. Safety and security of radiation sources in the aftermath of 11 September 2001. *Health Physics*, 83(2):155-164.

Lysgaard, L. 1969. Foreløbig oversigt over klimaet på Færøerne. Hovedsagelig baseret på observationer i normalperioden 1931-60 og en del observationer fra et kortere åremål. Meddelelser. Danish Meteorological Institute, Copenhagen. In Danish.

Matishov, G.G., D.G. Matishov, A.E. Namjatov, J.N. Smith, J. Carroll and S. Dahle, 2002. Radioactivity near the sunken submarine 'Kursk' in the southern Barents Sea. *Environmental Science & Technology*, 36:1919-1922.

Mayall, A., 2002. The regulation of technetium-99 discharges at Sellafield. *In:* International Conference on Issues and Trends in Radioactive Waste Management, Vienna. December 9-13 2002. pp. 1-11.

Meili, M., A. Aldahan and G. Possnert, 2002. ^{129}I in Arctic freshwaters: local or global sources? *In:* Strand, P. and T. Jølle (eds.). Proceedings of the fifth International Conference on Radioactive Contamination in the Arctic and Antarctic. St Petersburg, 16-20 June, 2002, pp. 62.

Miller, P. and N. Buske, 1996. Nuclear flashback: the return to Amchitka. Report of a Greenpeace Scientific Expedition to Amchitka Island, Alaska. Greenpeace.

Minzdrav, 1999. Norms of radiation safety (NRB-99). Hygienic regulations. Minzdrav of Russia, 1999, 116 p.

Mitchell, P.I., L.L. Vintro, H. Dahlgaard, C. Gasco and J.A. Sanchezcabeza, 1997. Perturbation in the Pu-240/Pu-239 global fallout ratio in local sediments following the nuclear accidents at Thule (Greenland) and Palomares (Spain). *Science of the Total Environment*, 202:147-153.

Moran, J.E., S. Oktay, P.H. Santschi and D.R. Schink, 1999. Atmospheric dispersal of ^{129}iodine from nuclear fuel reprocessing facilities. *Environmental Science & Technology*, 33:2536-2542.

Mortensen, L.E., 2002. Arctic mountain meteorology at the Sornfelli mountain in year 2000 in the Faroe Islands. *Annales Societatis Scientiarum Færoensis (Fróðskaparrit)*, Vol. 50.

NATO, 1998. Cross-border environmental problems emanating from defense related installations and activities, Phase II: 1995-1998. Final Report. Vol. 4: Environmental risk assessment for two defense-related problems, Report No. 227. North Atlantic Treaty Organization.

NCRP, 1991. Effects of ionising radiation on aquatic organisms. NCRP Report No. 109. National Council on Radiation Protection and Measurements, Bethesda.

NEFCO, 1996. Proposals for environmentally sound investment projects in the Russian part of the Barents Region, Volume 2: Radioactive contamination. The Nordic Environment Finance Corporation, Barents Region Environmental Programme, Phase I report, prepared by the AMAP Expert Group. AMAP, Oslo. 41p

Noie, A., P.H. Park, Yu.S. Tsaturov and M.S. Baxter, 1997. Investigation of the environmental radioactivity in waste dumping areas in the Northwest Pacific Ocean. Results from the second stage Japanese-Korean-Russian joint expedition 1995. Joint Japanese–Korean–Russian–IAEA Expert Group.

NRPA, 1993. A survey of artificial radionuclides in the Kara Sea. Results from the Russian-Norwegian 1992 expedition to the Barents and Kara seas. Norwegian Radiation Protection Authority, Østerås.

NRPA, 1996. Håndbok for atomberedskap i Norden. Norwegian Radiation Protection Authority, Østerås. In Norwegian.

NRPA, 1999. NRPA Bulletin, Seminar on waste treatment and disposal. StrålevernInfo, pp. 3-99. Norwegian Radiation Protection Authority, Østerås.

NRPA, 2001. Support in the development of regulatory procedures for licensing Lepse waste management operations. Final Report. Norwegian Radiation Protection Authority, Østerås.

NRPA, 2002. The Norwegian assistance program for increased reactor safety in Eastern Europe. StrålevernRapport 2002:3. Norwegian Radiation Protection Authority, Østerås.

Ølgaard, P., 2001. The potential risks from Russian nuclear ships. NKS-57. Nordic Society for Radiation Protection, Roskilde.

