Chapter 8

Radioactivity

Contents

8.1. Introduction

This chapter deals with the assessment of radioactive contamination of the environment, radiation sources and associated radiological consequences within the Arctic. The purpose of this chapter is to provide a balanced appreciation of the nature and risks posed by radionuclides in the Arctic derived from all relevant and known sources. Initially, a simplified explanation of the basis of radiological protection and the procedures for estimating radiological doses and risks is provided. The chapter subsequently deals with doses associated with existing radioactive contamination of the Arctic environment, routine releases from nuclear operations within, and close to, the Arctic, previous accidents in civil and military nuclear activities that result in exposures to Arctic residents, and potential releases from both such installations and the various packages of high-level waste reposing in the environment, such as those dumped in the Kara Sea by the former Soviet Union. This is followed by an evaluation of radiological vulnerability in the Arctic. The chapter ends with conclusions and recommendations.

The chapter was prepared under the guidance of an assessment group comprising scientists from the contracting parties to the international Arctic Environmental Protection Strategy, or Rovaniemi Agreement. Several other individuals have made substantial contributions to the report and the data upon which the report is based. In the preparation of this chapter, the explanatory text, data assembly and preparation of individual-related radiological assessments were provided by the assessment group and national staff. Most of the source-related assessments in the document, on the other hand, are based on studies carried out under the aegis of other agencies, either national or international. The interpretation and representation of these latter studies have been carried out by the assessment group in connection with the preparation of this document.

To the extent that appropriate data and information has been made available to the assessment group, the assessment goal has been achieved. Inevitably, however, because of the heterogeneity and varying comprehensiveness of the information available, some sections of the document are more complete and detailed than others.

The assessment serves to document what is currently known about radioactivity from sources in the Arctic and associated risks and effects. It also identifies where additional efforts are required to obtain more information or conduct additional assessments to improve the characterisation of the risks associated with specific human and industrial activities.

8.2. Fundamentals and definitions

8.2.1. Radioactivity

Radioactivity is the property of spontaneous disintegration, or decay, of atomic nuclei accompanied by the emission of ionizing radiation. **Activity** corresponds to the number of disintegrations per second of an isotope (with dimensions T^{-1}). The SI (*Standards Internationaux*) unit of activity is the reciprocal second (s^{-1}) with the name ***Becquerel*** (Bq). The older, non-SI, unit ***Curie*** (Ci) that was derived from the (presumed) activity of one gram of radium and is still used in some fora, corresponds to 3.7×10^{10} Bq. The major forms of ionizing radiation emitted during radioactive decay are ***alpha particles***, which are essentially charged helium nuclei, ***beta particles***, which are electrons, and ***gamma rays***, which are photons or electromagnetic waves. The nature, energy, charge and penetrating power of radiation is of relevance to the consequences of biological exposures. This is dealt with in more detail later in this introductory section.

The term '***radionuclide***' applies to all radioactive isotopes of all elements. The term '***radioisotope***' strictly refers to the radioactive isotope of an element having other isotopes of similar chemical properties but differing nuclear properties. These may include both stable and radioactive isotopes. The physical half-life of a radionuclide defines the time required for the activity of that radionuclide to decay, by purely physical processes, by a factor of two.

8.2.1.1. Natural radioactivity

Natural radioactivity is derived from the decay of nuclei in the Earth's crust and by the bombardment of the Earth by cosmic radiation producing radionuclides in the Earth's atmosphere. These natural radionuclides fall into three categories: the very long-lived **primordial radionuclides** (^{40}K,

^{238}U, ^{232}Th, ^{235}U) formed at the time the Earth was created; **decay chain radionuclides** (radionuclides in the uranium, thorium and actinium decay series) that are the products of decay of primordial nuclides; and **cosmogenic nuclides** produced by the interaction of high energy cosmic radiation with the Earth's atmosphere (e.g., ^{3}H, ^{7}Be, ^{14}C, ^{22}Na).

8.2.1.2. **Artificial radioactivity**

In the early days of the 20th Century, human abilities to create artificial radioactive sources were limited to chemical isolation and the concentration of natural radionuclides. Later in this century, linear accelerators were developed for producing beams of particles that could be used to artificially transmute nuclei. With the application of nuclear fission, for both peaceful and military purposes in the 1940s, the ability of humans to produce large quantities of artificial radionuclides was greatly expanded. The fission process itself, and the high neutron flux densities achieved in nuclear weapons explosions and fission reactor cores, led to the production of large quantities of fission and activation products. Fission products are the isotopes with atomic masses in the 70-170 range, formed by thermal fission of ^{235}U and other heavy fissile nuclei (e.g., ^{239}Pu). High-yield fission products include ^{89}Sr, ^{90}Sr, ^{91}Y, ^{95}Zr, ^{95}Nb, ^{99}Mo, ^{103}Ru, ^{131}I, ^{133}Xe, ^{137}Cs, ^{140}Ba, ^{140}La, ^{141}Ce, ^{144}Ce, ^{143}Pr and ^{147}Nd. However, in most situations, the most radiologically important fission products in the short term are ^{89}Sr, ^{90}Sr, ^{131}I and ^{137}Cs, and in the long term, ^{90}Sr and ^{137}Cs, because of their yields, half-lives and chemical properties. Activation products are the isotopes formed principally by the capture of neutrons by stable isotopes in high neutron flux environments. Typical activation products formed in the structure of nuclear reactors include ^{51}Cr, ^{54}Mn, ^{55}Fe, ^{59}Fe, ^{60}Co, ^{63}Ni, ^{65}Ni, ^{64}Cu, ^{65}Zn, ^{69}Zn, ^{110}Ag, ^{109}Cd, ^{134}Cs, ^{236}U and ^{239}U. These radionuclides are generally neutron-rich and decay primarily by gamma-ray and beta particle emission. Some activation products are isotopes of elements of atomic number larger than uranium and these are referred to as 'transuranic' nuclides. Prime examples are ^{239}Np and ^{239}Pu that are created in reactors as a result of the β-decay of the short lived activation product ^{239}U.

As a result of the use of fission reactors for electrical power generation, there are large quantities of fission products in spent nuclear fuel assemblies. There is also a large inventory of activation products in reactor assemblies and in other materials such as ^{60}Co radiation sources deliberately or inadvertently exposed to significant neutron fluxes. As a result of nuclear fuel reprocessing for the recovery of plutonium, substantial quantities of fission and activation products have been released to the environment in wastes.

Nuclear weapons explosions have provided the largest inventory of both fission and activation products in the global environment, and many of these have been, and remain, detectable world-wide. Indeed, nuclear explosions have produced the most pronounced global change in the character of environmental radioactivity. Various nuclear accidents have further contributed to the inventory of radionuclides in the environment. In recent years, the use of radiothermal power generators for space vehicles has, as a result of accidents, given rise to additional isotopes detectable in the environment, most notably ^{238}Pu.

8.2.2. Effects of radionuclides

The main concern about radionuclides and radiation are their adverse effects on organisms, including humans. However, it must be remembered that ionizing radiation has medical benefits in diagnosis and treatment of disease as well as in several industrial applications. Both facets of the existence and use of radionuclides and radiation have led to the creation of a major discipline called ***radiological protection***. Other terms, such as '*radiology*' and '*health physics*', originally used in a wider context, are now almost exclusively used in connection with nuclear medicine. The entire focus of radiological protection is the effects of radiation on living tissues and organisms, and mechanisms for the adequate protection of both deliberately and accidentally exposed humans and populations of other organisms. The remaining text of this section is intended to provide a synopsis of the basis and nature of health protection from the effects of radiation and the generation and use of radionuclides, including the regulation of the nuclear power industry. Intentionally, this synopsis does not go into great detail – it merely serves as background to much of the text of later sections of this chapter.

8.2.2.1. **The concept of risk**

There are many definitions of risk. Risk relates to quantities such as the probability that specific deleterious consequences may arise and the magnitude and character of such consequences. In this assessment, the term ***risk*** is used to mean the probability – the likelihood – that something unpleasant will happen. Clearly, however, the likelihood of an adverse happening cannot be considered outside of the context of the severity of the associated effect. If the consequences of happenings of equal probability are respectively fatality or minor personal financial loss, the former is going to be respected and considered far more seriously than the latter. A related term is that of hazard. A hazard is essentially a 'set of circumstances' that may result in harmful consequences. Harm is generally taken to include adverse effects on health or the quality of life; it can also be expressed in terms of loss, including loss of life, of working days, or material items, such as environmental amenities or money. It is often possible, therefore, to represent adverse effects as costs to society. Because costs are also incurred in reducing risks, the two sets of costs have often been used to estimate the optimum 'value for money' in relation to measures taken to reduce risk. In absolute terms, no set of human circumstances is entirely safe but, obviously, the lower the risk, the higher the degree of safety. The two terms (risk and safety) are, therefore, inversely related and what most people perceive as being 'safe' actually corresponds to an acceptable level of risk.

8.2.2.2. **Health effects and units of dose**

Exposure to radiation can cause detrimental health effects. At large acute doses, radiation effects – such as opacities in the lens of the eye sometimes leading to cataract, temporary or permanent sterility and, in severe cases of whole body irradiation, acute syndromes (such as damage to bone marrow, gastrointestinal tract, lungs and the nervous system) – can lead to death within a short period of time after exposure. Large chronic dose rates also cause clinically detectable deleterious effects. These various effects are called ***deterministic*** because they are **certain to occur** if the dose exceeds certain threshold levels.

At low doses, radiation exposure can also plausibly induce severe health effects, such as malignancies, which are statistically detectable in a population, but cannot be unequivocally associated with individual exposures. Hereditary effects due to radiation exposure have been statistically detected in mammals and are presumed to occur in humans as

well. All these statistically detectable effects are called ***stochastic*** effects because of their random (i.e., probabilistic) nature. These effects are expressed after a latency period, presumably over the entire range of doses without a threshold level. In addition, there is a possibility of health effects in children exposed to radiation *in utero* during certain periods of pregnancy, including a greater likelihood of leukaemia and severe mental retardation.

The fundamental dosimetric quantity in radiological protection is the ***absorbed dose***. This is the energy absorbed per unit mass and is expressed in units of joules per kilogram and given the name **gray** (Gy). The probability of stochastic effects depends not only on the absorbed dose but also on the type and energy of the radiation causing the dose. However, it is the absorbed dose averaged over a tissue or organ (rather than at a point) and weighted for the radiation type that is pertinent. The ***equivalent dose*** is the term used in a tissue or organ when these two components have been taken into account through the use of appropriate weighting factors. The relationship between the probability of stochastic effects and equivalent dose depends on the organ or tissue irradiated. It is, therefore, appropriate to define a further quantity, derived from equivalent dose, to indicate the combination of different doses to several different tissues in a way that correlates with the total of the stochastic effects. Once a weighting factor is introduced to account for the relative contribution of each organ or tissue to the total detriment resulting from uniform irradiation of the whole body, the term used to characterize the dose is ***effective dose***. The effective dose is then the sum of the weighted equivalent doses in all the tissues and organs of the body. While the units of effective dose are still joules per kilogram, it is given the name **sievert (Sv).**

Natural radiation and exposures

Cosmic radiation and ionizing radiation from radionuclides in the environment provide the major source of human radiation exposure.

The term 'cosmic radiation' refers both to the primary high-energy particles of extraterrestrial origin that strike the Earth's atmosphere and to the secondary particles generated by their interaction with the atmosphere. The primary galactic particles entering the Earth's atmosphere are high-energy protons (≈90%) and alpha-particles (≈10%).

Lower-energy charged particles are deflected back into space by the Earth's magnetic field. This effect is latitude-dependent and there is a greater flux of incident low-energy protons at the poles than at the equator, resulting in an increase in the dose rate at high latitudes. Furthermore, this latitude effect increases with altitude.

Buildings provide some shielding against the directly ionizing component of cosmic radiation, but the magnitude of the shielding depends strongly on the structural composition and thickness of the building material. The shielding effect of wooden houses reduces the dose rate of the direct ionizing component by less than 5% (Miller and Beck 1984), whereas the reduction is between 35 and 70% for some larger multi-storey concrete buildings (Miller and Beck 1984, Lin *et al.* 1986).

Taking into account shielding by buildings and the distribution of the world population with altitude and latitude, the population-weighted average annual effective dose from cosmic radiation has been estimated by the United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR) to be 380 μSv; the directly and indirectly ionizing components contributing 300 μSv and 80 μSv, respectively. The dose is assumed to be 10-20% higher at high latitudes (>72°N).

Of the large number of radionuclides produced by cosmic radiation only four of them (^{3}H, ^{7}Be, ^{14}C and ^{22}Na) contribute significantly to the dose to humans (NCRP 1987). The most radiologically significant of these four radionuclides is ^{14}C. The annual natural production of ^{14}C is 1 PBq and the specific activity of natural ^{14}C in the body is 230 Bq/kg leading to an annual effective dose of 12 μSv (UNSCEAR 1993). The contributions from the ingestion of ^{3}H, ^{7}Be and ^{22}Na are much smaller.

Primordial radionuclides are usually categorized as either 'series radionuclides' which decay in a chain of radionuclides to a stable isotope of lead, or 'non-series radionuclides' which decay directly to stable nuclides. There are several tens of non-series radionuclides in crystalline rocks and soils. However, most of the non-series radionuclides have a combination of half-life, isotopic abundance, and elemental abundance in the Earth's crust such that they have negligibly small specific activities and are not dosimetrically significant. The only non-series radionuclides having any dosimetric significance are ^{40}K and ^{87}Rb, which are both geochemically similar alkali elements. Whilst ^{87}Rb is a pure β-emitter, ^{40}K decay is accompanied by both β- and γ-radiation. The abundance of ^{40}K in the environment makes it a major source of both internal and external doses from naturally-occurring radiation. ^{40}K in rocks, soils and building materials is also a major contributor to external background radiation. According to UNSCEAR (1988) about 40% of the average annual dose to humans from external radiation is due to ^{40}K in the surroundings. Data on ^{40}K in the human body are well established, mainly from direct whole body measurements of persons of various ages. The average specific activity of ^{40}K in the body of adults is about 55 Bq/kg, averaged over both sexes. The annual effective doses to adults and children are estimated to be 165 and 185 μSv, respectively. Potassium is in homeostatic control in the human body, which means that the dose from ^{40}K is not influenced by the potassium intake with diet.

The radionuclides in the decay series headed by ^{238}U (uranium series), ^{232}Th (thorium series) and ^{235}U (actinium series) are called series radionuclides. The relative abundance of ^{235}U (0.73%) is low compared to ^{238}U (99.2%). The decay products in the actinium series are relatively short-lived. Thus, the actinium series is of much less dosimetric importance than the uranium and thorium series, and will not be discussed further.

Depending on local geology, there are large local and regional variations in outdoor gamma dose rates. Outdoor gamma radiation depends mainly on ^{226}Ra, ^{232}Th and ^{40}K in soil and rock. In certain granites and alum shales, ^{226}Ra activity concentrations of up to 500 and 5000 Bq/kg, respectively, have been found. External gamma radiation indoors, on the other hand, depends mainly on the activity concentration of the building materials. The average outdoor and indoor dose rates in air for the world population have been calculated by UNSCEAR (1993). Based on an indoor occupancy factor of 0.8, the average annual effective dose to the world population is estimated to be 0.46 mSv.

Exposure to ^{222}Rn (radon), ^{220}Rn (thoron) and their progeny comes mainly from the inhalation of the decay products of radon and thoron, which deposit inhomogeneously within the respiratory tract and irradiate the bronchial epithelium. The dose contribution from inhaled radon or thoron gas, both which are highly soluble in body fluids and tissues, is small (≈5%) compared with the doses from their progeny.

Outdoor radon concentrations depend on the amount of radon released from soil and the atmospheric factors controlling its upward dispersion. The annual global radon emis-

sion has been estimated by Harley (1972) to be about 10^{20} Bq and the atmospheric inventory as 1.5×10^{18} Bq. This would give a mean surface radon concentration of approximately 4 Bq/m^3 in the northern hemisphere with higher values of about 8 Bq/m^3 over the continents.

The concentration of radon (and its progeny) is usually much higher indoors than outdoors. Based on the available data, UNSCEAR (1993) estimated that the population-weighted world-wide average radon concentration is 40 Bq/m^3. Indoor surveys in different countries show that some of the highest levels in the world are found in Sweden, Finland and Norway. In these countries, radon activity concentrations two to three orders of magnitude above the average have been reported.

In areas of permafrost within the Arctic Circle, the radon exhalation from the ground is usually low. In addition, houses are usually built without a basement and generally on piles. Consequently indoor radon concentrations are generally low.

The average annual effective dose to the world population from inhalation of radon and its progeny has been estimated by UNSCEAR (1993) to be 1.2 mSv.

The reported activity concentrations of series radionuclides in the body vary widely (NCRP 1987). However, except for internal deposition of radon progeny in the respiratory tract, the only significant source of internally-deposited natural radioactivity results from the ingestion of ^{210}Pb and ^{210}Po. The mean daily intake of ^{210}Pb and ^{210}Po through ingestion is about 0.1 Bq. For populations in the Arctic and subarctic regions, with high consumption of reindeer/caribou meat, the daily intake can increase to more than ten times this mean value (Kauranen and Miettinen 1969). Reindeer breeders in northern Norway and Finland may ingest on average as much as 10 Bq/d of ^{210}Pb and ^{210}Po. High consumers of seafood are also expected to receive higher doses than normal owing to the elevated activity concentrations of ^{210}Pb and ^{210}Po in marine products (UNSCEAR 1993). This exposure route is assumed to be more important for some population groups living in the Arctic due to their high consumption of marine products. Cigarettes each contain about 20 mBq and 15 mBq of ^{210}Pb and ^{210}Po, respectively. Thus, smokers receive higher doses than non-smokers.

The world-wide average committed dose from annual intakes of natural radionuclides (excluding radon) is estimated to be 0.23 mSv, of which 0.17 mSv is from non-series radionuclides (mainly ^{40}K) and 0.06 mSv from radionuclides in the ^{238}U and ^{232}Th series (mainly ^{210}Pb and ^{210}Po) (UNSCEAR 1993). The annual effective dose due to the body content of non-series and series nuclides are estimated to be 0.18 and 0.13 mSv, respectively. Table 8·1 summarizes the doses received from natural exposures derived by UNSCEAR.

Table 8·1. Global average natural radiation doses (UNSCEAR 1993).

Component of exposure	Annual effective dose, mSv/y: in areas of normal background	in areas of elevated exposure[a]
Cosmic rays	0.38	2.0
Cosmogenic radionuclides	0.01	0.01
Terrestrial radiation:		
External exposure	0.46	4.3
Internal exposure (excluding radon)	0.23	0.6
Internal exposure from radon and its decay products		
^{222}Rn inhalation	1.2	10
^{220}Rn inhalation	0.070	0.1
^{222}Rn ingestion	0.005	0.1
Total	2.4	–

a. The elevated values are representative of large regions. The cosmic ray dose rate depends on height above sea level and on latitude. Annual doses in areas of higher exposure (locations with higher elevations) are about five times the average. The dose to a few communities living near some types of mineral sand may be up to about 100 times the average. The dose from radon decay products depends on local geology and housing construction and use, with the dose in some regions being about 10 times the average. Local geology and the type and ventiliation of some individual houses may combine to give exceptionally high dose rates from radon decay products of several hundred times the average.

8.2.3. The system of radiological protection

Many beneficial human activities involve the exposure of people to radiation from both natural and artificial sources. These activities, which are planned in advance, may be expected to increase the exposure that people already receive from natural background radiation. These activities are termed ***practices***. On the other hand, there are radiation exposures incurred *de facto* by people, such as those from natural radionuclides and nuclear accidents. Activities aimed at reducing these exposures are termed ***interventions***. The 'System of Radiological Protection' provides the basic requirements for the protection of people against undue radiation exposures. Its aim is to prevent the occurrence of deterministic effects due to radiation and to restrict the likelihood of stochastic effects.

8.2.3.1. Practices

In radiological protection, the primary objective is to minimize the risks to individuals and the collective detriment to the exposed population. Accordingly, the focus is on both ***individual exposures*** and ***collective exposures***. Individual exposures are those to individuals and attention is given primarily to the most (potentially) exposed group of individuals, referred to as the ***critical group***. Collective exposures are individual doses integrated over the entire exposed population and are indicative of the overall detriment to society posed by radiation exposures from specific sources and practices.

The basic provisions of the 'System of Radiological Protection' in relation to proposed and continuing ***practices*** are termed ***justification***, ***compliance with exposure limits, and optimization***. Simply stated, in the context of practices relevant to this assessment, these are:

- **Justification:** Practices involving the production or use of radionuclides should be justified as offering net benefit to society before being authorized.
- **Compliance with Exposure Limits:** Limits of radiation exposure to individuals (both radiation workers and members of the public) set to avoid deterministic and significant stochastic effects must not be exceeded.
- **Optimization of Protection:** All practices should be optimized to reduce radiation exposures to values as low as reasonably achievable, social and economic factors taken into account.

Individual doses, dose limits and dose constraints

For individual exposures, primary concern relates to ensuring the protection of the most exposed individuals, namely members of the critical group. It is to members of this group that the dose limit for members of the public is applied. Prior assessments of practices and sources are directed at ensuring that doses to members of the critical group from all relevant practices are below this limit – currently 1 mSv/y for members of the public. However, when dealing with individual practices, only a fraction of the dose limit can be used. This is called the 'dose constraint'. Dose constraints are designed to ensure that aggregate exposures from all sources and practices to individuals do not exceed the dose limit for members of the public. Dose calculations must take

account of all pathways of exposure and all radionuclides, however, in reality, a few of these will be dominant, and conservative (pessimistic) consideration of these ensures that the dose limit/dose constraint is not exceeded.