OSPAR, 1998. Report of the Ministerial Meeting of the OSPAR Commission. Sintra, Portugal, 22-23 July, 1998. OSPAR Commission, London.

Paatero, J., T. Jaakkola and T.K. Ikäheimonen, 2002. Regional distribution of Chernobyl-derived plutonium deposition in Finland. *Journal of Radioanalytical and Nuclear Chemistry-Articles*, 252(2):407-412.

Pálsson, S.E., 1996. Fallout from atmospheric nuclear tests in the sixties. Review of data from Iceland. *In*: Proceedings of the seventh Nordic Seminar on Radioecology. Reykjavík, pp. 379-384.

Pálsson, S.E., B.J. Howard and S.M. Wright, 2002a. Optimising methods of averaging when using precipitation data to predict radionuclide deposition. *In:* Børretzen, P., T. Jølle and P. Strand (eds.). Proceedings of the International Conference on Radioactivity in the Environment, CD. Monaco, 1-5 September, 2002.

Pálsson, S.E., Ó. Arnalds, M.A. Sigurgeirsson, K. Gudnason, B.J. Howard, S.M. Wright and Th. Pálsdóttir, 2002b. Cs-137 fallout inventories in Iceland – estimating deposition from precipitation data. *Radioprotection*, 37(C1):1223-1228.

Panteleyev, G.P., H.D. Livingston, F.L. Sayles and O.N. Medkova, 1995. Deposition of plutonium isotopes and Cs-137 in sediments of the Ob delta from the beginning of the nuclear age. *In*: Strand, P. and A. Cooke (eds.). Proceedings of the second International Conference on Environmental Radioactivity in the Arctic. Oslo, pp. 57-64.

Pentreath, R.J., 1998. Radiological protection criteria for the natural environment. *Radiation Protection Dosimetry*, 75:175-179.

Pentreath, R.J., 1999. A system for radiological protection of the environment: some initial thoughts and ideas. *Journal of Radiological Protection*, 19:117-128.

Pentreath, R.J. and D.S. Woodhead, 1988. Towards the development of criteria for the protection of marine fauna in relation to the disposal of radioactive wastes into the sea. Radiation Protection in Nuclear Energy, Vol. 2: 213-243. International Atomic Energy Agency, Vienna.

Pfirman, S.L., H. Eicken, D. Bauch and W.F. Weeks, 1995. The potential transport of pollutants by Arctic sea ice. *Science of the Total Environment*, 159(2-3):129-156.

Pfirman, S.L., J.W. Kogeler and I. Rigor, 1997. Potential for rapid transport of contaminants from the Kara Sea. *Science of the Total Environment*, 202:111-122.

Pollard, D., V. Smith., M. Fegan and J.D. Cunningham, 1998. Technetium-99 contamination in the Irish marine environment, 1988-1998. Internal meeting document no. OSPAR 98/4/8, OSPAR Commission, London, 14p.

Poole, A.J., D.C. Denoon and D.S. Woodhead, 1997. The distribution and inventory of ^{137}Cs in sub-tidal sediments of the Irish Sea. *In*: Guegueniat, P., P. Germain and H. Metivier (eds.). Proceedings of the International Symposium on Radionuclides in the Oceans, pp. 263-270. Cherbourg, 7-11 October, 1996.

Raisbeck, G.M. and F. Yiou, 1999. ^{129}I in the oceans: origins and applications. *Science of the Total Environment*, 237/238:31-41.

Rioseco, J.L., 1987. Technetium-99: radioanalytical and radioecological studies. PhD. Thesis. Department of Radiation Physics, Lund University.

Rissanen, K. and T.K. Ikäheimonen, 2000. Cesium and plutonium concentrations in salmon caught in the river Teno (Norwegian Sea) and in river Tornionjoki (Gulf of Bothnia). *In:* S.P. Nielsen (ed.). The Radiological Exposure to the Population of the European Community to Radioactivity in the Baltic Sea, pp. 439-448. Commission of the European Communities, Luxembourg.

Rissanen, K, T. Rahola and E. Illukka, 1987. Radioactivity in plants and foodstuffs in Lapland 1979-1986. Studies on Environmental Radioactivity in Finland in 1986. STUK-A55, June 1987. pp. 25-55.

Rissanen, K., D. Matishov and G.G. Matishov, 1995. Radioactivity levels in Barents, Pechora, Kara, Laptev and White Seas. *In:* Strand, P. and A. Cooke (eds). Proceedings of the second International Conference on Environmental Radioactivity in the Arctic. Oslo, 21-25 August, 1995. pp. 208-214.

Rissanen, K., T.K. Ikäheimonen, D. Matishov and G.G. Matishov, 2000. Radioactivity levels in fish, benthic fauna, seals and seabirds collected in the northwest Arctic of Russia. *Radioprotection – Colloques*, 32(C2):323-331.

Rissanen, K., T.K. Ikäheimonen, J. Ylipieti, D.G. Matishov and G.G. Matishov, 2001. Plutonium in algae, sediments and biota in the Barents, Pechora and Kara Seas. *In:* Inaba, J., S. Hisamatsu and Y. Ohtsuka (eds.). Distribution and Speciation of Radionuclides in the Environment, pp. 107-114. Institute for Environmental Sciences, Aomori.

Rissanen, K., J. Ylipieti and Y. Norokorpi, 2002. ^{137}Cs concentrations in mushrooms collected in 1989-2000 at different forest stands in Kivalo experimental area. *In:* S.E. Pálsson (ed.). Summaries of studies carried out in the NKS/BOK-2 project, Technical Report, NKS-35. pp. 65-69.

Rose, K.S.B., 1992. Lower limits of radiosensitivity in organisms excluding man. *Journal of Environmental Radioactivity*, 15: 113-133.

Roshydromet, 1993. *The Radiation Situation in the Territory of Russia and Adjacent States in 1992*. Annual book. K.P. Makhonko (ed.). Russian Federal Service for Hydrometeorology and Monitoring of the Environment (Roshydromet).

Roshydromet, 1998. *The Radiation Situation in the Territory of Russia and Adjacent Countries in 1997*. K.P. Makhonko (ed.). Russian Federal Service for Hydrometeorology and Monitoring of the Environment (Roshydromet). Hydrometoizdat, St. Petersburg.

Roshydromet, 2000. *The Radiation Situation in the Territory of Russia and Adjacent States in 1998*. K.P. Makhonko (ed.). Russian Federal Service for Hydrometeorology and Monitoring of the Environment (Roshydromet). Hydrometoizdat, St. Petersburg.

Roshydromet, 2001. *The Radiation Situation in the Territory of Russia and Adjacent States in 1999*. K.P. Makhonko (ed.). Russian Federal Service for Hydrometeorology and Monitoring of the Environment (Roshydromet). Hydrometoizdat, St. Petersburg.

RTCP, 1999. Long-term consequences of radioactive contamination in the Northern Areas. Interim report from the fieldwork performed in 1998-1999 in co-operation with the Norwegian Radiation Protection Authority. M.I. Balonov (ed.). Research and Technical Center 'Protection' (St. Petersburg).

RTCP, 2000. Long-term consequences of radioactive contamination in the Northern Areas. Interim report from the fieldwork performed in 1990-2000 in co-operation with the Norwegian Radiation Protection Authority. M.I. Balonov (ed.). Research and Technical Center 'Protection' (St. Petersburg).

Saltbones, J., A. Foss and J. Bartnicki, 1995. *SNAP: Severe Nuclear Accident Program. A Real Time Dispersion Model for Major Nuclear Emergency Management*. DNMI Research report No. 15. Det norske meteorologiske institutt, Oslo.

Saltbones, J., J. Bartnicki and A. Foss, 1997. *Atmospheric Transport and Deposition from a Potential Accident at Kola Nuclear Power Plant. Part 2: Worst Case Scenarios*. DNMI Research report No. 56. Det norske meteorologiske institutt, Oslo.

Sarkisov, A.A., 1999. Radiological consequences of the radiation accident on nuclear submarine in Chazhma Bay. *In*: Strand, P. and T. Jølle (eds.). Proceedings of the fourth International Conference on Environmental Radioactivity in the Arctic. Edinburgh, 20-23 September, 1999, pp. 78-80.

Sayles, F.L., T.C. Kenna and H.D. Livingston, 1999. Release and transport of artificial radionuclides from nuclear weapons related activities in the Ob river, Siberia, over the course of the nuclear age. *In*: Proceedings of an International Symposium on Marine Pollution, pp. 565-567. Monaco, October 5-9, 1998. International Atomic Energy Agency, Vienna.