Collective doses

Collective dose is the integral of dose within a population. The primary application of collective dose is in relation to optimization. Optimization focuses on minimizing the collective dose as a proxy for the overall health (radiological) detriment. This requires that estimation of collective dose be as realistic as possible, in contrast to individual dose predictions that can be conservative to ensure compliance with dose limits and the relevant dose constraints.

Exclusion and exemption

Any radiation exposure that is essentially beyond human control, such as the dose from ^{40}K in the body, is *excluded* from radiological control. Furthermore, practices, and sources within a practice, may be *exempted* if the associated individual risks are negligible and the collective radiological impact does not warrant regulatory concern. However, exemption is also subject to the practice or source being inherently safe in the sense of there being no significant likelihood of circumstances (i.e., accidents) in which the operational dose estimates would be exceeded.

8.2.3.2. **Intervention**

In some situations, the sources, pathways, and exposed individuals already exist when the decisions about control measures are being considered. Sometimes the new control measures can be defined as part of a review of the original practice, but, more commonly, they will constitute **interventions**. An important group of such situations involves enhanced exposures to natural sources of radiation. Accidents and emergencies will have been considered as sources of potential exposure when dealing with practices, but if they occur, they may call for intervention. In most situations, intervention cannot be applied at the source and has to be applied in the environment and/or to the freedom of action of individuals. The countermeasures forming a program of intervention, which always have some disadvantages, should be justified in the sense that they should do more good than harm. Their form, scale and duration should be optimized to obtain the maximum benefit. Dose limits are only applicable to practices. The use of dose limits established for the control of practices, or any other predetermined limits, as a basis for deciding on intervention might involve measures that would be out of all proportion to the benefit obtained and would therefore conflict with the principle of justification. Nevertheless, at some level of dose, approaching that which would cause serious deterministic effects, some kind of intervention will become virtually mandatory.

In judging the benefits and detriments of intervention aimed at reducing public exposure, the comparison should, in the first place, be made for those at risk, but there will also be an impact on the rest of society and the judgements will have to be wide enough to also cover these impacts. The application of intervention is to avert future doses. The dose potentially averted by the implementation of intervention measures is referred to as the ***avertable dose***.

8.2.3.3. **Radiological assessments**

Internationally agreed standards for radiological protection require that no **practice** involving ionizing radiation shall be adopted unless it accords with the basic principles of the 'System of Radiological Protection', namely: the practice is justified; protection is optimized; and there is compliance with the individual dose/risk limits.

A licence is normally required to operate a major installation. There are, however, other forms of control depending of the type of the practice. For major installations, justification of the practice normally goes beyond the radiological protection regime. Radiation safety aspects are only one consideration, although an important one, in justification. In order to get a licence for a practice, the applicant needs to make an assessment of the nature, magnitude and likelihood of the exposures attributable to the practice/source and to show that all reasonable measures for the protection and safety of both workers and the public have been taken. The following discussion is limited to the environmental aspects of radiological assessments and does not address either safety assessments or assessments of worker doses and worker safety.

The basic principles of radiological protection are in essence followed in all countries for civilian applications of ionizing radiation. However, the practical requirements for predictive assessments necessary for the issuing of a licence may vary from country to country. Likewise, the requirements for retrospective assessments may also vary among national jurisdictions.

The most common elements of assessments relating to limiting and minimizing the environmental consequences of a practice involving the production, use and release of radionuclides may be grouped as follows:

The applicant/licensee must carry out:

- **A source-related assessment, prior to licensing,** covering both normal operations and accident scenarios, providing input to the justification process and demonstrating compliance with the requirements for optimisation of protection including the relevant individual dose constraints. The assessment of collective dose is used to select the optimum options for protection.
- **Source-related assessments and monitoring during operation** to confirm the validity of the 'prior to licence' assessment; in other words, to confirm that the conditions are within those specified in the assessment and licence.
- **Dose/consequence assessments in the event of an accident** to predict the consequences and to select necessary and appropriate actions.

The licensing authority carries out the following types of assessments, independent of the applicant/licensee:

- Assessments to confirm the applicant's assessments for items 1 and 2 above.
- **Individual-related assessments** to check that the integrated dose contributions from all relevant sources/practices do not exceed the dose limits for individuals.
- **Source-related assessments**, in the event of an accident, either domestic or foreign, and, where appropriate, for chronic exposure situations, for predicting consequences.

In other words, the licensing authority assesses individual doses for critical population groups to determine whether any individual doses approach thresholds for deterministic effects and whether any individual has an excessively high probability of suffering stochastic effects. For intervention, the assessment of avertable doses by each protective action is required to justify and optimize the protective action. Thus, the avertable average individual and collective doses of the affected population need to be assessed.

The basic structure of source-related assessment, mostly predictive in nature, is similar irrespective of the application

of the assessment. Its purpose is to establish the relationship between the source (release rate) and the consequences to humans and biota (expressed in terms of dose rates). It should be noted that it has been shown that fauna will not be adversely affected at the population level provided that the system for limiting the exposure to humans is applied. This conclusion is conditional on the proximity of the exposed humans and fauna relative to the source. In cases where the exposed humans and fauna are at similar distances from a source, the conclusion is valid. However, where fauna are situated relatively close to a source compared to humans, such as in the case of sources situated on the deep remote ocean floor, faunal and human doses are so dissimilar as to require specific and independent consideration of the dose to fauna (IAEA 1988, 1992).

First, it needs to be shown that the releases will not cause higher individual doses than the dose constraints assigned by the relevant national authority. Second, it needs to be demonstrated that the doses have been reduced by appropriate measures to a level below which it is no longer reasonable to make further reductions taking into account social and economic factors (in other words that the protection is optimized).

In the optimization process, the alternative technical protective measures can be compared with each other in their ability to reduce the collective dose in relation to the resources spent, to identify the option of optimal protection. Social factors can play an important role in the optimization process, such as in the case of selecting intervention measures. However, there are many decision-aiding techniques that can be used to take social factors, which are often difficult to quantify, into account, and these are not discussed further here.

Individual-related assessments carried out by authorities concerning the public are basically retrospective assessments. These enable the authorities to ensure that: individuals do not receive doses exceeding the dose limits from licensed practices because of unanticipated overlapping of critical groups, and, following an accident or in a chronic exposure situation, individuals do not incur doses that would call for consideration of protective actions.

Such assessments are based on measurements of activity concentrations in environmental materials that can contribute to the internal and external doses to members of the public or, in some cases, on direct activity concentration measurements on humans.

If the individual exposures are excessive, intervention should be considered. An evaluation of the individual and collective doses avertable by potential intervention measures is required for justification purposes. Estimates of avertable dose should be as realistic as possible to avoid overestimation of the potential benefits of protective actions.

8.2.3.4. **The basis for intervention**

The measures needed to restrict the exposure of individuals, either in the control of a practice or by intervention, can be taken by applying action at any point in the paths linking the source to the individuals. The action may be applied to the source, to the environment, or to the individual, e.g. moving people, or personal protective measures. Actions that can be applied at the source will be the least disruptive. They can be made as effective as required, unless they fail as a result of an accident or for other reasons. For example, such could be the case if disposed waste is removed from one part of the environment to another without careful assessment of the consequences. Action at the source influences all the pathways and individuals associated with that source. Thus, whenever possible, controls applied at the source are to be preferred. Actions applied to the environment, or to individuals, are more obtrusive and may have social disadvantages, not all of which are foreseeable. Their effectiveness will be limited because they apply only to some of the pathways and individuals.

It is essential to avoid confusion between **'Dose Limits and Constraints'** restricting releases from normal operations and the **'Intervention Levels or Action Levels'** for chronic exposure situations or accidents, that trigger intervention or action. Although similar principles apply to normal operations of practices and to intervention in post-accident or chronic exposure situations (i.e., justification and optimization), they are applied to different quantities. For the control of planned releases, the benefit from the source itself is compared with the additional radiation exposures it produces. In the case of intervention, the disadvantages of the intervention are compared with the reduction in total radiation exposure (irrespective of the origin of the exposure) achievable. Intervention levels, based on the justification and optimization principles, are either generic or specific, and are primarily expressed in terms of avertable dose in Sv (typically as mSv) but can also be expressed in directly measurable quantities, as dose rates or activity concentrations.

For accident situations a set of 'Generic Intervention Levels' has been derived and internationally recommended. They are given in terms of avertable dose achieved by major protective actions applicable in case of a nuclear accident. Also, 'Action Levels' (or interdiction levels) for food were recommended by the FAO-WHO Codex Alimentarius Commission and the International Atomic Energy Agency (IAEA) (Table 8·2). They can be used if there is no shortage of food and there are no other compelling social or economic factors. They were originally developed for the international trade in food contaminated with radionuclides but are also applied to food as consumed.

Table 8·2. Generic action levels for foodstuffs.

Radionuclide	Foods destined for general consumption, Bq/kg	Milk, infant foods and drinking water, Bq/kg
^{134}Cs, ^{137}Cs, ^{103}Ru, ^{106}Ru, ^{89}Sr	1000	1000
^{131}I	–	100
^{90}Sr	100	–
^{241}Am, ^{238}Pu, ^{239}Pu	10	1

Otherwise no international consensus on Action Levels in chronic exposure situations yet exists, except for radon in dwellings. Thus, the basis for intervention is to justify and optimize the available options for protective actions in chronic exposure situations on a case-by-case basis. However, work is underway in both the International Commission on Radiological Protection (ICRP) and the IAEA to reach a consensus on these matters.

8.2.3.5. **Other issues relevant to radiological assessment**

8.2.3.5.1. **Relationship between radiation exposure and risk of adverse health effects**

Health protection from radiological exposures at low doses (stochastic effects regime) is based on an important *a priori* assumption – that the risk of adverse health effects increases in direct proportion to radiation exposure **without threshold**. This permits extrapolation of the dose-response relationship into low dose regimes from that at higher dose where the relationship can be epidemiologically or experimentally determined. There is an established relationship between

probability of serious health defects (fatal cancer induction) and dose, of 0.05/Sv averaged over the population. This means that a dose of 1 mSv corresponds to an increased risk of serious health defect of 5×10^{-5}.

There is an interesting consequence of the basic assumption of linear no-threshold dose-response in the low dose stochastic regime. If, for example, a practice results in a large population of people suffering increased radiation exposure and the integrated (collective) dose in this population is 100 manSv then the expected number of serious additional health defects in the population is 5 irrespective of the size of the exposed population. Obviously, the smaller the population over which the exposure is distributed, the more seriously this collective (health) detriment might be regarded because of the increased individual doses.

The concept of ***detriment*** is used as a measure of the total harm that would eventually be experienced by an exposed group and its descendants as a result of the group's exposure to radiation. Health detriment is part of the total detriment, however, in practice, in radiological protection, the term is used solely in relation to health detriment. In optimization studies, special allowance needs to be made for other forms of detriment, as appropriate.

8.2.3.5.2. Transport processes and exposure pathways

Radiation exposure can be grouped into two main types: external and internal. External exposures are those resulting from sources outside the person or organism. Internal exposures (comprising inhalation and ingestion) are those resulting from the incorporation of radionuclides into an organism. There are a wide range of pathways, summarized below, that can lead to exposures of organisms. In constructing assessments of prior exposure, the objective is to ensure that all potential pathways of exposure are considered, although in many cases there will be one or two exposure routes that will be dominant – these are referred to as ***critical pathways***. In Arctic ecosystems, certain critical pathways are particularly important for this assessment. These pathways are discussed here in order to provide background information for discussions in subsequent sections. It is important that appropriate models are available, and that they adequately describe the transport of radionuclides in the environment. The types of models and their uses are also considered later in this section.

External exposure arises from radionuclides deposited onto many different surfaces. The dose varies with the radionuclide deposited, with different exposures occurring for various alpha-, beta- and gamma-emitting radionuclides. In addition, the dose will change with time, as radionuclides migrate down soil profiles or are weathered from plant surfaces, particularly in forests or urban areas where significant interception of radionuclides can occur above the ground.

Internal exposure occurs largely through both inhalation and ingestion. Inhalation exposure occurs when radionuclides are breathed into the lung with air and can either impart direct exposures to the lung or be retained in lung tissue and possibly absorbed into the plasma. Ingestion exposure can arise through drinking or eating contaminated foodstuffs and can therefore result from a large number of different exposure pathways following releases to the atmospheric and marine/aquatic environments. The most important factors which lead to variation in rates of transfer via these pathways differ for each radionuclide and, hence, the environmental mobility of different radionuclides also varies considerably. This assessment focuses primarily on radiocaesium, radiostrontium, plutonium radioisotopes and, to a lesser extent, radioiodine, because of their source strengths, mobilities or radiotoxicities.

Atmospheric transport

Radionuclides can be released from a wide variety of different sources and can be ejected into a variety of atmospheric layers under different conditions. Weather conditions at the time of atmospheric release will generally determine the extent of atmospheric dispersion. The mean residence time of radionuclides in the Arctic stratosphere is in the order of one year. The transfer of radionuclides from the stratosphere to the troposphere occurs preferentially in the spring, when the tropopause (the interface between them) is most 'permeable' (Brewer 1949, Dobson 1956). The mean residence time of radionuclides in the troposphere is only a few weeks. Radionuclides in the troposphere are transferred to the surface of the Earth as wet or dry fallout.

Radionuclides have been introduced into the Arctic atmosphere from nuclear weapons testing and from accidental (e.g., Chernobyl) or routine (e.g., the Kola nuclear power plant) releases from nuclear facilities. The testing of thermonuclear weapons (in the Megatons TNT equivalent range) in the atmosphere usually injected most of the radionuclide yield into the stratosphere. Venting from underground nuclear explosions and releases from reactor accidents, atmospheric tests of fission weapons (in the kilotons TNT range) mainly entered the troposphere.

Marine transport

Releases into Arctic marine ecosystems can either occur directly, through routine releases from nuclear reactors into cooling water streams, leakage from dumped solid wastes, direct dumping of liquid wastes, or indirectly via atmospheric deposition. In addition, radionuclides released elsewhere may be transported into Arctic marine systems. Typical examples of the latter include the releases from Sellafield and Cap de La Hague reprocessing plants. Furthermore, releases into freshwater, either directly or via catchment contamination, may eventually be transported into the Arctic marine environment via river systems. Waterborne discharges have occurred from a variety of different Russian nuclear establishments to the Ob and Yenisey river systems. This has undoubtedly resulted in the transport of some mobile radionuclides (e.g., ^{90}Sr) through aquatic pathways to Arctic marine ecosystems, but it is presently difficult to quantify the amounts of radionuclides transported in this way.

Terrestrial transport

Once radionuclides are deposited onto the Earth's surface, their subsequent behavior is dependent on a number of factors including their physico-chemical form and the type of environment into which they have been released. Terrestrial and freshwater environments generally receive most of their radioactive contamination through precipitation (wet fallout). Vegetation may be contaminated directly by deposition of the radionuclides onto the surface of the plants, or indirectly by uptake from the soil through the roots. Further transfer of radionuclides in the food chain occurs when animals, including humans, consume food, drink water or breath air. A common example, with which most people are familiar, is the grass → cow → milk → man pathway, whereby grassland is contaminated through atmospheric fallout and the contamination is transferred to humans through the consumption of grass by cattle, and the subsequent production and consumption of milk. Certain processes are of central importance in determining the rates of transfer and these are summarized below with particular reference to Arctic ecosystems:

Interception
The rate of interception of aerially-deposited radionuclides varies with surface characteristics, meteorological conditions and the ratios of surface area to biomass, and is particularly high for many tree species, lichens and mosses. In addition, rates of interception vary seasonally, particularly for annual crops. Intercepted fallout is gradually lost from the intercepting surfaces by a variety of processes, collectively termed 'weathering'. The initial rates of interception and subsequent rates of weathering are important factors in agricultural systems because they determine the degree of external contamination of crops and pasture grasses in the initial phase after, for example, an accident. In Arctic food chains, the ability of lichen to intercept, absorb and retain most of the deposited radiocaesium is particularly important because of the utilization of lichen as a winter foodstuff for reindeer.

Soil-to-plant transfer
In temperate areas, the variation in the rate of soil-to-plant transfer of radionuclides is one of the most important factors influencing the extent of food contamination for both agricultural and semi-natural products. In Arctic areas, the comparative importance of this exposure route for agricultural products is potentially much lower than in temperate areas because fewer agricultural plant products are grown.

Plants obtain nutrients and radionuclide contaminants from the soil solution. Thus, the rate of uptake from soil by plants is determined by the rate at which the plant roots absorb different elements or compounds and the activity concentrations of radionuclides in the soil solution. If a radionuclide has a close chemical analogue, the rate of transfer of the radionuclide will be heavily dependent on its interaction with the analogue, particularly any competitive effects. When radionuclides are deposited onto the soil they are chemically bound by different soil constituents and it is the relative strength of these associations that determines the activity concentration of the radionuclide in the soil solution.

Many radionuclides are either taken up by plant roots at very low rates, or form strong bonds with various soil constituents. Therefore, the rate of plant uptake of many radionuclides is low compared with nutrient ions. The main exceptions are radiostrontium, which has significant rates of uptake from many different soil types, and radiocaesium, which is absorbed by plant roots much more readily from organic soils and, to a lesser extent sandy soils, than from more mineralized soils with a higher content of clay minerals which strongly bind radiocaesium.

In addition to the soil-based factors, there are marked differences in the capacity of different plant species to absorb radionuclides. However, these differences are usually smaller than those determined by the soil type.

For radiocaesium, a further important exposure route from the soil involves uptake by fungal hyphae. Many soils contain large amounts of fungal hyphae that have a pronounced ability to absorb radiocaesium from the soil. When the fruiting bodies (e.g., mushrooms) appear they often contain much higher radiocaesium activity concentrations than most other food products. The extent of radiocaesium contamination of fruit bodies is highly variable, both within and among fungal species.

Plant-to-animal transfer
Animal products form an important part of the diet of many Arctic peoples. Whilst some animal products are similar to those of temperate regions, such as milk, pork and lamb, a wide range of game animals and, of course, both semi-domesticated and wild reindeer are also heavily utilized in the Arctic. The rate of contamination of food products from these animals depends on three major factors:

- Diet selection.
  In temperate regions, diet selection by food producing agricultural animals is comparatively unimportant as the range of herbage available is highly regulated and often comprises only a few major herbage sources. In contrast, animals in semi-natural ecosystems ingest a wide range of different plants and fungi at different times of the year. This leads to considerable seasonal variation in the amounts of radionuclide ingested by different species.

  A classic Arctic example is the consumption of lichen by reindeer in winter which leads to substantially higher radiocaesium activity concentrations in reindeer meat during the winter period. In addition, radiocaesium contamination of game, such as roe deer and moose often substantially increases in autumn due to the consumption of highly contaminated fungi.
- Availability for absorption in the gut.
  Radionuclides are absorbed to different extents in animal guts. After ingestion of contaminated vegetation the three most available radionuclides, in order of decreasing fractions of gut absorption are: radioiodine (100%) > radiocaesium (80%) > radiostrontium (ca. 20%). Most other radionuclides, including plutonium, are absorbed in the gut in fractions of less than 1%.
- Metabolism of the radionuclide.
  Once radionuclides have been absorbed through the gut wall they are distributed within animal tissues. The tissues in which they accumulate and the subsequent rates of loss, via urine and faeces, vary. The most important radionuclides are those which contaminate parts of the animal which are eaten by humans, namely meat, offal and milk. Again, the most important radionuclides are radioiodine, radiocaesium and radiostrontium, all of which are readily transferred to milk. In addition, radiocaesium contaminates all soft tissues, and therefore ingestion of radiocaesium via meat is also important. The effective biological half-lives of these radionuclides vary, but the rates of radioiodine uptake and loss are generally faster than those of radiocaesium and radiostrontium. Changes in radionuclide activity concentrations in ingested food will be reflected in milk and meat within a few days. Target organs for the other different radionuclides vary, but notably include bone and offal.

The effective biological half-life of a radionuclide in an organism is a function of both the biological half-life of the element in the organism and the physical half-life of the radionuclide.

$$1/T_{1/2\ \text{eff-biol}} = 1/T_{1/2\ \text{biol}} + 1/T_{1/2\ \text{phy}}$$

The effective ecological half-life of a radionuclide is a function of both the half-life of the element in a component of an ecosystem and the physical half-life of the radionuclide.

$$1/T_{1/2\ \text{eff-eco}} = 1/T_{1/2\ \text{eco}} + 1/T_{1/2\ \text{phy}}$$

Freshwater pathways
Freshwater systems, such as lakes, rivers and groundwater, may also be contaminated by atmospheric deposition of radionuclides or direct releases into rivers. The transfer of radionuclides from such systems occurs mainly through consumption of freshwater fish and from exploitation as drinking water. The mobility of a radionuclide depends on its ability to bind to river sediments and its competitive interac-

tions with other ions. Strontium is one of the more mobile elements in aquatic systems because it does not bind strongly to sedimentary material.

Marine pathways

Exposure from marine pathways arises from the consumption of marine food products, including fish and shellfish, mammals such as seals and whales, and seaweed. In general, contamination of marine biota is much less than that arising from terrestrial pathways, largely because of the strong sorption of many radionuclides by aquatic sediments and also because of the enormous dilution which occurs in marine water bodies.