Saxen, R., 2003. Long-term behaviour of ^{90}Sr and ^{137}Cs in surface waters in Finland. *In:* V. Paile (ed.). Radiological Protection in the 2000s – Theory and Practice. Turku, Finland, August 25-

29, 2002. Nordic Society for Radiation Protection, Roskilde. pp. 479-483.

Saxen, R. and E. Ilus, 2001. Discharge of Cs-137 and Sr-90 by Finnish rivers to the Baltic Sea in 1986-1996. *Journal of Environmental Radioactivity*, 54:275-291.

Schink, D.R., P.H. Santschi, O. Corapcioglu, P. Sharma and U. Fehn, 1995. ^{129}I in Gulf of Mexico waters. *Earth and Planetary Science Letters*, 135:131-138.

Shutov, V.N., G.Ya. Bruk, I.G. Travnikova, M.I. Balonov, M.V. Kaduka, L.N. Basalaevam, L. Skuterud, H. Mehli and P. Strand, 1999. The current radioactive contamination of the environment and foodstuffs in the Kola region of Russia. *In:* Strand, P. and T. Jølle (eds.). Proceedings of the fourth International Conference on Radioactive Contamination in the Arctic, Edinburgh, pp. 307-309.

Sigurgeirsson, M.Á., Ó. Arnalds, S.E. Pálsson, K. Guðnason and B.J. Howard, 2002. Behaviour of radiocaesium fallout in volcanic soils in Iceland. The fifth International Conference on Environmental Radioactivity in the Arctic and Antarctic. St. Petersburg, pp. 294-296.

Sivintsev, Yu and O. Kiknadze, 1998. Components of nuclear reactors dumped in the Kara Sea and the Sea of Japan, and assessment of radionuclides contained in them. *In*: Afterword to the 'White Book' (Yablokov Commission), pp. 172-181. Proceedings of the International Seminar, January 19-21, 1998. International Science and Technology Center, CDB 'Lazurit'- Nizhni Novgorod.

Sivintsev, Yu.V., V.L. Vyotsky and V.A. Danilyan, 1994. Radiological impact of radiation accident on nuclear submarine in Chazhma Bay. *Atomic Energy*, 76:158-160.

Skuterud, L., L.D. Blytt, S. Borghuis, P. Strand and B. Howard, 1999. Assessment of relative importance of food production systems in Europe – A first approach. Technical deliverable from the project 'Spatial analysis of vulnerable ecosystems in Europe (SAVE)'. EC Contract FI14-CT95-0015. Norwegian Radiation Protection Authority, Østerås..

Smith, G. and A. Amundsen, 2002. Overview of Norwegian bilateral initiatives on environmental protection and nuclear safety in north west Russia. *In:* Strand, P. and T. Jølle (eds.). Proceedings of the fifth International Conference on Radioactive Contamination in the Arctic and Antarctic. St Petersburg, 16-20 June, 2002. pp. 141.

Smith, J.N., K.M. Ellis and L.R. Kilius, 1998. ^{129}I and ^{137}Cs tracer measurements in the Arctic Ocean. *Deep-Sea Research I*, 45: 959-984.

Smith, J.N., K.M. Ellis and T. Boyd, 1999. Circulation features in the central Arctic Ocean revealed by nuclear fuel reprocessing tracers from Scientific Ice Expeditions 1995 and 1996. *Journal of Geophysical Research*, 104:29663-29667.

Smith, V., M. Fegan, D. Pollard, S. Long, E. Hayden and T.P. Ryan, 2001. Technetium-99 in the Irish marine environment. *Journal of Environmental Radioactivity*, 56:269-284.

Sneve, M.K., C. Bergman, M. Westerlind, I. Sokolova and V. Markarov, 2001. Phase 1 of the regulatory Lepse project: project report. StrålevernRapport 2001:4. Norwegian Radiation Protection Authority, Østerås. 121p.

State of Alaska and Aleutian/Pribilof Island Association, 2001. White Paper: radiological assessment and long-term stewardship, Amchitka Island, Alaska, underground nuclear test area. Alaska Department of Environmental Conservation, 5p.

Stokke, E., 1997. Kola nuclear power plant. Consequence assessment for hypothetical severe accidents. Source term assessment. Research report. IFE/HR/F-97/1088. Institute for Energy Technology, Halden.

Strand, P., 2002. Protection of the environment and radioecology. *In:* Børretzen, P., T. Jølle and P. Strand (eds.). Proceedings of the International Conference on Radioactivity in the Environment. Monaco, 1-5 September, 2002, pp. 17-21.

Strand, P. and D.H. Oughton, 2002. Radiation protection in the 21st century: Ethical, philosophical and environmental issues. Consensus conference on protection of the environment from ionising radiation. Norwegian Radiation Protection Authority and Agricultural University of Norway.

Strand, P., M. Sickel, A. Aarkrog, J.M. Bewers, Y. Tsaturov and S. Magnusson, 1996. Radioactive contamination of the Arctic marine environment. *In*: Radionuclides in the Oceans. Inputs and Inventories, pp. 96-119. Institut de Protection et de Sûreté Nucléaire, Cherbourg.

Strand, P., A.I. Nikitin, B. Lind, B. Salbu and G.C. Christensen, 1997. Dumping of radioactive waste and investigation of radioactive contamination in the Kara sea. Results from 3 years of investigations (1992-1994) in the Kara Sea. Joint Norwegian-Russian Expert Group for investigation of radioactive contamination in the northern areas. Norwegian Radiation Protection Authority, Østerås.

Strand, P., J.E. Brown, D.S. Woodhead and C-M. Larsson, 2000. Delivering a system and framework for the protection of the environment from ionising radiation. *In*: Proceedings of the tenth International Congress of the International Radiation Protection Association. Hiroshima. (CD Publication).

Strand, P., N. Beresford, R. Avila, S. Jones and C-M. Larsson, 2001. Identification of candidate reference organisms from a radiation exposure pathways perspective. Deliverable of the FASSET project. Swedish Radiation Protection Institute, Stockholm.

Strand, P., B.J. Howard, A. Aarkrog, M. Balonov, Y. Tsaturov, J.M. Bewers, A. Salo, M. Sickel, and K. Rissanen, 2002. Radioactive contamination in the Arctic – sources, dose assessment and potential risks. *Journal of Environmental Radioactivity*, 60:5-21.

STUK, 1983. Studies on environmental radioactivity in Finland 1982. Annual Report. Report STL-A47. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

STUK, 1992. Limitation of public exposure in the environment of and limitation of radioactive releases from nuclear power plants (YVL 7). Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

STUK, 1998. Compliance with the general regulations for the safety of nuclear power plants (Decision 395/1991) – The Loviisa plant (STUK-B-YTO 179) and the Olkiluoto plant (STUK-B-YTO 180). Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

STUK, 2000. Finnish support programme for nuclear safety: progress report biannual summary 2000. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki. 57p + Ann.

Tossavainen, K. (ed.), 1998. Regulatory control of nuclear safety in Finland. Annual report 1997. STUK-B-YTO 175. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

Tossavainen, K. (ed.), 1999. Regulatory control of nuclear safety in Finland. Annual report 1998. STUK-B-YTO 192. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

Tossavainen, K. (ed.), 2000. Regulatory control of nuclear safety in Finland. Annual report 1999. STUK-B-YTO 202. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

Tossavainen, K. (ed.) 2001. Regulatory control of nuclear safety in Finland. Annual report 2000. STUK-B-YTO 216. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki.

Tossavainen, K. (ed.), 2002. Regulatory control of nuclear safety in Finland. Annual report 2001. STUK-B-YTO 216. Finnish Radiation and Nuclear Safety Authority (STUK), Helsinki. 43p. + App. 22p.

Turvey, F.J. and C. Hone, 2000. Storage of Liquid High-level Radioactive Waste at Sellafield. Radiation Protection Institute of Ireland, Report RPII-00/3.

UNCED, 1992. Report of the United Nations Conference on Environment and Development. Rio Declaration on Environment and Development. United Nations, New York.

UNSCEAR, 1996. Effects of radiation on the environment. Annex 1. *In*: Sources and Effects of Ionizing Radiation. UNSCEAR 1996 Report to the United Nations General Assembly. United Nations Scientific Committee on the Effects of Atomic Radiation.

UNSCEAR, 2000. *Sources and Effects of Ionizing Radiation*, United Nations Scientific Committee on the Effects of Atomic Radiation UNSCEAR 2000 Report to the United Nations General Assembly with scientific annexes. Volume I: Sources.