8.2.4. Modeling

Measurements and models necessary for radiological assessments can be grouped as follows:

- In cases where the releases of radionuclides from the source are not known or not measured, scenarios and models for existing, projected or potential release rates have to be developed (e.g., releases from reactor accidents, *Komsomolets* submarine, dumped packaged material, dumped reactors, contaminated marine sediments, etc.).
- Environmental pathways from the source to humans have to be identified and the transport of radionuclides modeled. In box models, the pathways of radionuclides from the release point to humans are described by transfers among trophic levels in the environment, such as the transfer of airborne radionuclides from 'air' to 'pasture' to 'milk'. The radionuclide transfer between compartments is commonly described by transfer parameters. In simple models, these transfer parameters represent the ratio of concentrations of a radionuclide in two compartments under equilibrium conditions. Changes with time are described using appropriate rate functions for decay or removal. In more complex models, an attempt is made to represent the time-dependent movement of radionuclides among the various environmental compartments. These time-dependent models are referred to as 'dynamic' models.

Simple equilibrium models have been well described and documented in the literature and many of the transfer parameters have become virtually 'standardized'. In contrast, the parameters used in dynamic models tend to be both model and situation specific and their values depend, amongst other things, on the assumptions made in establishing the model. Equilibrium box models are often sufficiently robust and reliable for radiological protection purposes.

Most predictive models, both dynamic and equilibrium, have been developed to describe temperate conditions and assume that external exposure arises predominantly from soil, forest or urban surfaces and that internal exposure arises from inhalation and ingestion of contaminated foodstuffs produced as a result of normal agricultural or fishing activities. It is, therefore, important to consider whether existing models can be adapted to Arctic areas. For marine exposure, the transport of radionuclides in ice is an additional pathway which needs to be incorporated into an assessment. The main difference with regard to external exposure is the presence of snow and the lack of forested areas, which may change exposure rates. This can be compensated for within current models using appropriate factors. For estimating internal exposure, the main difference lies in the origin of the foodstuffs consumed in the Arctic. Agricultural production is limited in the Arctic, and most foods are either harvested from semi-natural or marine ecosystems or imported. Contamination routes in semi-natural terrestrial ecosystems can usually be represented within existing model structures, with minor changes necessary to allow, for example, for surface contamination of lichen, rather than root uptake, as a long-term exposure route via reindeer. However, transfer values used in such models need to be appropriately quantified to describe adequately the movement of radionuclides between environmental compartments in the Arctic. This cannot be accomplished for Arctic pathways with the same degree of confidence as is possible for temperate conditions because there is much greater individual and seasonal variation in rates of transfer, associated, for example, with diet selection by animals. Thus, parameterization is a more significant problem in the application of models to the Arctic because of the relatively greater importance of semi-natural ecosystems.

To parameterize transfer in Arctic areas, it is often necessary to use both individual and aggregated transfer parameters. Individual parameters describe the direct transfer from one environmental compartment to another. An example is the transfer from feed to cattle. Aggregated parameters describe the transfer via a complete chain of compartments for which each compartmental transfer has an individual transfer rate. An example is a transfer parameter which relates a radionuclide activity concentration in soil to that in the meat of grazing animals. The diet of almost all humans comprises food derived from diverse locations in the world. It is, therefore, not always possible to directly correlate the radionuclide concentration in food with the contamination of the area where humans are living. However, for some foodstuffs this is a less serious problem in the Arctic than for most other areas.

Radiation dose (dose equivalent from external exposure plus committed effective dose from intakes of radioactive substances) is the relevant quantity in radiological protection for assessing health consequences. The procedure for calculating doses to humans are based on different approaches according to the nature of the pathway. For external exposures, the dose to individuals from radionuclides in air, water or ground surfaces is obtained by applying the appropriate dosimetric models and taking into account shielding effects, annual rates of occupancy and any other factors characterizing the behavior of the individuals. For internal exposures, doses are calculated using metabolic models that incorporate inhalation, food ingestion and gut absorption rates. Such models have been internationally established and provide dose-intake factors for different radionuclides.

There are several ways to assess and predict individual dose commitments. In short, the following approaches are used in this assessment:

Individual dose commitments:

- Annual intakes of ^{137}Cs and ^{90}Sr via food and drinking water have been assessed and integrated over the time period from 1950 to ∞. The results have been multiplied by relevant dose/intake factors to obtain the internal dose contributions from these radionuclides via consumption. Doses via the inhalation pathway have been calculated in the same way, as appropriate. Contributions from other radionuclides and external doses are based on ratios derived from UNSCEAR (1993).
- The internal dose from ^{137}Cs can also be assessed by using wholebody measurement. The body burden, expressed in Bq/kg, is then multiplied by a dose factor given by UNSCEAR to obtain the dose commitment.
- UNSCEAR applies a compartment model (Figure 8·1) to assess the dose commitment from releases of radioactive substances to the atmosphere from atmospheric nuclear

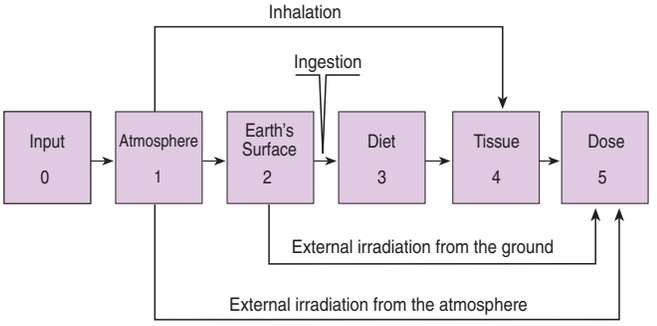


Figure 8·1. Compartment model used to assess doses from releases of radioactive materials to the atmosphere from nuclear testing (UNSCEAR 1982).

weapons testing. The dose commitment for a specific radionuclide D_c, due to an environmental input A_0 into the atmosphere is given by:

$$D_c = P_{01}(P_{12}P_{23}P_{34}P_{45} + P_{14}P_{45} + P_{15} + P_{12}P_{25})A_0$$

where P_{xy} is the transfer coefficient from box x to box y. P_{01} is the integrated concentration of a radionuclide in air at a specific location (or averaged for a broader region), divided by the amount released. The four terms in the parentheses account for the ingestion, inhalation and external exposure (cloud gamma and ground gamma, respectively) pathways. The values of the various transfer coefficients adopted by UNSCEAR have, to a large extent, been derived from observations made in northern temperate latitudes. This assessment has considered the validity of this approach for Arctic areas.

Integrated transfer factors

The transfer of radionuclides from deposition to diet has been calculated by UNSCEAR (1993) according to the model:

$$C_i = b_1F_i + b_2F_{i-1} + b_3\sum_{n=1}^{\infty} e^{-\lambda n}F_{i-n}$$

where C_i is the activity concentration of the radionuclide in a food component in the year i due to the deposition density rate in the year i, F_i, in the previous year, F_{i-1}, and in all previous years reduced by exponential decay. This decay, with decay constant λ, reflects both radioactive decay and environmental loss of the radionuclide, i.e., it corresponds to the effective decay constant $\lambda = \ln 2/T_{eff}$, where T_{eff} is the effective ecological half-life (see section 8.2.3.5.2). The coefficients b_1, b_2, b_3 and the parameter λ are determined by regression analysis of measured deposition and diet data. The so-called transfer coefficient P_{23} (see example in section 8.4) from deposition to diet is defined as:

$$P_{23} = b_1 + b_2 + b_3\, e^{-\lambda n} / (1 - e^{-\lambda n})$$

with the units of Bq y/kg per kBq/m^2. P_{23} is the infinite time-integral of the activity concentrations of a radionuclide in a product (e.g., milk) arising from the deposition of 1 kBq/m^2 of the radionuclide.

The above UNSCEAR model has been applied successfully to nuclear weapons fallout on agricultural ecosystems in temperate regions, but may not be readily applicable to Arctic ecosystems where the pathways of radionuclide accumulation in a given product may vary locally as well as temporally. In particular, the UNSCEAR model omits the lichen-reindeer pathway, which is important in the Arctic, and is limited in its ability to take account of temporal variability in production and/or harvesting which can vary considerably in semi-natural systems. For example, some terrestrial mammals may consume large amounts of mushrooms in years in which there are many mushrooms produced, and consume essentially no mushrooms in poor years. This could easily change the annual body concentrations of radiocaesium significantly in the absence of any similar changes in the annual depositions, which are used for the annual predictions in the UNSCEAR model. Similarly, for example, in a given year reindeer could move to an area where lichen is less abundant, which would reduce radiocaesium body levels in that year. Such effects would not be predictable from the UNSCEAR model because changes in food consumption patterns are not taken into account. However, if enough years of observations of environmental contamination levels in the Arctic are available, it is possible to apply the UNSCEAR model for the calculation of the transfer coefficient P_{23} from deposition to diet, because the temporal and local variations are effectively smoothed out.

Integrated transfer factors provide a comprehensive assessment of transfer over the long term. However, for many products, especially in the Arctic, the data is too limited to calculate these factors. As an alternate approach, the use of aggregated transfer coefficients can be considered.

Aggregated transfer coefficients (T_{ag}s)

In the event of a radioactive release, there is often an urgent need for information on transfer to food products. In agricultural production systems, soil and vegetation properties are typically fairly homogeneous and models have relied on such parameters to define the concentration ratio for soil–plant transfer and the transfer coefficient for feed–animal transfer. In contrast, semi-natural ecosystems are inherently more variable in both soil and vegetation characteristics, and animals graze much more selectively. Consequently, assessment of transfer using established parameters may not adequately account for all the pathways that contribute to an activity concentration in a given foodstuff.

The aggregated transfer coefficient (T_{ag}) has, therefore, been developed as a simple means of quantifying transfer between different environmental compartments in semi-natural ecosystems after an accident (see review by Howard *et al.* 1996). By comparing activity concentrations in a food product with representative measurements of total deposition, on an areal basis, aggregated values of transfer can be obtained that can be easily applied in radiological assessment models. For the current assessment, three different approaches have been identified: 1) comparison of foodstuff activity concentration with total deposition, to give an estimate of transfer from all sources, 2) transfer from an individual release, Chernobyl fallout, which has been quantified by studying changes in the ratio between ^{134}Cs and ^{137}Cs, and 3) T_{ag} values have been calculated from the organic layer of the soil to mushrooms. The second and third options may be optimal parameters for modeling in Arctic ecosystems following an accident and have been used preferentially in this assessment where available.

Provided specific characteristics of an ecosystem are known, T_{ag} values offer a convenient means of assessing transfer between most environmental compartments. However, like all transfer parameters they must be combined with estimates of time-dependent changes in transfer, expressed as effective ecological ($T_{1/2\ eff\text{-}eco}$) half-lives. This is particularly important in semi-natural ecosystems where contamination can persist for many years. Furthermore, immediately after a release T_{ag} values are affected by the interception of fallout radionuclides by vegetation cover (this is further affected by season). Therefore, T_{ag} values are of limited value for quantifying transfer when deposition is continuous over a long period. The rate of transfer of intercepted deposition to the soil surface may also be subject to seasonal

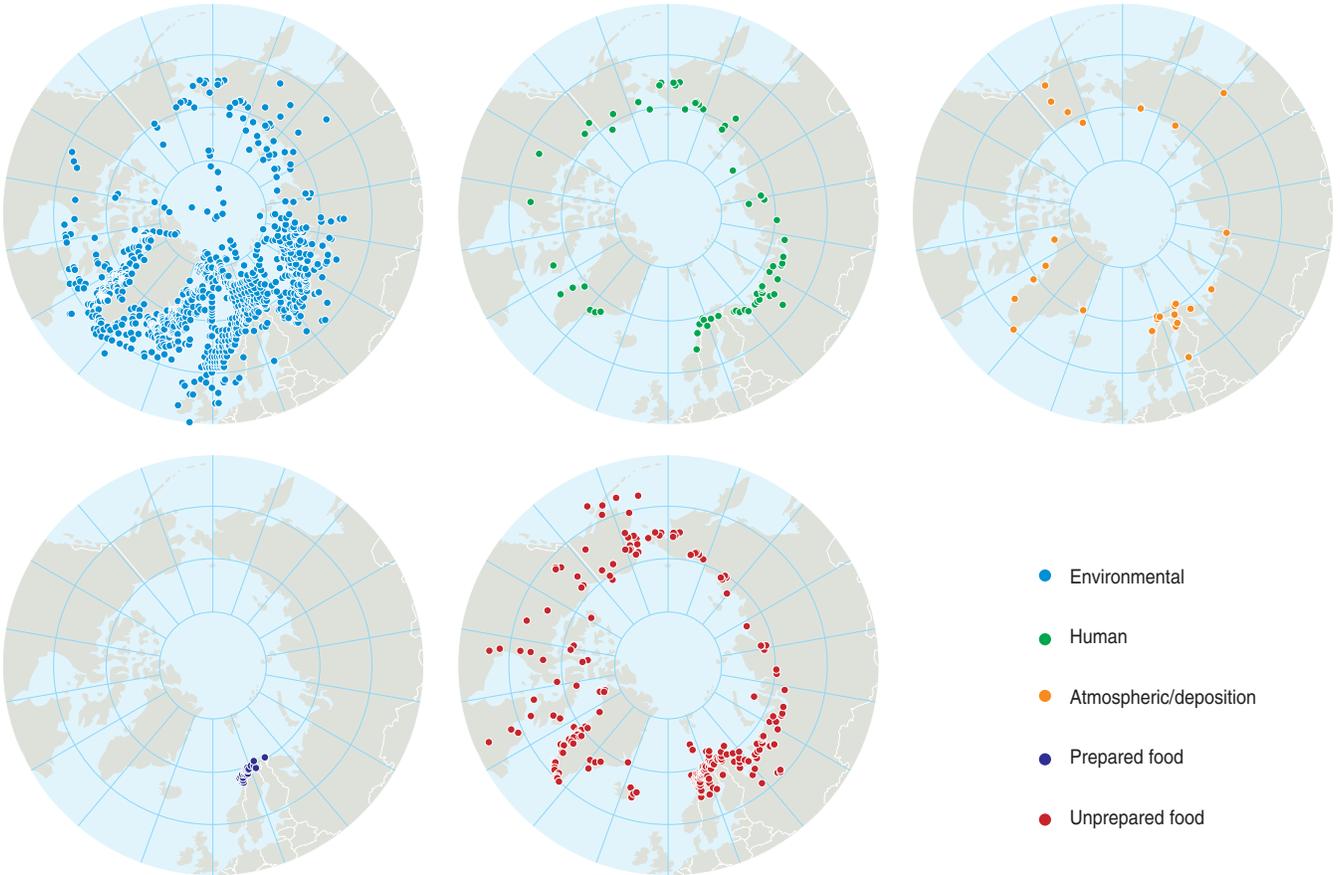


Figure 8·2. Geographical distribution of sample information in the AMAP radioactivity database.

influences. In contrast, lichens act as a surface trap for deposition, and therefore T_{ag} values can be determined and applied immediately for transfer from lichen to reindeer.

8.2.5. The AMAP assessment

Prior assessment of proposed practices is a common requirement imposed by legislative jurisdictions in both nuclear and non-nuclear fields. Surveillance and retrospective assessments are also employed to ensure that practices adopted by society have consequences (detriments and benefits) that are consistent with those outlined in the prior assessment. The latter can be specific to particular practices or more general, such as assessments of conditions in a specific regional area like the AMAP Assessment.

The sections of this document dealing with radiological assessments comprise three parts:

- Individual-related dose assessments for individuals in average Arctic populations and to individuals within selected real or hypothetical population groups within the Arctic considered to have comparatively high radionuclide intakes. These assessments are based on observed activity concentrations in environmental media, dietary intakes and dose-intake factors and, wherever relevant, take account of occupancy times, shielding factors and dosimetric models (section 8.4).
- Source-related assessments of **present and future doses** to Arctic populations and selected population groups **from operational releases** from nuclear power generating plants, other civilian and military reactors, nuclear fuel reprocessing installations, mining activities, nuclear explosions, both military naval and civilian, and **previous accidental releases** of radionuclides to the environment (section 8.5).
- Source-related assessments of **future releases** as a result of potential nuclear accidents or from contained sources in the environment such as objects containing radioactive wastes dumped at sea. These assessments are based on estimates of the inventories and release rates of radionuclides from such contained sources and modeling of the subsequent environmental transport and human exposure pathways (section 8.6).

It should be noted that some of the sources considered under item 2 above (i.e., in section 8.5) are both current and potential sources of releases to the environment. In such cases, for example that of the sunken *Komsomolets* submarine, the relevant source-related assessment is presented in section 8.5. rather than 8.6.

8.3. Past and present radioactive contamination of the Arctic

During the AMAP assessment period, data about the past and present contamination have been collected mainly through the AMAP radioactivity data center which, together with the AMAP radioactivity assessment group, has compiled data to describe levels and trends in the Arctic area. The sources of the data include results from AMAP monitoring programs, national reports compiled for the assessment group, and the open literature. In total, measurements of more than 20 000 samples have been reported. There are, of course, gaps in both temporal and spatial information. However, considerable information on radioactive contamination in the Arctic is available. An overview of the geographical distribution of the samples in the database is given in Figure 8·2. Full references to all data are available in the database.

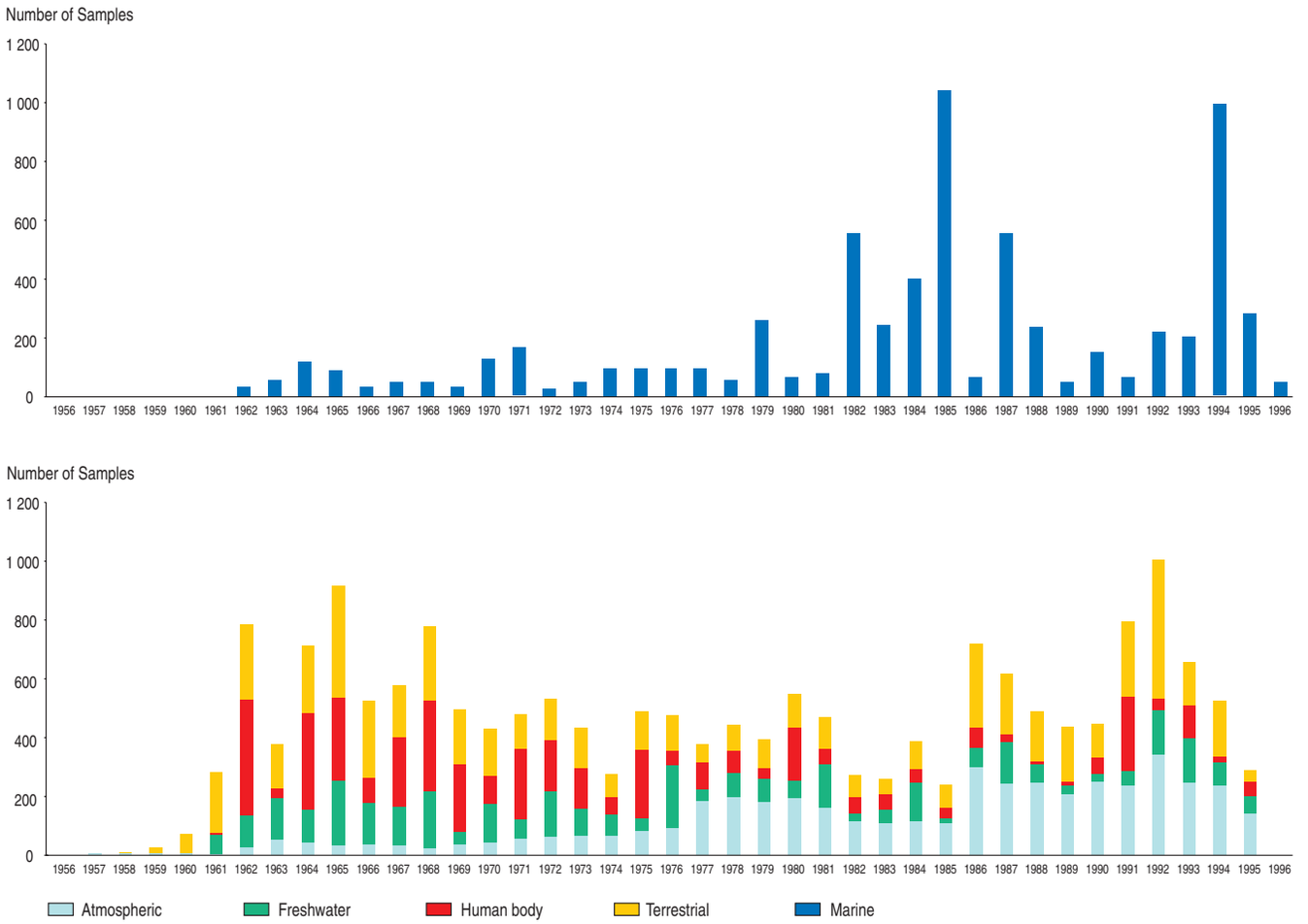


Figure 8·3. Available data in the AMAP radioactivity database as a function of time.

The gaps in the data are particularly notable for natural food products, such as mushrooms, wild animals (except reindeer) and freshwater fish (for some areas). In addition, there is a lack of data prior to 1960 for Russian rivers (especially the Ob) which most probably contained radionuclides in the 1950s as a result of releases from the Mayak reprocessing plants to the Ob river system. Another notable lack of data relates to radionuclides in sea ice, for which there is some information from recent years, but few data prior to the 1990s.