Uranium Institute, 1998. Technetium-99 discharges from Sellafield. Internal meeting document No. OSPAR 98/4/NGO.2. OSPAR Commission, London, 11p.

U.S. Air Force, 1970. Project Crested Ice. *USAF Nuclear Safety*, 65:1-97.

U.S. Air Force, 1999. Environmental assessment for the Burnt Mountain seismic array power supply. 354 CES/CEVP, Eielson Air Force Base, Alaska.

U.S. Congress, 1994. Power sources for remote Arctic applications. OTA-BP-ETI-129. Office of Technology Assessment, Washington DC.

U.S. DOE, 2002. DOE Standard. A graded approach for evaluating radiation doses to aquatic and terrestrial biota. DOE STD-1153-2002. U.S. Department of Energy, Washington DC.

U.S.S.R. Goshydromet, 1991. The radiation situation in the territory of Russia and adjacent states in 1990. K.P. Makhonko (ed.). SPA Typhoon, Obninsk.

Vakulovsky, S.M., 1993. Transport of artificial radioactivity by the Ob to the Arctic Seas. *In*: Strand, P. and E. Holm (eds.). Proceedings of the International Conference on Environmental Radioactivity in the Arctic and Antarctic. Kirkenes, 23-27 August, 1993, pp. 107-110.

Vakulovsky, S.M., I.I. Kryshev, A.I. Nikitin, Yu.V. Savitsky, S.V. Malyshev and E.G. Tertyshnik, 1995. Radioactive contamination of the Yenisei River. *Journal of Environmental Radioactivity*, 29(3):225-236.

Van Oostdam, J., A. Gilman, E. Dewailly, P. Usher, B. Wheatley, H. Kuhnlein, S. Neve, J. Walker, B. Tracy, M. Feeley, V. Jerome and B. Kwavnick, 1999. Human health implications of environmental contaminants in Arctic Canada: a review. *Science of the Total Environment*, 230:1-82.

Veðráttan, 1959-1983. [A general reference to Veðráttan – a monthly journal of the Icelandic Meteorological Office, Reykjavik].

Vestergaard, T. and P. Zachariassen, 1987. Føðslukanning 1981-82 (Nutritional investigation 1981-82). *Annales Societatis Scientiarum Færoensis (Fróðskaparrit)*, vol. 33. In Faroese.

Vintró, L.L., C.A. McMahon, P.I. Mitchell, D. Josefson, E. Holm and P. Roos, 2002. Transport of plutonium in surface and subsurface waters from the Arctic shelf to the North Pole via the Lomonosov Ridge. *Journal of Environmental Radioactivity*, 60(1-2):73-89.

Webb, G., 2002. International approaches to prevention of and response to radiological terrorism. Forty-seventh Annual meeting of the Health Physics Society, abstracts of papers presented at the meeting. *Health Physics*, 82(6):S146.

Weeks, W., 1994. Possible roles of sea ice in the transport of hazardous material. *Arctic Research of the United States*, 8:34-52.

Wheatcraft, S.W., 1995. Sea water intrusion model of Amchitka Island, Alaska. U.S. Department of Energy, Report#DOE/NV/10845-59.

WISE-Paris, 2001a. Possible toxic effects from the nuclear reprocessing plants at Sellafield (UK) and Cap de La Hague (France). Final report for STOA (European Parliament Scientific and Technical Options Assessment). August 2001.

WISE-Paris, 2001b. Airliner crash on nuclear facilities – the Sellafield case. WISE-Paris. October 2001.

Wright, S.M., B.J. Howard, P. Strand, T. Nylén and M.A.K. Sickel, 1999. Prediction of ^{137}Cs deposition from atmospheric nuclear weapons tests within the Arctic. *Environmental Pollution*, 104: 131-143.

Yablokov, (ed.), 1993. Facts and Problems Related to Radioactive Waste Disposal in Seas Adjacent to the Territory of the Russian Federation. ('Yablokov' Report'). In Russian (Report of the Governmental Commission, Moscow). In English (Albuquerque, Small World Publishers), 82 p.

Yiou, F., G.M. Raisbeck, Z.Q. Zhou, L.R. Kilius and P.J. Kershaw, 1995. Improved estimates of oceanic discharges of ^{129}I from Sellafield and la Hague. *In*: Strand, P. and A. Cooke (eds.). Proceedings of the second International Conference on Environmental Radioactivity in the Arctic, pp 113-116. Oslo, August 21-25, 1995.