As a result of atmospheric nuclear weapons testing, most countries started monitoring of radionuclides in various samples in the late 1950s or early 1960s. Regrettably, many of the terrestrial monitoring programs were terminated in the late 1960s or early 1970s due to the decreased fallout following the atmospheric nuclear test ban in 1963. Nevertheless, a few time series were continued to the present and give temporal information on radionuclides in the Arctic environment.

Marine sampling increased in the late 1970s and early 1980s. This is probably due to interest in the increased discharges of radionuclides, especially ^{137}Cs, from the British Nuclear Fuel's reprocessing plant in Sellafield, UK, and other European reprocessing plants. At the end of the 1980s, numerous new monitoring programs for radionuclide were initiated as a response to the Chernobyl nuclear power plant accident and to the information released by the Russian Federation about dumping of nuclear waste in the Kara and Barents Seas by the former Soviet Union. Figure 8·3 summarizes the number of available sample data as a function of time.

8.3.1. Geographical distribution of radioactive contamination

Radioactive contamination of the Arctic has occurred at two different scales:

1. Widespread contamination, such as that associated with global nuclear weapons testing, Sellafield releases and the Chernobyl accident.
2. Localized contamination of smaller areas (e.g., resulting from the Thule nuclear weapons accident and radioactive wastes dumped at sea).

The following presentation focuses on ^{137}Cs and ^{90}Sr, since these radionuclides are important for determining dose to humans, and considerable data exist on each of them.

8.3.1.1. Widespread contamination of land and sea

Terrestrial contamination

The two major sources of fallout in the Arctic region have been nuclear weapons testing and the Chernobyl accident.

A total of 520 atmospheric nuclear weapons tests have been carried out (UNSCEAR 1993). Of these, 88 have taken place in the Arctic on the island of Novaya Zemlya. The majority of radiocaesium deposition occurred during the period 1955-1966 as a result of (global fallout from) atmospheric nuclear weapons tests performed in the northern hemisphere. Precipitation and latitude have been identified as the principle factors determining the spatial variation in global fallout (UNSCEAR 1993).

The AMAP radioactivity assessment group, through the AMAP radioactivity data center, devised a novel method of estimating spatial variation in ^{137}Cs deposition in the Arctic using Geographical Information Systems (GIS). An annual relationship between precipitation and radiocaesium deposition has been derived for Tromsø in Norway (Playford *et al.* 1993). The annual relationships have been combined with mean annual precipitation (at a resolution of 0.5° × 0.5°) for the northern hemisphere (Leemans and Cramer 1991) to calculate the input of radiocaesium within each year using GIS. A latitudinal correction based on UNSCEAR was also applied initially. It is possible to calculate the decay corrected ground deposition or the total, integrated, ground deposition from nuclear weapons testing for any spatial unit in any year. This approach does not consider physical losses by lateral or vertical transport.

The predicted values of deposition were compared to measurements of soil inventories contained in the AMAP database, and those published for the UK by Cawse and Horrill (1986). From these comparisons, it appeared that the use of a latitudinal correction resulted in an overprediction of ground deposition, possibly because the latitudinal variations in ^{137}Cs deposition and precipitation are interrelated. Without latitudinal correction, predictions were in reasonable agreement with measured deposition. Consequently latitudinal correction has not been used in deriving the fallout map but since this is a novel approach further validation is required. An estimate of the distribution of

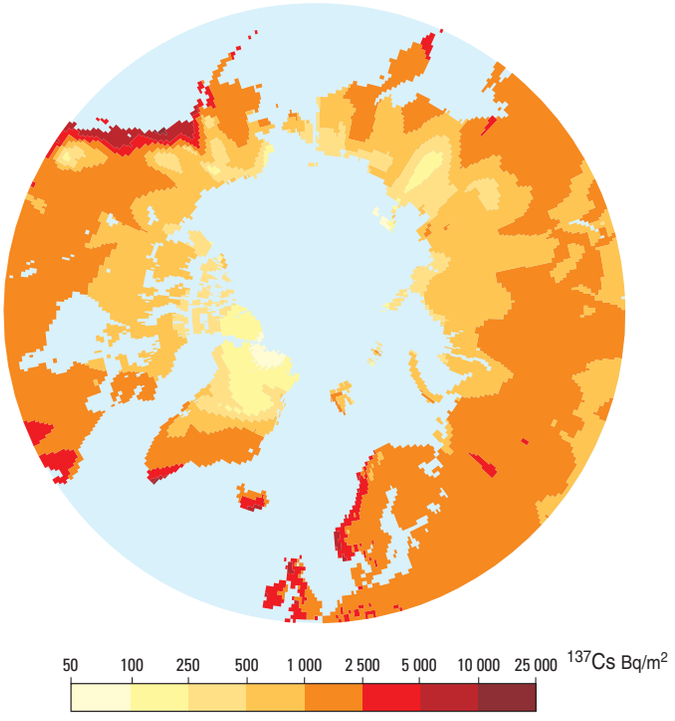


Figure 8·4. Estimated ground deposition of nuclear weapons fallout of ^{137}Cs based on precipitation data, decay corrected to 1995.

^{137}Cs ground deposition from fallout from nuclear weapons tests derived by the AMAP radioactivity data center is shown in Figure 8·4. Estimates of ^{137}Cs ground deposition based on mean precipitation values do not take into account years with extreme rainfall or drought which, especially in the early 1960s, could have produced local variations in ground deposition. Nevertheless, the validation carried out thus far suggests this approach gives reasonable estimates of ^{137}Cs ground deposition from global fallout.

The GIS approach could also be used to predict ground deposition of ^{90}Sr by applying the ratio between ^{137}Cs and ^{90}Sr in global fallout of 1.6 : 1 (UNSCEAR 1988). Estimates of the mean ground deposition of ^{137}Cs from nuclear weapons testing for different countries, derived from the GIS, are shown in Table 8·3. The areas with the lowest estimated ground deposition from nuclear weapon testing, are in the Russian Far East, North Greenland and northern and central Canada. The highest estimated ground deposition has occurred in the coastal areas of Norway and Alaska, the southern tip of Greenland and the southwest coast of Canada.

Table 8·3. Predicted mean ground deposition of ^{137}Cs in Arctic countries from atmospheric nuclear weapons testing, decay corrected to 1995, using GIS-based approach.

Arctic country/ region	Predicted mean ground deposition, Bq/m^2
Canada	730
Greenland [a]	1400
Iceland	2900
Norway	1900
Sweden	1500
Finland	1400
Russia (east)	700
Russia (west)	1000
United States (Alaska)	1300

a. Coastal inhabited areas of Greenland only, not including inland ice cap.

The predictions are based on measurements of combined wet and dry deposition collected at Tromsø. However, using this approach, ground deposition in regions which receive low annual precipitation will be underestimated. For instance, at Thule, Greenland, where annual precipitation is low (150-200 mm/y) compared with much of the Arctic, a value of predicted decay-corrected ground deposition in 1970 of 510 Bq/m^2 can be compared to a measurement of 950 Bq/m^2 (Aarkrog 1978). This indicates a discrepancy of 440 Bq/m^2, probably due to the higher proportion of dry deposition at Thule, compared to Tromsø. However, as the contribution of dry deposition will vary according to precipitation rate, it is not possible to apply a simple correction to the total predicted surface. Hence, when predicting deposition from global fallout using the approach described, it is necessary to bear in mind that ground deposition will be underestimated in drier areas.

The GIS-based approach used for Figure 8·4 can also be used to estimate the total inventory of ^{137}Cs and ^{90}Sr from global fallout over land. The estimated inventory over land north of 60°N is 35.0 PBq of ^{137}Cs and 21.9 PBq for ^{90}Sr. These estimates can be compared with estimates based on UNSCEAR data suggesting a total integrated deposition of ^{90}Sr above 60°N of 41.8 PBq. Assuming that the total area of land above 60°N is 50.2%, and accounting for latitudinal variations, then the amount of deposition on land using the UNSCEAR approach would be 25.6 PBq (16.2 PBq for the sea). As the ratio between ^{137}Cs and ^{90}Sr in global fallout is 1.6:1, the present inventory of global fallout ^{137}Cs to Arctic land masses based on the UNSCEAR approach is 41.0 PBq which is in reasonable agreement with the AMAP radioactivity data center estimate given above.

Following the Chernobyl accident in the Ukraine in April 1986, radionuclides were released to the atmosphere and fallout was observed in most European countries. The fallout levels in the different areas were dependent on the daily discharge rates, distance from the source and climate (wind direction, wet deposition). Adjacent areas in the Ukraine, Belarus and Russia were particularly heavily affected with high contamination levels of up to about 37 GBq/m^2 (100 Ci/km^2) of ^{137}Cs observed outside the 30 km Chernobyl exclusion zone. The prevailing winds during the accident were southeasterly. Rainfall during the passage of the contami-

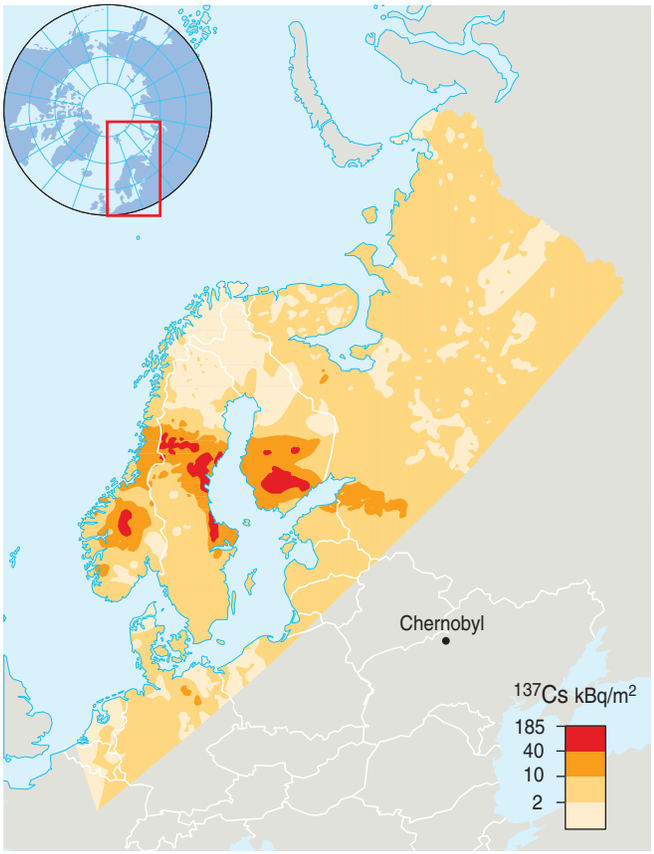


Figure 8·5. Ground deposition of ^{137}Cs from the Chernobyl accident (values normalized to May 10, 1986) (after EU/CIS JSP-6 1996, and data from national sources).

nated plume over central Norway and Sweden gave rise to considerable fallout and many other areas in Europe were also affected. The fallout was heterogeneously distributed. Chernobyl fallout affected a smaller part of the Arctic region than nuclear weapons fallout, and the extent of deposition of ^{137}Cs in 1986 after the Chernobyl fallout can be seen in Figure 8·5. In Scandinavia, areas just south of the Arctic Circle were considerably affected by the Chernobyl fallout with radiocaesium deposition of up to 200 kBq/m^2 .

In the northern parts of Scandinavia and Finland, ground deposition was only about 1-2 kBq/m^2, which was similar to that due to nuclear weapons global fallout. In the Russian Arctic, ground deposition of Chernobyl ^{137}Cs fallout on the Kola Peninsula was up to 1 kBq/m^2. At sites further east, ^{137}Cs deposition declined; in the region of Arkhangelsk about 220 Bq/m^2 was deposited, and in the Asian part of Zapolyarie about 40 Bq/m^2. The deposition of ^{90}Sr after the Chernobyl accident was an order of magnitude lower than that of ^{137}Cs.

Radionuclide contamination of land led to uptake in a variety of environmental flora and fauna. At present, the highest activity concentrations can be observed in natural foodstuffs such as reindeer meat, mushrooms, freshwater fish and berries (Figure 8·6). Products such as goat milk, goat cheese and lamb meat, derived from animals ingesting vegetation in semi-natural ecosystems, also have ^{137}Cs activity concentrations higher than those in many agricultural ecosystems. Contemporary ^{137}Cs activity concentrations in reindeer meat from different parts of the Arctic can be seen in Figure 8·25 later in this section.

Marine contamination

The anthropogenic sources contributing to the contamination in the marine environment are mainly nuclear weapons fallout and releases from Sellafield and the Chernobyl accident. Caesium-137 activity concentrations in surface seawater are shown for different years in Figures 8·7 and 8·8 (next page). Relatively high values occur in the vicinity of the North Pole over the Lomonosov Ridge (Figure 8·8) compared with other areas. A slight increase in ^{137}Cs activity concentration can also be seen in the Laptev Sea compared with the Kara and Barents Seas. East of the outlets of the large Siberian rivers, namely the Ob, Yenisey and Lena, the ^{137}Cs activity concentrations decrease. This geographical trend follows a decrease in salinity and probably reflects dilution of ^{137}Cs in the seawater by the input of river water. This is further confirmed by the increased ^{90}Sr/^{137}Cs ratio in

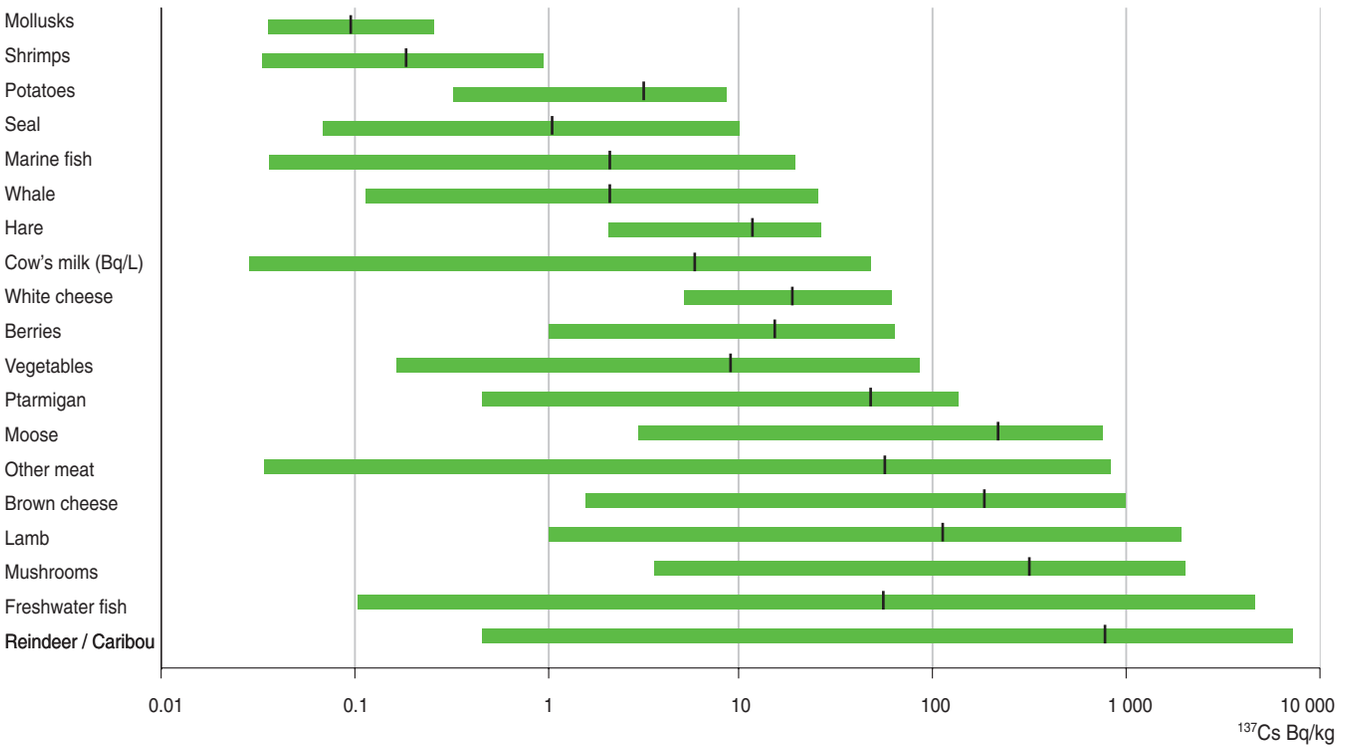


Figure 8·6. Ranges and average values of ^{137}Cs activity concentrations in food products, from data in the AMAP radioactivity database.

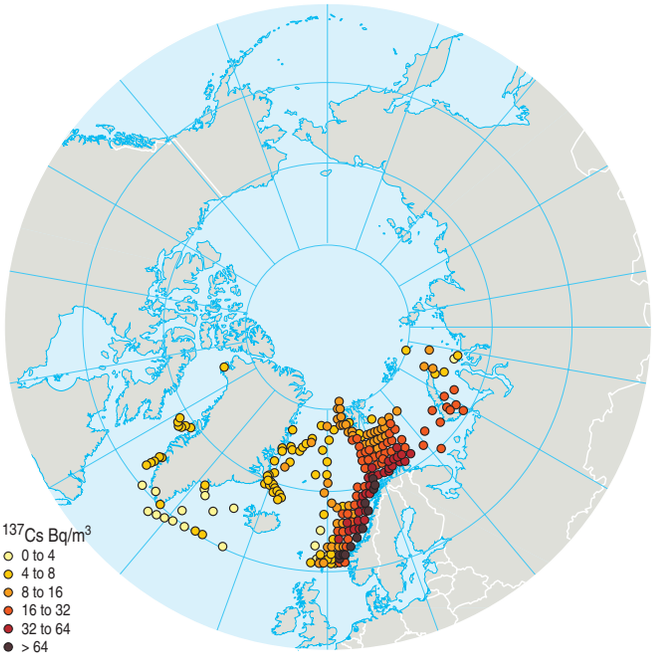


Figure 8·7. ^{137}Cs activity concentrations in surface seawater in 1979 and 1982.

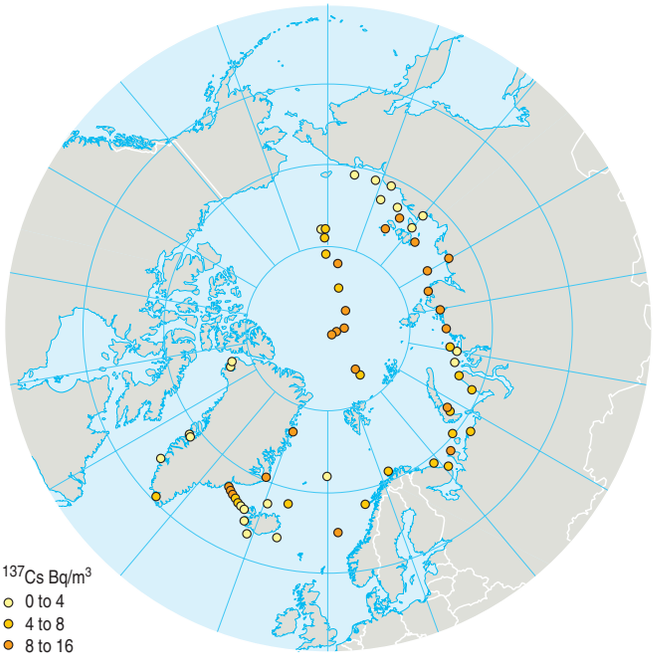


Figure 8·8. ^{137}Cs activity concentrations in surface seawater in 1994.

the low-salinity waters of the northeastern Kara Sea. In the vicinity of Greenland, the highest activity concentrations are found along the east coast. The lowest measured activity concentrations are found in the East Siberian Sea and over the southeastern part of the Makarov Basin.

Measurements of radiocaesium in Barents Sea water are available since 1970. As seen in Figure 8·9, the pattern of releases of ^{137}Cs from Sellafield are reflected in the levels of ^{137}Cs measured in the Barents Sea, with a lag time due to transportation of approximately four to five years. No measurements of ^{137}Cs are available in the Barents Sea before 1970. However, earlier ^{137}Cs activity concentrations can be estimated from measurements of ^{90}Sr which are available since 1963. Before the increase in liquid discharges from Sellafield in the late 1960s/early 1970s, the only major source contributing ^{137}Cs to the Barents Sea was fallout from nuclear weapons testing, in which the ratio of ^{137}Cs/^{90}Sr was 1.6. In the years of maximum global fallout (1963-1964), the measured activity concentration of ^{90}Sr in surface water in the Barents Sea was about 20 Bq/m^3 (RCRA 1997). Applying the ^{137}Cs/^{90}Sr fallout ratio of 1.6 to this value, the ^{137}Cs activity concentration can be estimated to be about 30 Bq/m^3. This estimate is probably too high because caesium has a higher affinity for particles than strontium, and is therefore removed more quickly from the water column. Consequently, the peak in ^{137}Cs activity concentrations of around 50 Bq/m^3 measured in the Barents Sea in the early 1980s, which was primarily due to releases from Sellafield, is probably the highest activity concentration which has occurred in this Sea.

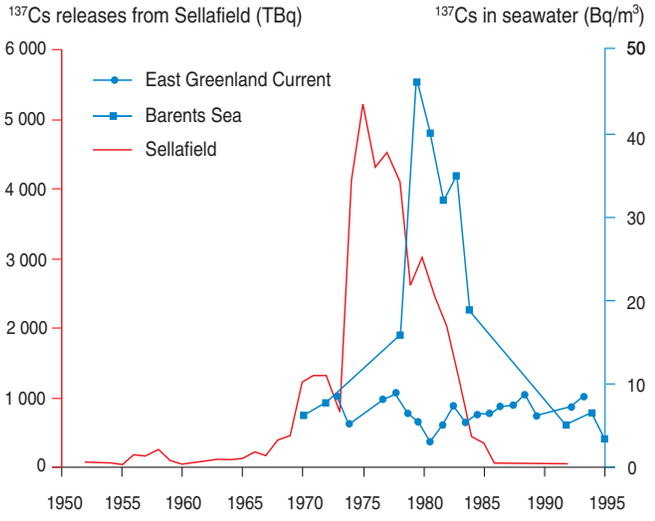


Figure 8·9. Seawater concentrations of ^{137}Cs in the Barents and East Greenland Seas compared to the yearly releases from Sellafield.

In Figures 8·7 and 8·8, ^{137}Cs activity concentrations in surface seawater are shown for 1979, 1982 and 1994, respectively. These figures also show clearly the input from Sellafield to the Arctic area. The distribution pattern is consistent with the transport of ^{137}Cs from Sellafield, which indicates that the maximum releases in the 1970s are now reflected in the peaks found in the vicinity of the North Pole and in the Laptev Sea. In 1994, the rate of input of ^{137}Cs from Sellafield to the Barents Sea was estimated to be 200-300 TBq/y (Kershaw and Baxter 1993a) with a total inventory of 10-15 PBq. Recent measurements of the distribution of ^{137}Cs in the Arctic show increased activity concentrations in waters east of Greenland which are of polar origin. These are attributable to radiocaesium from earlier Sellafield discharges that have entered the Arctic Ocean circulation and been transported back into the Atlantic through the East Greenland Current. However, in addition to the contribution from atmospheric fallout, the Arctic Sea may also be contaminated by marine transport from the North Sea and the Baltic Sea, the catchments of which have both received considerably more fallout from Chernobyl (by combined fallout and runoff) than the Arctic Ocean. Contamination of the Baltic Sea, as a result of atmospheric fallout following the Chernobyl accident, was very heterogeneous. The highest ^{137}Cs activity concentrations occurred in the Gulf of Bothnia and the Gulf of Finland where they were about two orders of magnitude higher than before the Chernobyl accident. ^{90}Sr activity concentrations in the waters of the Gulfs of Finland and Riga in May 1986 were increased by 20% compared with pre-accident levels of about 20-25 Bq/m^3 . Thus, the ^{137}Cs inventory in the Baltic Sea increased about tenfold as a result of the Chernobyl accident. This is further discussed in section 8.5.

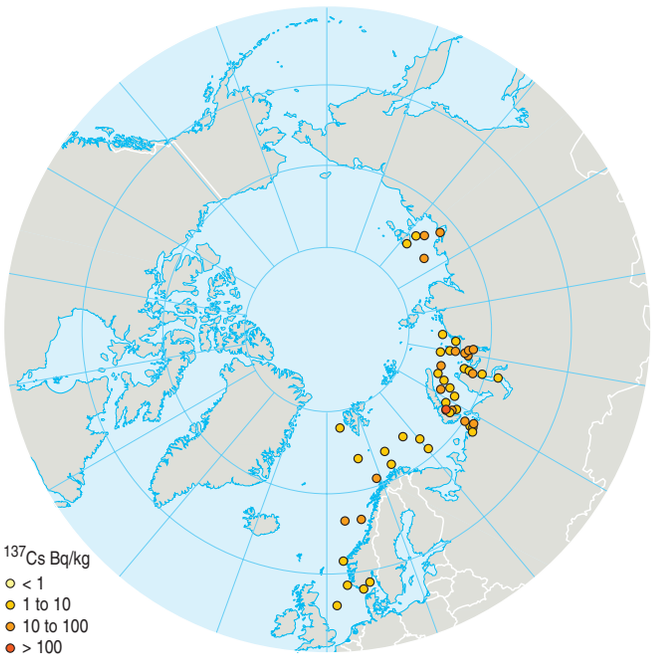


Figure 8·10. Average ^{137}Cs activity concentrations in surface sediments of some Arctic seas sampled from 1992 to 1995.

An overview of average levels of ^{137}Cs in surface sediments from some Arctic seas sampled between 1992 and 1995 is given in Figure 8·10. For most locations, average ^{137}Cs activity concentrations are below 10 Bq/kg. In the Norwegian Sea and in the open Kara Sea, some locations had samples with activity concentrations up to 100 Bq/kg. This may reflect unusual sedimentation rates or sediment characteristics. The only locations where samples with activity concentrations over 100 Bq/kg were found were in Chernaya Bay, on the southern coast of Novaya Zemlya, and close to dumped nuclear wastes adjacent to Novaya Zemlya, where levels up to several thousand Bq/kg are found. Events involving leakage from the Russian Northern Fleet's storage sites for spent nuclear fuel on the Kola Peninsula have occurred, and have probably contaminated sediments adjacent to the stores. However, it has not been possible to obtain quantitative data on the contamination of these areas.

Radionuclides can be taken up from seawater and sediments by marine plants and animals. Current activity concentrations in selected marine fish from Arctic and adjacent marine areas are shown later in this section, together with ^{137}Cs activity concentrations in marine mammals such as whales and seals. Marine food products are generally much less contaminated by radionuclides than food products from the terrestrial environment.

8.3.1.2. Localized contamination

8.3.1.2.1. Short-range fallout from Novaya Zemlya tests

There have been some 130 tests at Novaya Zemlya, 88 in the atmosphere, 3 underwater and 39 underground (Mikhailov *et al.* 1996). These tests have been mainly confined to three areas, identified as A, B and C, with local radionuclide contamination, as shown in Figures 8·11 and 8·12.

Radioactive contamination in area A of the island can be associated with five different explosions.

- A radioactive trace which crossed the Koushny Peninsula in a southerly direction following the underwater nuclear explosion in 1955. This trace has a width of approximately 2 km and an area of several km^2. The radioactive contaminants (^{137}Cs, ^{90}Sr and ^{60}Co) are largely present in the top 6-10 cm of the sediments. At the present time, ground contamination by ^{137}Cs and ^{90}Sr is in the range (30-480 kBq/m^2 ≈0.8-13 Ci/km^2) with maximum dose rates of 30 μR/h.
- A radioactive trace from the explosion conducted on the surface in 1957 extended from the southern shore of the Chernaya Bay to the eastern coast of Novaya Zemlya, crossing the whole of southern Novaya Zemlya. The resulting plume extended approximately 1500 km from the site of the explosion. Radioactive contamination within a 400 m radius around the crater comprised ^{90}Sr, ^{137}Cs, ^{60}Co, ^{152}Eu and ^{239}Pu and is now characterized by dose

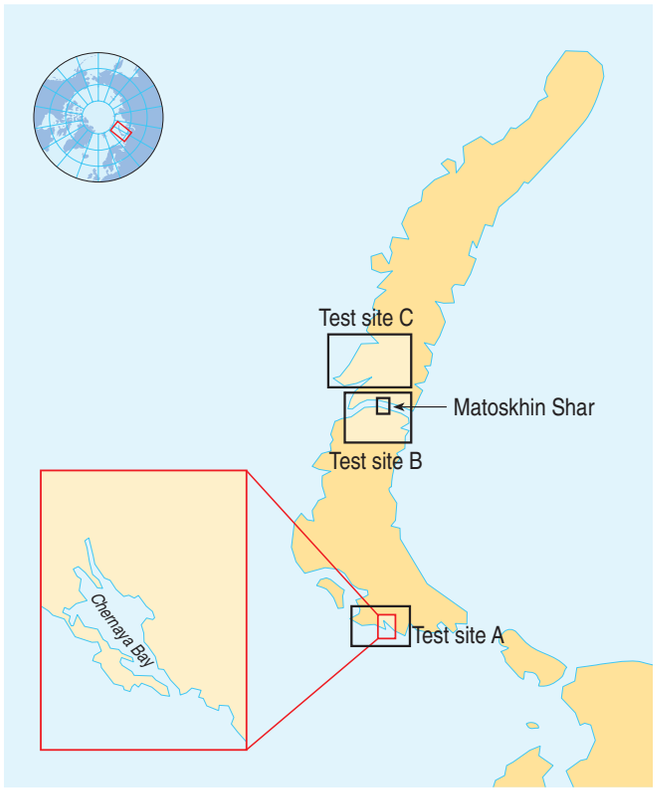


Figure 8·11. Nuclear weapons test sites on Novaya Zemlya.

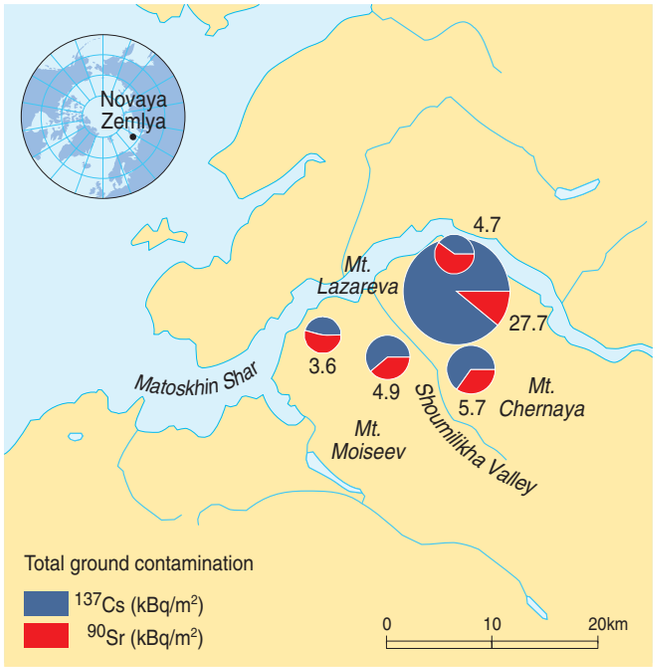


Figure 8·12. Local radionuclide contamination on Novaya Zemlya.

rates of 10-150 μGy/h (1000-15 000 μR/h). The current ground contamination of ^{137}Cs is 9.3-1110 kBq/m^2 (0.25-30 Ci/km^2), and of ^{90}Sr, 1.5-555 kBq/m^2 (0.04-15 Ci/km^2). Thirty km from the test crater current contamination by ^{137}Cs plus ^{90}Sr is 26 kBq/m^2 (0.7 Ci/km^2).

- A radioactive trace from a low level atmospheric explosion in September 1957. At the present time, an area with a diameter of approximately 0.5 km has a dose rate of approximately 0.30 μGy/h (30 μR/h) and ground contamination of ^{152}Eu of ≈2-130 kBq/m^2 (0.05-3.5 Ci/km^2), ^{60}Co of ≈22 kBq/m^2 (0.6 Ci/km^2), and ^{90}Sr plus ^{137}Cs of approximately 2 kBq/m^2 (0.05 Ci/km^2).
- A radioactive trace from an explosion that took place on the surface of the sea in 1961 extends from Chernaya Bay to the northeastern part of Novaya Zemlya and is characterized by current dose rates of 0.20-0.25 μGy/h (20-25 μR/h) and contamination values for ^{90}Sr plus ^{137}Cs of ≈ 4-40 kBq/m^2 (0.1-1.2 Ci/km^2).
- In 1973, there was an underground explosion following which a vented release of gas products of 20 minutes duration produced a plume extending in a southeasterly direction. Current dose rates near the detonation site are 0.25 μGy/h (25 μR/h), part of which arises from approximately 37 GBq (1 Ci) of ^{137}Cs created from disintegration of the vented ^{137}Xe.

In area B, maximum external dose rates in several places, close to the epicenters of the explosions, reach 1 μGy/h (100 μR/h), but in the rest of the area, the values decline to 0.10-0.20 μGy/h (10-20 μR/h).

Table 8·4. Radioactive contamination in four regions of area C of Novaya Zemlya.

Region	Area, km^2	Dose level, Gy/h (μR/h)	^{137}Cs kBq/m^2 (Ci/km^2)	^{152}Eu kBq/m^2 (Ci/km^2)	^{60}Co kBq/m^2 (Ci/km^2)
Eastern part	0.4	0.30-0.40 (30-40)	3 (0.07)	5.6-20 (0.15-0.6)	2 (0.05)
Central part	0.3	0.25-0.35 (25-35)	2.2 (0.06)	17 (0.45)	–
Western part	0.5	0.25-0.30 (25-30)	19 (0.5)	–	–
Northern part	0.3	0.20-0.25 (20-25)	2.0 (0.05)	–	–

In area C, four regions of radioactive contamination were measured, and results are given in Table 8·4.

The highest doses and deposition levels occur in area A, where levels are 10-100 higher than in areas B and C. In addition, there is a larger range of radionuclides present at the A area site, including ^{239}Pu.

To summarize, the current average ground contamination of ^{137}Cs at the Novaya Zemlya test site is 3 kBq/m^2 (0.09 Ci/km^2) of ^{137}Cs and 2 kBq/m^2 (0.06 Ci/km^2) of ^{90}Sr. During the underground nuclear explosions carried out from 1964 to 1990, most of the radionuclides were retained underground at the location of the explosions. Only a small proportion of the activity (1-10%) escaped into the atmosphere leading to localized radionuclide contamination of the Novaya Zemlya test site territory.

8.3.1.2.2. **Chernaya Bay**

Chernaya Bay is a 15 km long fjordic inlet on the southwestern coast of Novaya Zemlya (Figure 8·11). It has a variable width of 1-6 km and is connected to the Pechora Sea which occupies the southeastern extremity of the Barents Sea. The Bay contains finer, organic rich sediments compared with those of the Pechora Sea, although there also exist deeper sedimentation basins within the Pechora Sea that contain increased proportions of fine material. Underwater nuclear explosions took place within the Bay in October 1955, and September 1957, and in the vicinity of the Bay in 1961. The distribution of several radionuclides in the sediments of the Pechora Sea, including Chernaya Bay, has examined by Smith *et al.* (1995a). The surface sediments of Chernaya Bay contain elevated ^{239,240}Pu (≈8500 Bq/kg), ^{241}Am (≈430 Bq/kg) and ^{137}Cs (≈160 Bq/kg) activity concentrations compared to external areas. The activation product ^{60}Co also occurs in measurable amounts (≈90 Bq/kg). Smith *et al.* (1995a) have closely examined the ratios among the plutonium and americium isotopes and made comparisons with ratios in global fallout and with areas contaminated by other nuclear explosions, including underwater devices. Whilst some contribution from radionuclides contained in wastes dumped in Chernaya Bay in 1991 (OPRF 1993) cannot be ruled out, the distribution and relationships among the transuranic isotopes suggest that the underwater explosions were the primary source of the enhanced radionuclide activity concentrations in the sediments.

The ^{239,240}Pu inventory in central Chernaya Bay is approximately 300 kBq/m^2. This is similar to other sites of major plutonium contamination, such as the vicinity of the test explosion sites at Enewetak Lagoon (Nelson and Noshkin 1973), the most contaminated area of Bylot Sound (Aarkrog *et al.* 1987) and the Irish Sea in the vicinity (i.e., within an adjacent 100 km^2 area) of the Sellafield nuclear fuel reprocessing plant (Pentreath *et al.* 1986) where sediment inventories are of comparable magnitude but exceed 300 kBq/m^2. The integrated Chernaya Bay sediment inventory of 3 TBq estimated by Smith *et al.* (1995a) is comparable with that of Bylot Sound (1 TBq) resulting from the weapons plutonium spill at Thule (Aarkrog *et al.* 1987) and of the same order as that of Enewetak Lagoon (8.5 TBq). It is, however, dwarfed by the inventory in the upper 30 cm of Irish Sea sediments of 280 TBq.

8.3.1.2.3. **The Thule accident**

In January 1968, an American B-52 aircraft carrying four nuclear weapons crashed on the ice in Bylot Sound near Thule, Greenland. The impact triggered the conventional explosive, fragmenting the weapons, and resulting in the release of plutonium onto the ice. Debris and the upper-layer of contaminated snow was removed, but the ice had been broken around the point of impact and some plutonium was deposited on the underlying sediments (see also section 8.5.3.2).

Plutonium in Bylot Sound seawater

In the summer of 1968, half-a-year after the accident, the seawater in Bylot Sound contained about 0.2 Bq ^{239,240}Pu/m^3. This was higher than measured at several other locations along the Greenland coast (Qaanaaq, Qeqertarsuaq (Godhavn), Nuuk (Godthåb), Ammassalik and Danmarkshavn) (Figure 8·13) which are assumed to be contaminated by global fallout only. As the total volume of water in Bylot Sound is 50 km^3, the amount of ^{239,240}Pu in the Thule seawater in 1968 was estimated to be 10^{10} Bq, of which half (5 GBq or 2 g plutonium) was due to the accident.

In 1970, seawater activity concentrations (0.02-0.1 Bq/m^3) of ^{239,240}Pu did not differ significantly from the global fallout background. This has been verified by subsequent sampling. The only increase found in seawater was in particle-bound ^{239,240}Pu in near-bottom water at the point of impact, probably due to resuspension of contaminated sediments.

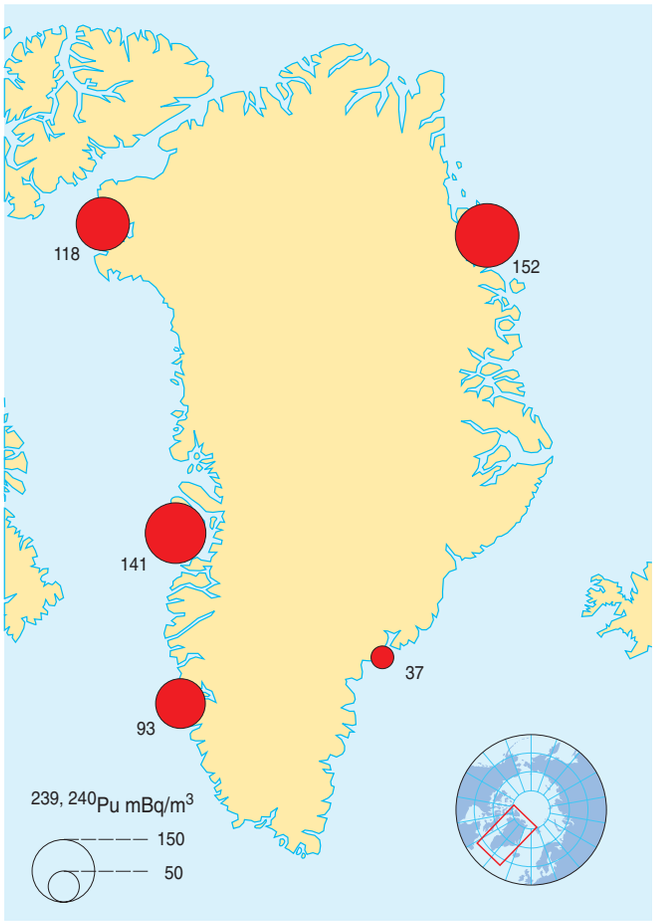


Figure 8·13. Concentrations of ^{239,240}Pu in seawater around Greenland, 1968.

Plutonium in Bylot Sound sediments

From measurements of plutonium in marine sediments collected during expeditions to Thule in 1968, 1970, 1974, 1979 and 1984 (Aarkrog 1971, 1977, Aarkrog *et al.* 1984, 1987, Smith *et al.* 1994) it was calculated that about 1 TBq, or half a kilogram of plutonium, was deposited on the bottom of Bylot Sound from the Thule accident. The amount of Pu left on the ice after the decontamination effort in 1968 was also estimated to be 1 TBq (±50%). It therefore seems likely that a substantial part of the plutonium in the sediments is derived from the melting of sea ice. On the other hand, it is evident that the highest levels are found beneath the point of impact and, as some of the contaminated ice drifted away before it melted, it seems likely that some debris entered the sea directly through the impact hole in the ice.

The low solubility of PuO_2 (high Kd value) makes it probable that most of the plutonium that might have entered the sea from the Thule accident would be bound to marine sediments on the bottom of Bylot Sound. The extensive sampling has shown that the Pu from the accident has only been minimally dispersed from the crash site (Figure 8·14). Thus, the site currently does not constitute a significant source of plutonium contamination to the surrounding environment.

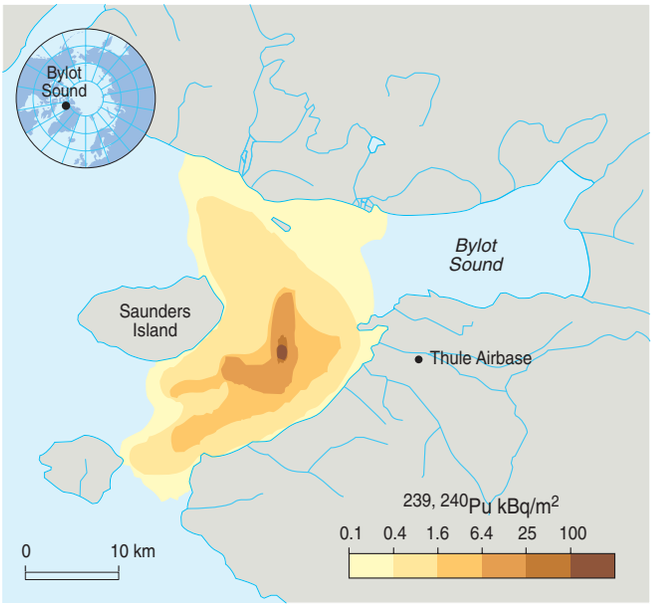


Figure 8·14. Activity concentrations of ^{239,240}Pu in sediments near Thule, Greenland.