Yiou, F., G.M. Raisbeck, G.C. Christensen and E. Holm, 2002. ^{129}I/^{127}I, ^{129}I/^{137}Cs and ^{129}I/^{99}Tc in the Norwegian coastal current from 1980 to 1998. *Journal of Environmental Radioactivity*, 60:61-71.

Glossary

Abbreviations

AMAP . . . . . Arctic Monitoring and Assessment Programme
AO . . . . . Autonomous Okrug
BAT . . . . . Best available technology
BNFL . . . . . British Nuclear Fuels
CCI. . . . . Common cause initiator
CEG . . . . . (IAEA) Contact Expert Group
CFSU . . . . . central former Soviet Union
CMF . . . . . Core Melt Frequency
COSR . . . . . Continued Operation Safety Report
DNA . . . . . Deoxyribonucleic acid
dw . . . . . dry weight (basis of determination)
EARP. . . . . Enhanced Actinide Removal Plant
EC . . . . . European Commission
EIA . . . . . Environmental impact assessment
EMAS . . . . . (EC) Eco-Management and Audit Scheme
EPIC . . . . . Environmental Protection from Ionising Contaminants in the Arctic (project)
EU . . . . . European Union
FASSET. . . . . Framework for Assessment of Environmental Impact (project)
HAL . . . . . Highly Active Liquor
HSE . . . . . (U.K.) Health and Safety Executive
IAEA . . . . . (UN) International Atomic Energy Agency
IASAP. . . . . International Arctic Seas Assessment Project
ICRP . . . . . International Commission on Radiological Protection
IMO . . . . . (UN) International Maritime Organization
INES . . . . . International Nuclear Event Scale
IUR. . . . . International Union of Radioecologists
JNREG. . . . . Joint Norwegian–Russian Expert Group on Environmental Cooperation
KMCIC . . . . . Krasnoyarsk Mining and Chemical Industrial Complex
LOCA . . . . . Loss of coolant accidents
NAO . . . . . Nenets Autonomous Okrug
NATO . . . . . North Atlantic Treaty Organization
NEFCO. . . . . Nordic Environment Finance Corporation
NPP. . . . . Nuclear power plant
NRPA . . . . . Norwegian Radiation Protection Authority
OSPAR . . . . . Oslo and Paris Commissions
RHS . . . . . Radioisotopic heat source
RPII . . . . . Radiation Protection Institute of Ireland
RTG . . . . . Radioisotope thermoelectric generator
SCC . . . . . Siberian Chemical Combine
SD . . . . . Standard deviation
SIPEX . . . . . Site Ion-Exchange Effluent Plant
STUK. . . . . Finnish Radiation and Nuclear Safety Authority
T_{ag} . . . . . Aggregated transfer coefficient (see Box 3.2)
T_{ec} . . . . . Ecological half-life (see Box 3.1)
T_{eff} . . . . . Effective ecological half-life (see Box 3.1)
TNT . . . . . trinitrotoluene
U.K. . . . . United Kingdom
UNSCEAR . . . . United Nations Scientific Committee on the Effects of Atomic Radiation
US DOE . . . . . United States Department of Energy
ww . . . . . wet weight (or fresh weight) (basis of determination)

Main radionuclides discussed

Am . . . . . Americium
Cs . . . . . Cesium, caesium
I . . . . . Iodine
Pu . . . . . Plutonium
Sr. . . . . Strontium
Tc . . . . . Technetium
U . . . . . Uranium

Prefixes

μ . . . . . micro-, 10^{-6}, e.g. μSv
G . . . . . giga-, 10^{9}, e.g. Gw
k . . . . . kilo-, 10^{3}, e.g. kt, kBq
m. . . . . milli-, 10^{-3}, e.g. mBq
M . . . . . mega-, 10^{6}, e.g. Mt, MW, MBq
P . . . . . peta-, 10^{15}, e.g. PBq
T . . . . . tera-, 10^{12}, e.g. TBq

Units

Bq . . . . . Becquerel
Ci . . . . . Curie
d . . . . . day(s)
Gy . . . . . Gray
h . . . . . hour(s)
Mt . . . . . Megaton
MW(e) . . . . . Megawatt (electrical energy)
MW(th) . . . . . Megawatt (thermal energy)
Sv. . . . . Sievert
t . . . . . tonne (1000 kg)
yr. . . . . year(s)