8.3.1.2.4. Contamination at sea dumping sites

Between 1960 and 1991, the former Soviet Union carried out dumping of radioactive waste in the Kara and Barents Seas. The wastes dumped at sea included liquid and solid waste, the latter including reactor compartments and entire submarines. Some of the reactors contained spent nuclear fuel. In addition to the official information provided by the government of the Russian Federation (OPRF 1993), the International Arctic Sea Assessment Project (IASAP) of the IAEA has produced revised inventories (see also section 8.6.4).

During joint Norwegian-Russian expeditions from 1992-1994 (Strand *et al.* 1997) to the dumping sites, the dumped material was visually inspected and samples collected both beside and further away from the dumped material.

On the east coast of Novaya Zemlya, in the Abrosimov and Stepovogo Fjords, enhanced ^{137}Cs, ^{90}Sr, ^{60}Co and Pu-isotope activity concentrations were measured in sediments collected in the immediate vicinity of the dumped nuclear waste (Table 8·5). In addition, ^{152}Eu and ^{154}Eu were identified in one sample collected close to the hull of the dumped nuclear submarine in the Stepovogo Fjord. In Abrosimov Fjord, radionuclide contamination was measurable in the upper 5 cm (^{90}Sr) and upper 10 cm (^{137}Cs, ^{60}Co) sediment layer. Figure 8·15 shows the ^{137}Cs activity concentrations in the sediments. In Stepovogo Fjord, enhanced ^{137}Cs, ^{90}Sr, ^{60}Co and Pu-isotope activity concentrations were observed in the upper 5 cm of sediments collected in the close vicinity of the dumped containers (Figures 8·16 and 8·17). Substantially lower ^{137}Cs and ^{90}Sr activity concentrations were observed in sediments at greater distances from the localized objects. Activity concentrations at these latter sites were similar to values in the open Kara Sea.

In the Tsivolky Fjord, traces of ^{60}Co found in the sediments close to the dumped vessel could indicate leakage

Table 8·5. Range of radionuclide activity concentrations (Bq/kg dw) in sediments near to the dumped objects in Abrosimov Fjord and Stepovogo Fjord (Strand *et al.* 1997)

Location/ object(s)	Radionuclide activity concentrations in sediments, Bq/kg dw			
	^{137}Cs	^{90}Sr	^{60}Co	^{239,240}Pu
Abrosimov Fjord				
Containers	23-31000	4-8850	0.4-180	1-18
Vessels	38-196	0.3-3	0.5-53	0.7-2.6
Submarine reactor compartments	33-8445	0.4-3250	1-61	1-5
Stepovogo Fjord				
Containers	14-109000	1-310	<0.3-3150	0.2-28
Submarine	4-1670[a]	0.4-6	<0.1-6	<0.1-6

a. Result was not confirmed by additional sampling.

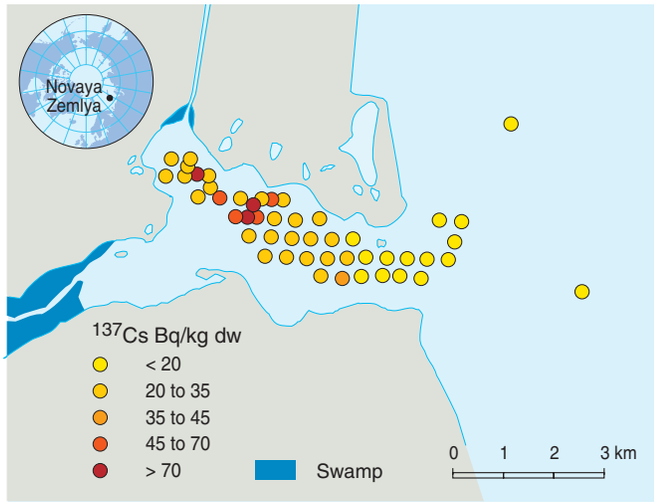


Figure 8·15. Activity concentrations of ^{137}Cs in sediments of Abrosimov Bay.

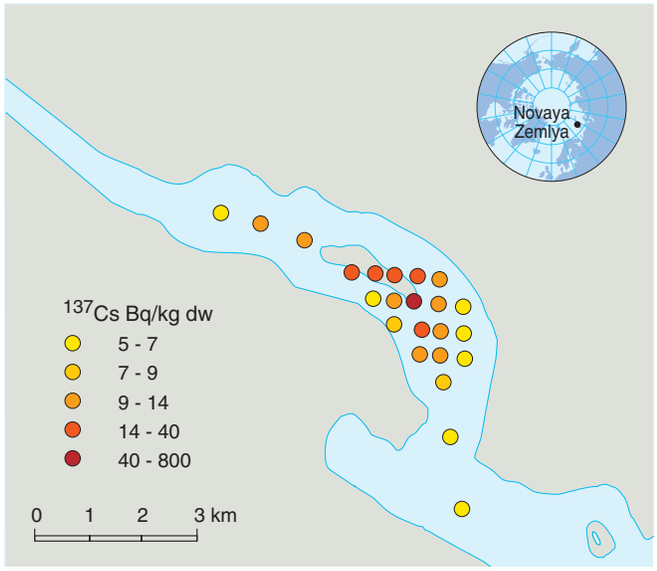


Figure 8·16. Activity concentrations of ^{137}Cs in sediments of Stepovogo Bay.

Figure 8·17. Some of the dumped containers in Stepovogo Bay.

from dumped waste. Water and sediment samples obtained from the Novaya Zemlya Trough showed no indication of leakage from dumped wastes. The only location of enhanced levels of radionuclide contamination in water was the inner Stepovogo Fjord with an increased ^{90}Sr activity concentration in the bottom water (Table 8·6). However, radionuclide

Table 8·6. Range of radionuclide activity concentrations, Bq/m^3 in seawater in 1993-94.

	Activity concentrations in seawater, Bq/m^3							
	Abrosimov Fjord		Stepovogo Fjord		Tsivolky Fjord		Open Kara Sea	
	^{137}Cs	^{90}Sr	^{137}Cs	^{90}Sr	^{137}Cs	^{90}Sr	^{137}Cs	^{90}Sr
Surface water	4-7	2-4	3-9	2-7	4-6	4-6	3-8	3-11
Near-bottom water	4-9	2-4	6-31	3-26	6-14	3-4	8-20	4-6

activity concentrations in fish from the area showed no similar enhancement. Activity concentrations in Arctic cod (*Boreogadus saida*) and Arctic char (*Salvelinus alpinus*) ranged from 0.5-2.2 Bq/kg ^{137}Cs and 0.9-1.6 Bq/kg ^{90}Sr. The activity concentration of 238Pu and ^{239,240}Pu were below the detection limits of 0.01 Bq/kg and 0.008 Bq/kg, respectively.

8.3.1.2.5. Sunken *Komsomolets* submarine

On April 7, 1989, the Soviet nuclear submarine *Komsomolets* caught fire and sank southwest of Bear Island in the Norwegian Sea. This submarine contains two torpedoes with nuclear warheads and a nuclear reactor. The reactor was shut down prior to sinking. An estimate of the inventory of the main long-lived radionuclide constituents of the reactor and nuclear weapons of the *Komsomolets* is given in Table 8·7 (Høibråten and Thoresen 1995) (see also section 8.5.3.5.1).

Scientific studies in the vicinity of the site where the *Komsomolets* is located indicate that only minor contamination can be attributed to the submarine (Table 8·8) (Kolstad 1995, Strand *et al.* 1996).

Table 8·7. Inventory of selected radionuclides in *Komsomolets*, decay corrected to 1 January, 1995 (Høibråten and Thoresen 1995).

Radionuclide	TBq
^{137}Cs	2700
^{90}Sr	2400
^{239}Pu	22

Table 8·8. Radionuclide activity concentration in sediments near to and approximately 1 nautical mile in different directions from Komsomolets compared to average values for the North European Seas in 1995.

	Radionuclide activity concentrations, Bq/kg dw				
Sample site	^{239,240}Pu	^{238}Pu	^{241}Am	^{137}Cs	^{134}Cs
Close to *Komsomolets*,					
1993[a]	0.3±0.2	90[b]	0.2±0.1	7±4	1.4±0.8
1994[a]	0.4±0.4	–	–	8±4	0.5±0.3
1 nm south (1994)	0.85	0.04	<0.7	5.1	n.d.
1 nm west (1994)	–	–	0.65	6.6	n.d.
1 nm north (1994)	0.95	0.03	<0.96	5.4	0.3
1 nm east (1994)	0.96	0.13	0.57	9.7	n.d.
Close to *Komsomolets*, 1995[a]	1.16±0.08	0.04±0.01	0.86±0.066	7.1±0.4	0.6±0.3
Other North European seas[a]	1.4±0.8	0.1±0.1	0.8±0.5	–	–

a. These numbers are averages of many measurements, the uncertainty given is the standard deviation of the measurements.
b. Plutonium-238 was detectable in only one sample.
n.d.: not detected.

8.3.2. Time dependence of radioactive contamination

A number of examples of the changes with time in radionuclide activity concentrations in Arctic environmental samples have been collated in this section. The purpose is to give a selected overview of radionuclide contamination of the Arc-

Table 8·9. Available sample data for showing time trends in radionuclide activity concentrations.

Sample type	^{90}Sr	^{137}Cs
Air		Finland, Norway, Russia
Deposition	Finland, Greenland	Finland, Russia
Lichens	Finland, Greenland	Finland, Greenland, Russia
Reindeer meat		Finland, Greenland, Norway, Russia
Freshwater	Finland, Greenland, Russia	Finland
Freshwater fish		Finland
Seawater	Greenland waters, Barents Sea, Kara Sea	Greenland waters, Barents Sea
Marine fish		Greenland waters
Marine mammals		Greenland waters
Whole body measurements		Finland, Norway, Sweden, Russia

tic, both with regard to local and temporal variations. As ^{137}Cs and ^{90}Sr are important in determining dose to man, the presentation is focused on these radionuclides and is limited to the most comprehensive radionuclide-specific data sets. This assessment focuses on specific environmental data sets, as identified in Table 8·9, from particular areas where it has been possible to show trends in the selected sample types.

In a few cases, logarithms of activity concentration have been used due to the high between-year variations. The examples given below, showing trends with time in radionuclide activity concentrations, are deliberately not reported with specific uncertainties. This is because such uncertainties would not have been comparable and, thus, might be misinterpreted. For example, if we consider the ^{137}Cs activity concentrations in lichen, as shown later in Figure 8·24, the data are given as 5-year means. These means may, for some periods and countries, have been based on all years and sampling locations, while in other cases they may be based on less complete data. Furthermore, some countries have had a more comprehensive net of sampling locations than others. If, for example, the data arise from only a limited part of an Arctic country, the sampling errors may be lower than those for a country with a more widespread sampling net, but the representativeness of the data may be better in the latter case than in the former. Hence, error indications would have been misleading.

Within the larger Arctic regions (northern Russia, northern Canada, Greenland and Alaska) local variations in radionuclide activity concentrations in environmental samples may be an order of magnitude, mainly due to heterogeneity in the deposition of nuclear weapon global fallout. For certain types of sample (e.g., ^{137}Cs in freshwater fish and mushrooms), variations may be even higher due to ecological differences. In comparison, the variation in radionuclide activity concentrations within the Arctic areas of Finland, Norway and Sweden is usually less than for the larger Arctic countries. Because of the similar dietary habits, this is reflected in the estimated individual doses from ^{137}Cs received by populations in the Arctic parts of the above Nordic countries, which appear to be similar (see section 8.4.2.).

8.3.2.1. Air and deposition

Measurements of radionuclide activity concentrations in air provide indications of recent releases. Figure 8·18 shows the ^{137}Cs activity concentration in air samples collected since the early seventies in the Tromsø area (Tromsø and Skibotn) (Norway), in the Helsinki area of Finland (outside the Arctic), and in Russia (Naryan Mar and Norilsk). Since 1986, measurements have been made in Arctic Finland at Rovaniemi (Sinkko *et al.* 1987, Aaltonen *et al.* 1990–1991, Toivo-

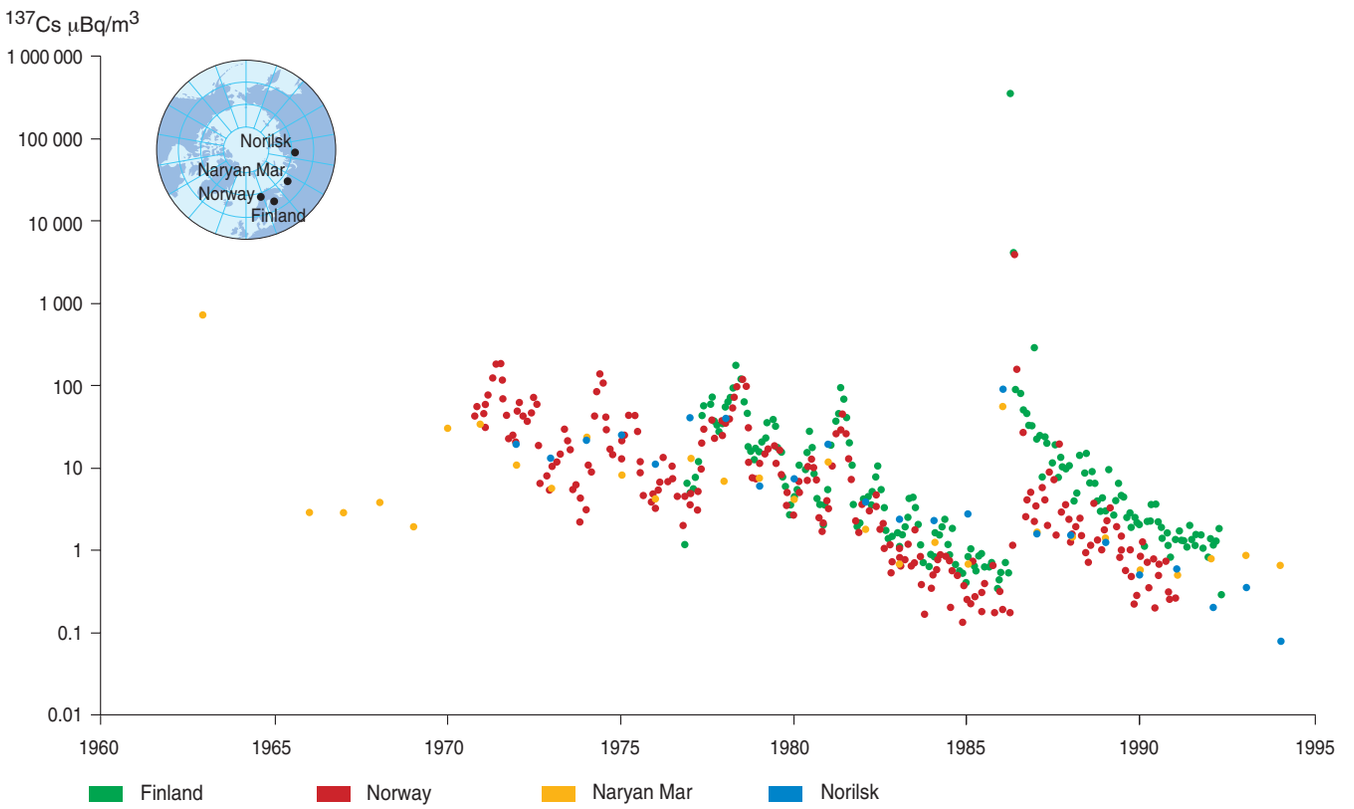


Figure 8·18. Changes with time in ^{137}Cs activity concentration in air in Norway, Finland and Russia.

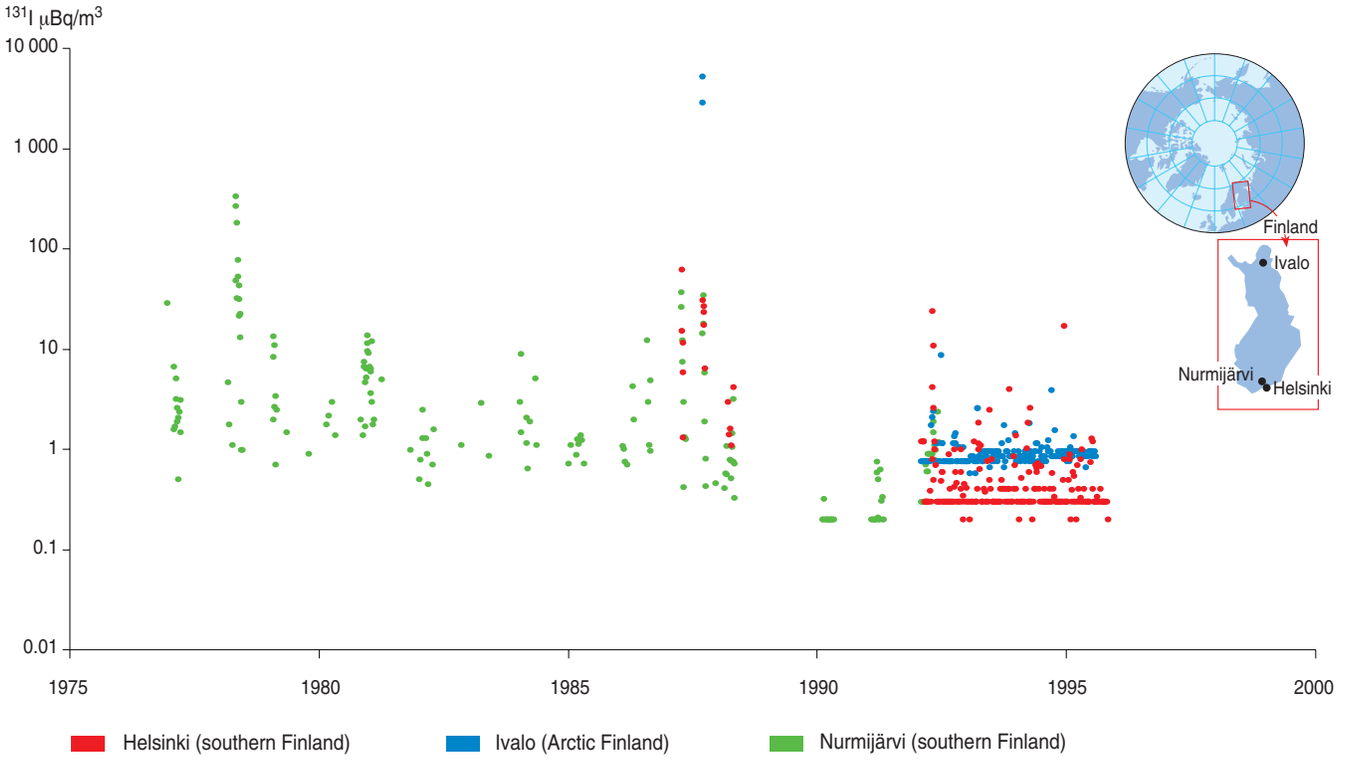


Figure 8·19. Changes with time in ^{131}I activity concentration in Finnish air samples.

nen *et al.* 1992). Activity concentrations decreased rapidly after 1981, which was the year after the last atmospheric nuclear test carried out in China. In 1986, the Chernobyl accident increased the air concentrations of ^{137}Cs by several orders of magnitude in certain Arctic areas. Thus, in 1986, the annual mean air activity concentration measured at Skibotn was about 350 μBq $^{137}Cs/m^3$, and that at Rovaniemi, Finland, approximately 1200 μBq $^{137}Cs/m^3$. Since 1986, air concentrations of ^{137}Cs have decreased to 0.5 μBq/m^3 at Skibotn and about 2.5 μBq/m^3 at Rovaniemi. In recent years, the decrease has been less rapid than that observed in the first few years following the Chernobyl accident, probably due to the resuspension of deposited ^{137}Cs.

Iodine-131 activity concentrations in Finnish air samples are given in Figure 8·19. Due to its high fission yield, volatility and short physical half-life (8 days), ^{131}I may be used as a short-term indicator of recent releases of fission products, such as leakage from underground nuclear test explosions or accidental atmospheric releases from nuclear installations. The peak shown in 1987 was due to a Soviet underground nuclear weapons test carried out in August that year at Novaya Zemlya (Bjurman *et al.* 1990).

Wet deposition, in rain or snow, is the main mechanism for the transfer of radionuclides from the atmosphere to terrestrial and aquatic ecosystems. Three of the Arctic areas are represented by a nearly complete time series of measurements for wet + dry deposition: Finland, Russia and Greenland. Data are available for both ^{90}Sr (Figure 8·20) and ^{137}Cs (Figure 8·21) in Arctic Finland (Salo *et al.* 1966-1996a, 1966-1996b), for ^{90}Sr in Greenland (Figure 8·20) and for ^{137}Cs in northwest Russia (Figure 8·21).

All data show the same trend with time. Deposition of fallout from atmospheric nuclear weapons testing peaked in 1963. The deposition rate of ^{90}Sr and ^{137}Cs in global fallout

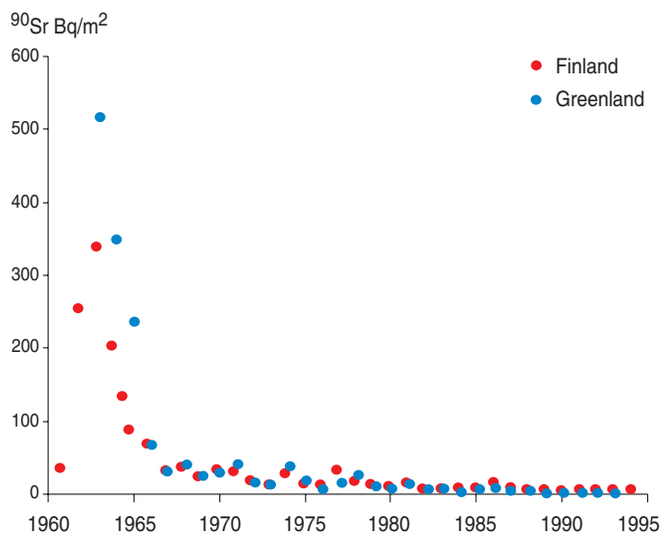


Figure 8·20. Changes with time in wet and dry deposition of ^{90}Sr in Arctic Finland and Greenland.

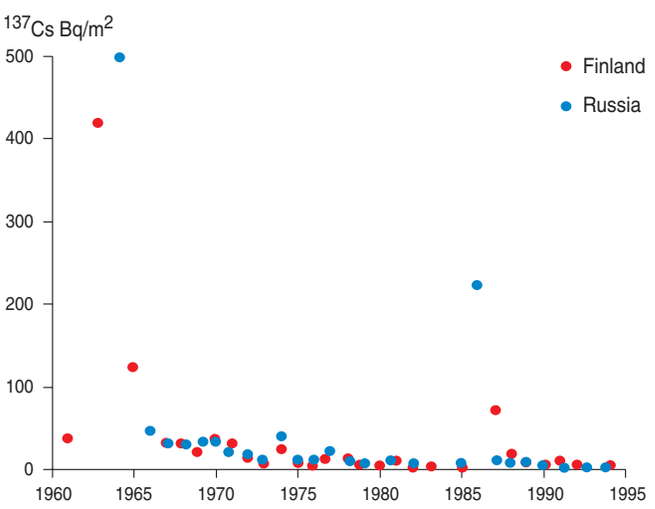


Figure 8·21. Changes with time in wet and dry deposition of ^{137}Cs in Arctic Finland and north west Russia (Nenets Autonomous Okrug).

has declined with an effective half-life of 3.6 ± 0.3 years since the peak in 1963. The decay in the rate of deposition to the ground was rapid from 1963 to 1966 with a half-life of a little more than one year, followed by a period of slower decay until 1981 with an effective half-life of 4-5 years. From 1981 until the Chernobyl accident, the decrease in air concentrations of ^{137}Cs again became more rapid, corresponding to an effective half-life of about two years. These observed variations in the effective decay rate of ^{90}Sr and ^{137}Cs deposition may be explained as follows: from 1963 to 1966 atmospheric radioactive contamination was predominantly in the stratosphere and the decay in the atmospheric levels depended mostly on the stratospheric mean residence time. From the middle of the sixties until 1980, China and France performed thermonuclear tests in the atmosphere and this contributed more radionuclides to the atmosphere, hence the effective decay was reduced. In 1986, the Chernobyl accident increased ^{137}Cs and ^{90}Sr annual deposition rates.

Table 8·10. Estimated integrated ground deposition densities of ^{90}Sr and ^{137}Cs in Arctic countries from global fallout, kBq/m^2 since 1950 without decay correction (decay corrected values are given in Table 8·3).

Region	Integrated ground deposition using data compiled in AMAP database, kBq/m^2		Mean GIS-based integrated ground deposition, kBq/m^2
	^{137}Cs	^{90}Sr	^{137}Cs
Arctic Canada			1.4
Arctic Finland	1.7	2.5	2.8
Greenland	2.7	4.3[a]	2.7[c]
Iceland			5.8
Arctic Norway	2.5[b]	4.4	3.7
Arctic Russia	1.7[b]	3.1	
Arctic Russia, west			2.1
Arctic Russia, east			1.4
Arctic Sweden	1.6[b]	2.9	3.0
United States, Alaska			2.6

a. Value estimated from ^{137}Cs.
b. Value estimated from ^{90}Sr.
c. Includes the land area covered by the ice cap.

The estimated integrated ground deposition of ^{90}Sr and ^{137}Cs, for various Arctic regions since 1950 (without decay-correction), based on measurements or calculated from the ^{137}Cs : ^{90}Sr ratio of 1.6, are given in Table 8·10. For the years prior to measurements in the various countries, deposition densities were estimated from measurements of ^{90}Sr deposition carried out by USAEC in New York (HASL 1958-1978). These values have been compared with (non-decay corrected) estimates from GIS-based analysis of the integrated ground deposition by the AMAP radioactivity assessment group. GIS-based estimates decay-corrected to 1995, shown in Table 8·3, are ca. 50% lower (see also section 8.3.1.1).

8.3.2.2. **Terrestrial and freshwater ecosystems**

8.3.2.2.1. **Lichen**

Lichen is an efficient collector of atmospheric contamination due to its large surface area, nutritional uptake characteristics and its slow growth. Lichen is the winter fodder of reindeer and is thus an important determinant of ^{137}Cs activity concentrations in most populations of reindeer during winter.

Nearly complete time series data are available for ^{90}Sr and ^{137}Cs in lichen from Arctic Finland, Greenland and Arctic Russia. Mean ^{137}Cs activity concentrations in lichen from Arctic Finland (Rissanen and Rahola 1996, Rahola and Rissanen 1996), Arctic Russia and Greenland are shown in Figure 8·22. Activity concentrations peaked in 1965-1969. The Chernobyl accident was clearly reflected in ^{137}Cs activity concentrations in Finnish lichen, whilst only a modest signal was observed in lichen from Greenland and Arctic Russia.

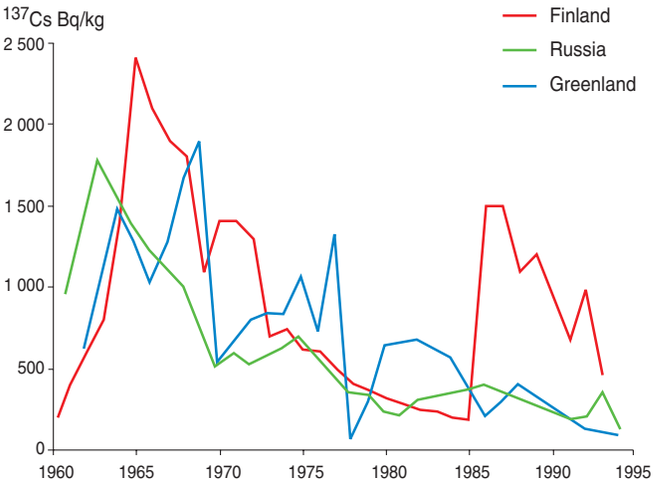


Figure 8·22. Changes with time in ^{137}Cs activity concentration in lichen from Arctic Finland, north west Russia, and Greenland.

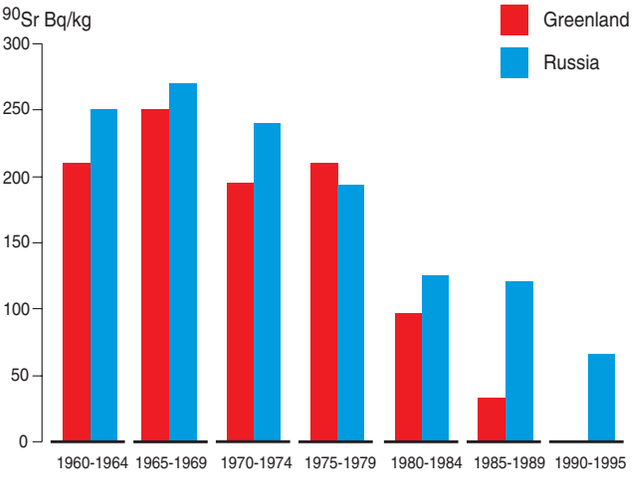


Figure 8·23. Changes with time in ^{90}Sr activity concentration (5-year means) in lichens in Greenland and Russia.

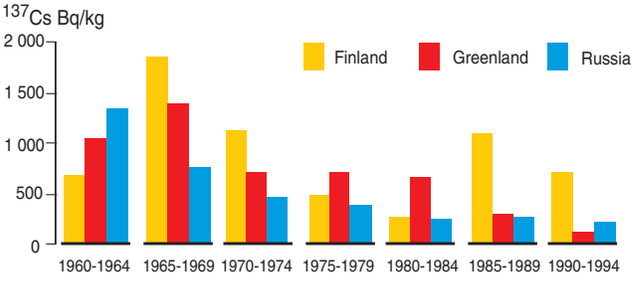


Figure 8·24. Changes with time in ^{137}Cs activity concentrations (5-year means) in lichens in Arctic Finland, Greenland, and Russia.

The observed effective ecological half-life of ^{137}Cs in lichens from Arctic Finland and Arctic Russia since the mid-1960s and until the Chernobyl accident was 5-6 years. The real effective ecological half-life should be shorter but the true value is masked by fresh global fallout added during the period of observation from atmospheric nuclear tests conducted by China and France between 1974 and 1980.

The ^{90}Sr activity concentrations in lichen from Greenland and Arctic Russia were similar until the Chernobyl accident when lichen in Russia became significantly more contaminated (Figure 8·23). Compared with ^{137}Cs (Figure 8·24), ^{90}Sr activity concentrations in lichen were significantly lower, by almost a factor of 3-5. Although the Chernobyl accident increased ^{137}Cs activity concentrations in some Arctic lichen, contamination was still lower than in 1965-1969 in most of these Arctic areas.

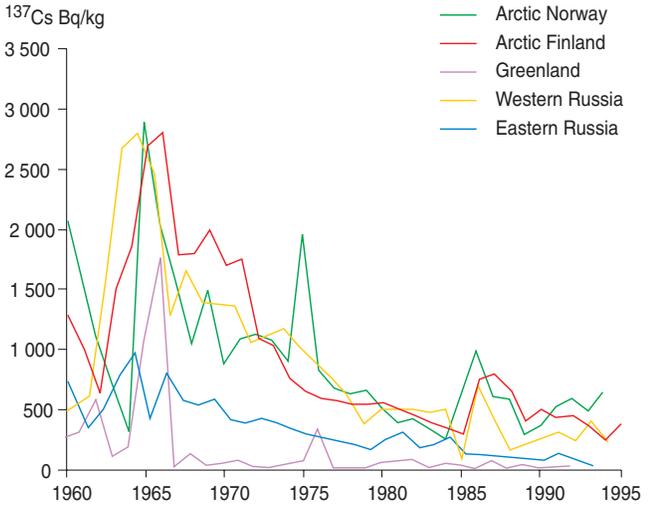


Figure 8·25. Changes with time in activity concentration of ^{137}Cs in reindeer meat in Arctic Norway, Arctic Finland, Greenland, and Arctic Russia.

8.3.2.2.2. **Reindeer meat**

Reindeer meat is consumed by some Arctic populations in substantial amounts. Due to the relatively high ^{137}Cs contamination of reindeer meat, this important component of diet is a major contributor to intake of anthropogenic radionuclides by some Arctic populations.

Radionuclide activity concentrations in reindeer meat are determined by those in fodder. In summer time, reindeer mainly eat herbaceous vegetation, whereas in winter they eat lichen. Because of the higher ^{137}Cs activity concentrations in lichen than in other vegetation, reindeer meat is more highly contaminated in winter and spring than in summer. Maximum ^{137}Cs activity concentrations in reindeer in all Arctic countries were reached in the middle of the 1960s (Figure 8·25). Since then, ^{137}Cs activity concentrations have decreased gradually with an observed effective ecological half-life of 4.9 ± 1.1 years in western Arctic Russia, 8.3 ± 4.7 years in eastern Arctic Russia, 5.4 ± 1.1 years in Arctic Finland (Rissanen and Rahola 1996, Rahola and Rissanen

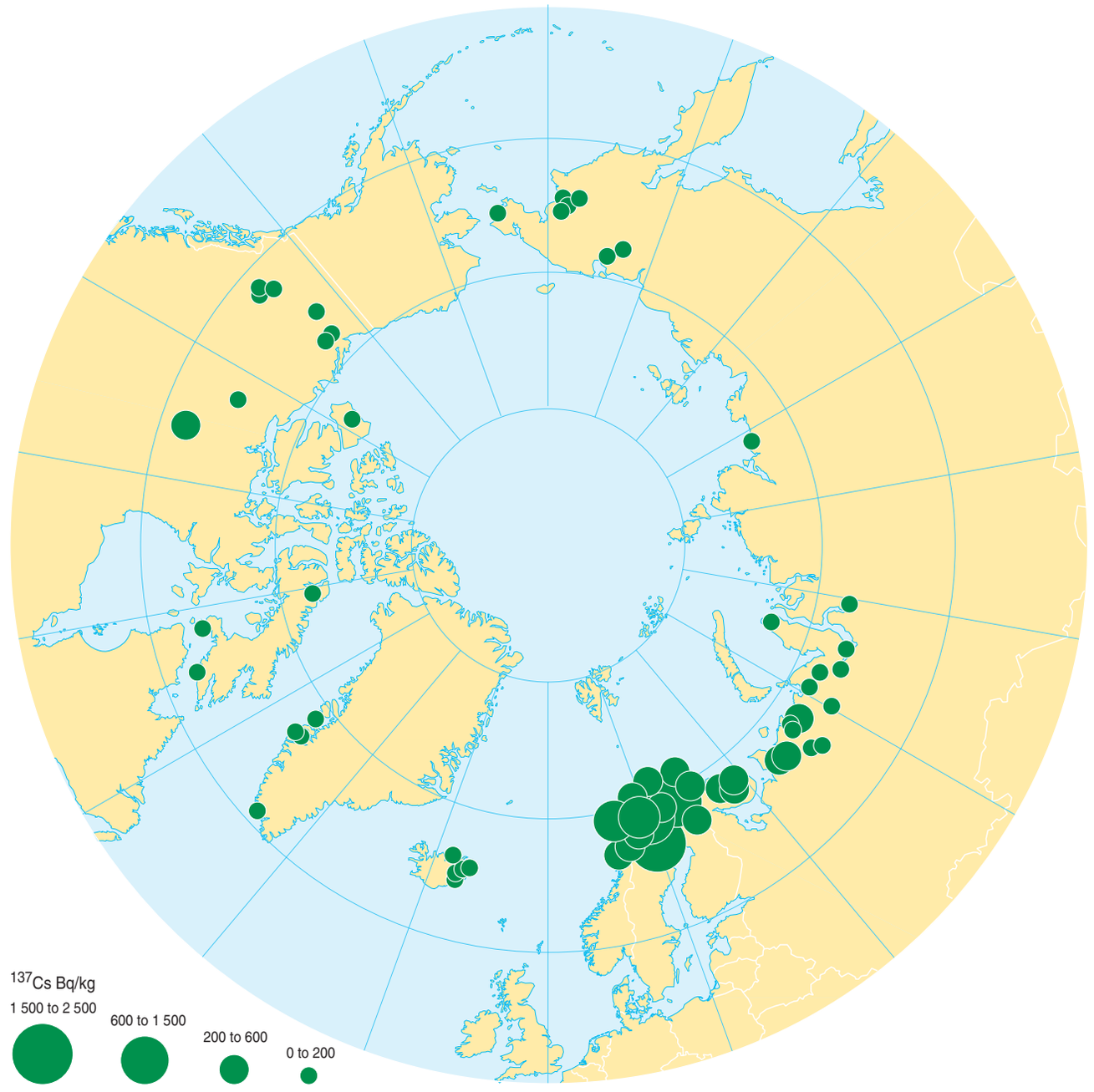


Figure 8·26. Average activity concentrations of ^{137}Cs in reindeer meat after 1990.

1996), and 7.2 ± 1.3 years in Arctic Norway. The decrease was slowed by the atmospheric tests carried out by France and China during the late 1970s. In 1986, a clear increase in reindeer meat contamination was observed in Arctic Russia, Sweden, Norway and Finland due to the Chernobyl accident. Five to ten years after the Chernobyl accident, ^{137}Cs activity concentrations in reindeer have stabilized and are expected to decrease more slowly in the future. In recent years, the highest ^{137}Cs activity concentrations in reindeer meat have been observed in the western part of Russia and in Norway, Sweden and Finland (Rissanen and Rahola 1996). In contrast, ^{137}Cs activity concentrations are lowest in the eastern part of Russia, in Canada, Greenland and Iceland (Figure 8·26). In Canada, after 1986, ^{137}Cs activity concentrations were generally about 100 Bq/kg. For Iceland, data from the 1990s are available showing ^{137}Cs activity concentrations in the order of 10 Bq/kg; such low levels are due to reindeer feeding almost entirely on herbaceous vegetation which is less highly contaminated than Cladonia lichen species (Palsson *et al.* 1994).

8.3.2.2.3. **Freshwater ecosystems**

One of the most mobile radionuclides from global fallout entering freshwater systems is ^{90}Sr. Unlike many other radionuclides, including ^{137}Cs, it is not significantly retained by soils.

Measurements of ^{90}Sr in Russian river water have been carried out since the beginning of the 1960s. The levels peaked around 1964 and have generally declined since then.

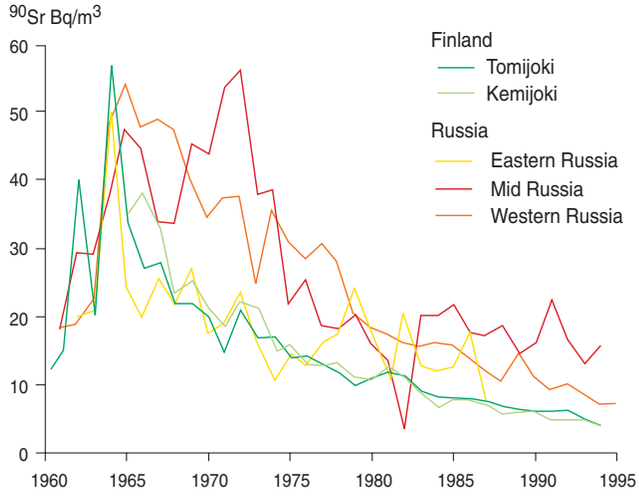


Figure 8·27. Changes with time in activity concentration of ^{90}Sr in Russian and Finnish rivers.

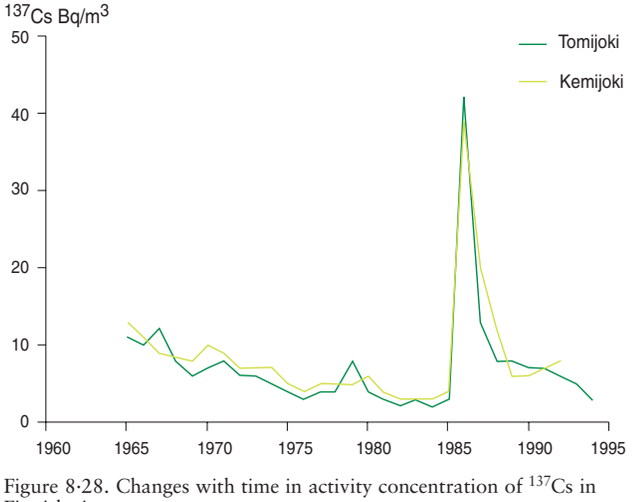


Figure 8·28. Changes with time in activity concentration of ^{137}Cs in Finnish rivers.

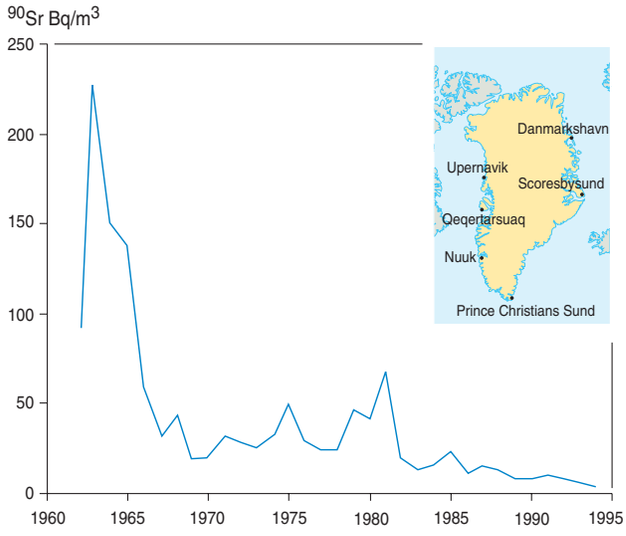


Figure 8·29. Changes with time in average activity concentration of ^{137}Cs in drinking water in Greenland.

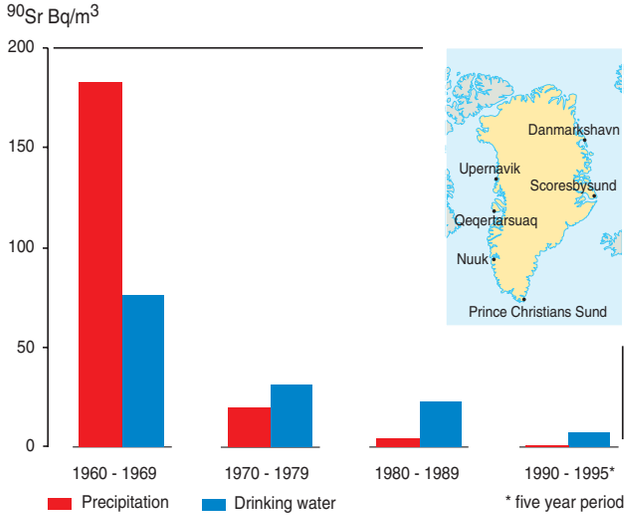


Figure 8·30. Ten-year averages of ^{90}Sr activity concentrations in drinking water and precipitation in Greenland.

The sources of the contamination of Russian river water is partly direct deposition of global fallout from the atmosphere and partly runoff of previously deposited global fallout from catchments. A third source is waterborne discharges from nuclear facilities, such as Mayak in the Urals. The major discharges of ^{90}Sr from Mayak to the Ob river system occurred around 1950. However, data on the levels of ^{90}Sr in the Arctic parts of the Ob river from that time are not available. From 1964 to 1994, ^{90}Sr activity concentrations in both Russian and Finnish river water decreased by a factor of ten (Figure 8·27). By comparison, ^{137}Cs activity concentrations in Finnish river water (Figure 8·28) (Salo *et al.* 1966-1996a, 1966-1996b) decreased from 20 Bq/m^3 in 1964 to 1 Bq/m^3 in 1985. The Chernobyl accident increased ^{137}Cs activity concentrations to 40 Bq/m^3 in 1986 but, by 1993, they had decreased to 3.4 Bq/m^3. This clearly demonstrates that ^{137}Cs activity declines more rapidly in river water than ^{90}Sr because of the higher particle reactivity of Cs.

Drinking water has been analysed in Greenland for ^{90}Sr since 1962 (Figure 8·29). The drinking water has been collected from six locations: Danmarkshavn, Scoresbysund, Prins Christiansund, Nuuk (Godthåb), Qeqertarsuaq (Godhavn) and Upernavik. The drinking water in Greenland is mostly derived from ice and snow for the northern localities. At the southern locations, surface water plays a greater role. Figure 8·30 shows 10-year mean values of ^{90}Sr in precipita-

tion and drinking water collected at these six locations in Greenland. Local variations in the concentrations exist. The highest drinking water activity concentrations of ^{90}Sr were found in the south at Prins Christiansund and the lowest in the northwest at Qeqertarsuaq and Upernavik. It is interesting to note that the ratio between ^{90}Sr in drinking water and precipitation has been increasing with time, illustrating the contribution of ^{90}Sr from previous precipitation (in ice) to the drinking water. This contribution is, as would be expected, highest at the northern stations.

Data for a few fish species have been selected to illustrate the levels and trends in ^{137}Cs activity concentrations in Arctic freshwater fish. The species from lakes in Finnish Lapland (Jokelainen 1965, Kolehmainen *et al.* 1966, Rissanen unpubl.) selected as examples were pike, whitefish, perch and trout (Figure 8·31). The first species represents predatory fish, and the last a non-predatory species consuming bottom fauna, plankton or both. The two intermediate species are partially predatory. The uptake of radionuclides by lake fish depends on hydrology and lake type. Cs-137 uptake, particularly for the non-fish-eating species, increases from eutrophic to oligotrophic lakes. The lakes in Finnish Lapland and in Arctic Scandinavia are dysoligotrophic and oligotrophic and differences in the measured ^{137}Cs activity concentrations in fish among the lakes were not substantial.

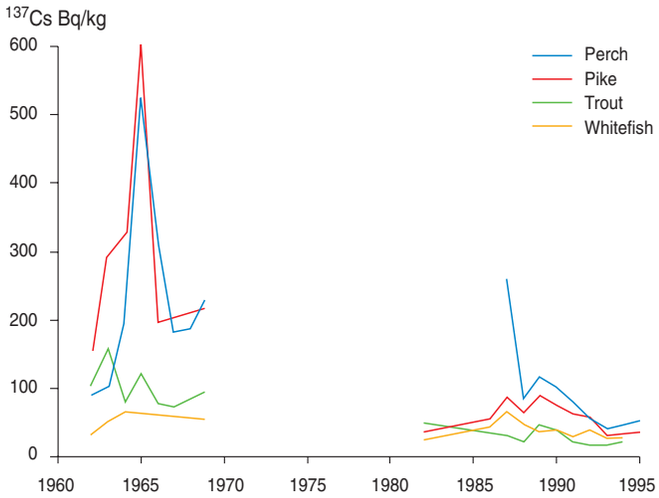


Figure 8·31. Changes with time in activity concentration of ^{137}Cs in freshwater fish in Arctic Finland.

In several studies in other areas in Finland that received higher deposition from the Chernobyl accident, the maximum transfer of Chernobyl radionuclides to fish occurred within the first three years for most species with ^{137}Cs peaks in the plankton-feeding fish occurring before those in predatory fish such as pike.

8.3.2.3. Marine ecosystems

8.3.2.3.1. Seawater

Monitoring of radioactive contamination of the western Arctic seas of Russia has been carried out since the early 1960s. The highest ^{90}Sr activity concentrations in the seas were reported in 1963-1964 and were: 85 Bq/m^3 in the Kara Sea, 52 Bq/m^3 in the Laptev Sea, 22 Bq/m^3 in the East Siberian Sea and 26 Bq/m^3 in the Chukchi Sea. ^{90}Sr activity concentrations in Arctic Seas have mainly been influenced by global fallout from the testing of nuclear weapons in the atmosphere.

Cs-137 activity concentrations, on the other hand, have largely been determined by discharges of this radionuclide

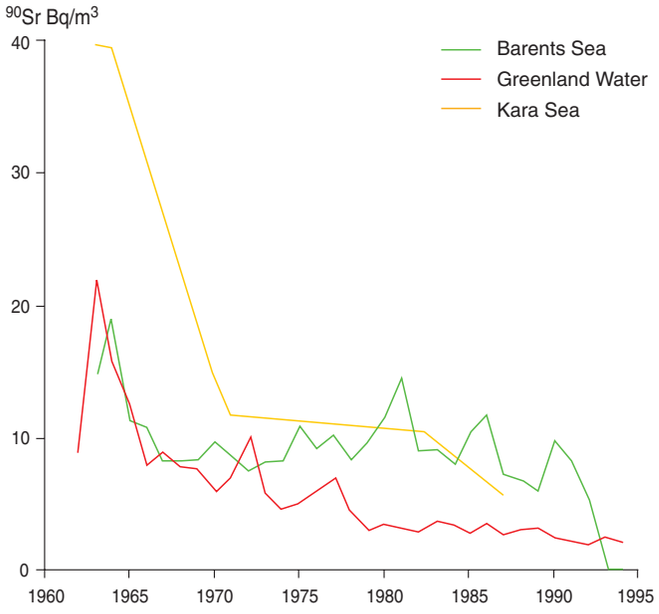


Figure 8·32. Changes with time in ^{90}Sr activity concentrations in surface seawater from Greenland waters and the Barents and Kara Seas.

from European reprocessing plants, particularly from Sellafield. These discharges peaked in the mid-1970s and were evident in the Barents Sea around five years later as shown in Figure 8·9.

Strontium-90 activity concentrations in surface seawater from Greenland during the last 35 years have been decreasing with an observed effective half-life of about 13.5 years (Aarkrog 1995). This decrease is probably representative of the Arctic Ocean as a whole (Figure 8·32). The highest activity concentrations around Greenland occur in the East Greenland Current reflecting surface seawater concentrations in the Arctic Ocean that are higher than those in the North Atlantic.

Since the early 1970s, ^{137}Cs has been measured in surface seawater collected around Greenland. In contrast to ^{90}Sr, no significant decrease in ^{137}Cs activity concentrations has been observed. This is due to the input of ^{137}Cs from the (1974-1982) liquid discharges to the Irish Sea from Sellafield in the UK. The Chernobyl accident in 1986 also added ^{137}Cs to the Arctic Ocean. As is the case of ^{90}Sr, the highest ^{137}Cs activity concentrations have been observed in the East Greenland Current. The ^{137}Cs activity concentrations in surface seawater along the East Greenland coast are about two times higher than those measured along the west coast of Greenland.

8.3.2.3.2. Fish and marine mammals

Samples of marine fish, seals and whales have been collected in Greenland and Icelandic waters since the early 1960s and have been analysed for ^{137}Cs (Figures 8·33 - 8·35). A slight decline has been observed in ^{137}Cs activity concentrations in all such samples. Contamination of fish and marine mammals was similar. Present ^{137}Cs activity concentrations are about 0.2-0.5 Bq/kg.

8.3.3. Human wholebody measurements

The ^{137}Cs content of Arctic human populations is significantly influenced by their consumption of locally produced food products, particularly intake of reindeer meat. Significant amounts of ^{137}Cs may, however, also be contributed by consumption of mushrooms and freshwater fish from, e.g., oligotrophic lakes.

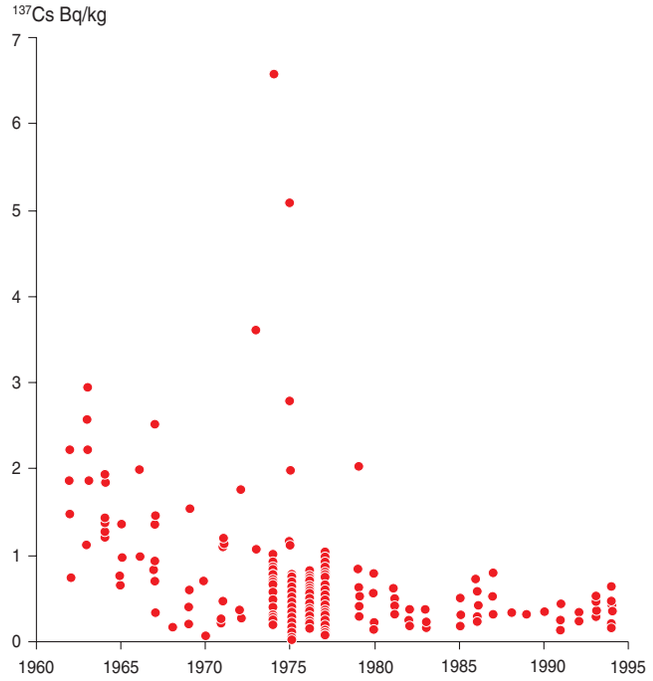


Figure 8·33. Changes with time in ^{137}Cs activity concentration in marine fish from Greenland waters.

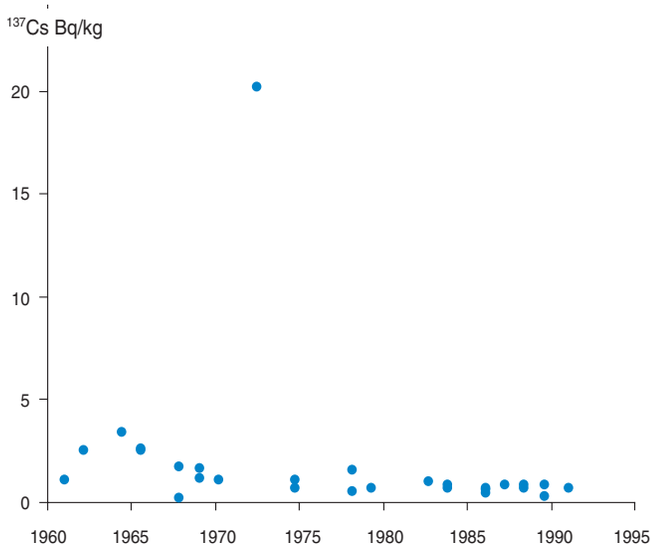


Figure 8·34. Changes with time in ^{137}Cs activity concentration in whales from Greenland waters.

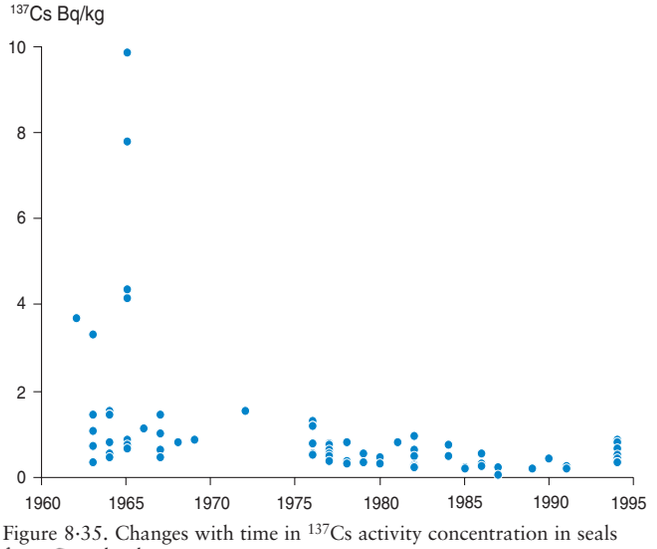


Figure 8·35. Changes with time in ^{137}Cs activity concentration in seals from Greenland waters.

Maximum ^{137}Cs activity concentrations in reindeer meat (Figure 8·25) and in reindeer herders from various Arctic regions (Figures 8·36 and 8·37) were reached in the middle of the 1960s. Average wholebody concentrations have decreased gradually over the following 20 years by a factor of 3 to 7. The peak in ^{137}Cs wholebody concentrations after 1986 is due to fallout from the Chernobyl accident. The highest body burdens of ^{137}Cs were observed in reindeer herders living on the Kola Peninsula in western Russia. Except where affected by Chernobyl fallout, lower body burdens were found in Finnish (Rahola *et al*. 1993) and Norwegian reindeer herders (by a factor of 2-3) and in the far eastern part of the Russian Arctic (by a factor of 1.5-10). The difference between Nordic and western Russian herdsmen is probably due to the greater consumption of less-contaminated, imported food by the former group. In eastern Russia, the lower levels are due to lower ^{137}Cs contamination of this area by global nuclear weapons fallout.

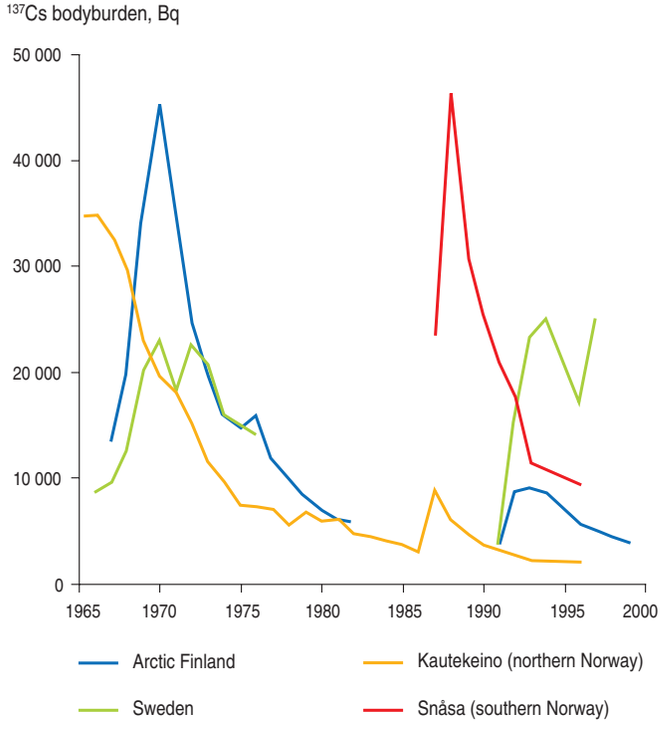


Figure 8·36. Changes with time in ^{137}Cs wholebody measurements of reindeer herders in northern and central Norway, Arctic Finland, and Sweden.

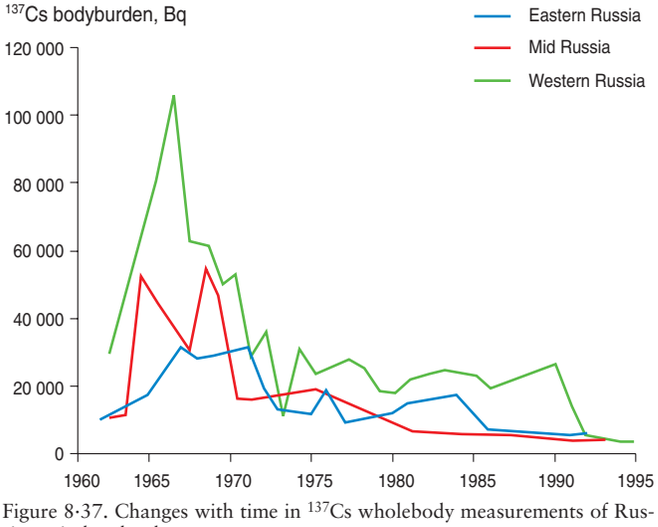


Figure 8·37. Changes with time in ^{137}Cs wholebody measurements of Russian reindeer herders.

8.3.4. Summary

Radionuclide activity concentrations in air and precipitation have closely reflected the rates of emission of radionuclides into the atmosphere from above-ground nuclear weapons tests, with identifiable peaks in the period when most tests were conducted, or associated with specific events, such as vented underground nuclear tests or accidental releases. Radionuclides are accumulated in terrestrial ecosystems and water bodies, and the rates of decline in contamination levels in biota in both types of ecosystem are slower than those in the atmosphere.

In general, radionuclide contamination levels in terrestrial biota have consistently been higher than those in marine biota. Within the terrestrial environment, the highest activity concentrations have been found in products harvested from natural or semi-natural ecosystems, followed by products produced by extensive farming in semi-natural ecosystems. For representative foodstuffs, the order of contamination generally decreases as follows:

> reindeer, mushrooms, freshwater fish > lamb meat, goat cheese > potatoes, vegetables >> marine fish, whale and seal meat.

In most Arctic areas, levels of radionuclide contamination in terrestrial biota reached a maximum in the second half of the 1960s due to global fallout from nuclear weapons tests. The geographical distribution of fallout reflects the patterns of precipitation for much of the Arctic. Fennoscandia and western Russia were also affected by fallout from the Chernobyl accident. In parts of Norway and Sweden, peak radiocaesium activity concentrations in terrestrial biota due to Chernobyl fallout attained values similar to those during the period of atmospheric nuclear weapons testing.

The highest radiocaesium activity concentrations in the terrestrial environment have usually been found in components of natural or semi-natural ecosystems, especially lichen and mushrooms, due to the high rate of interception or uptake of radiocaesium by these organisms. These high contamination levels are then transferred up the food chain and are especially reflected in the meat of Arctic reindeer or caribou, which largely depend on lichen for winter fodder. Similarly, in animals which consume mushrooms, such as moose, lamb or cattle, high levels of radiocaesium can be found in the autumn. During the late 1960s, ^{137}Cs activity concentrations in reindeer/caribou meat varied over the Arctic. In Alaska, Canada, Greenland and the Asian part of Russia, such activity concentrations were up to 1000 Bq/kg, whereas in the northern part of the European continent, they were up to 2000-3000 Bq/kg. After the mid-1960s, ^{137}Cs activity concentrations in reindeer meat decreased with an observed effective ecological half-life of about 5-10 years until the Chernobyl accident in 1986.

In all Arctic countries, population groups with high intakes of reindeer meat exist. Wholebody measurements on some of these groups have shown average body burdens of ^{137}Cs up to 50 000 Bq in the late 1960s. The observed half-lives of wholebody ^{137}Cs have been shorter than for reindeer meat since the food consumption patterns of some of these population groups have changed since the 1960s with an increasing portion of their food originating from agricultural ecosystems.

In the marine environment, the highest levels of radiocaesium contamination were found in the North European seas in the late 1970s and early 1980s due to releases from the Sellafield reprocessing plant. During the late 1960s, ^{90}Sr activity concentrations in seawater were about 20-30 Bq/m^3. No measurements of ^{137}Cs in seawater from that period are available, but by applying the established ratio between ^{137}Cs and ^{90}Sr in nuclear weapons fallout, activity concentrations of ^{137}Cs in seawater can be estimated to have been up to 50 Bq/m^3. Equivalent and higher values were also found in the Barents Sea in the early 1980s due to Sellafield releases. Radionuclide activity concentrations in marine biota have consistently been low in the Arctic compared to the levels found in terrestrial and freshwater biota. Even during the heaviest fallout period in the mid-1960s, average ^{137}Cs activity concentrations in marine fish or marine mammals in the North European seas did not exceed 10 Bq/kg.

In addition to contamination from the sources affecting large areas of the Arctic, some places have locally enhanced contamination due to specific sources or events. The most significant of these are at Thule, Greenland, following a nuclear accident, and Novaya Zemlya in northwest Russia where terrestrial or underwater nuclear testing occurred and where solid nuclear waste has been dumped along the eastern coast. At these latter sites, contamination levels may be orders of magnitude above the average for the Arctic. It should be stressed that such elevated contamination only occurs within a few kilometers of these sources.

8.4. Individual doses to man estimated from environmental measurements

Arctic populations are exposed to ionizing radiation from three major sources. Exposures from natural sources deliver the main part of the dose. Medical exposures are a second potentially important source but are not dealt with here. Exposures arising from the exploitation of nuclear energy, for military as well as peaceful purposes, is the third major source. Some of these exposures show considerable variation with time and location, whereas others, such as those from natural sources, are generally less variable. Exposures to anthropogenic sources depend on source characteristics, ecological factors, which influence rates of transfer, and on the living habits of the exposed population.

8.4.1. Natural radiation

External exposures from natural sources (see also section 8.2.1.1)

According to UNSCEAR (1993) (Table 8·1) the typical individual doses received from external exposures from natural sources, including cosmogenic radionuclides, are 0.39 mSv/y from cosmic rays and 0.46 mSv/y from terrestrial gamma rays, giving an aggregate individual dose rate of 0.85 mSv/y. In regions with elevated natural radiation, these values can reach 2.0 mSv/y and 4.3 mSv/y, respectively, giving a total of 6.3 mSv/y. The AMAP radioactivity assessment group has assumed that members of the Arctic population receive, on average, an external exposure from natural sources equal to that considered typical by UNSCEAR of 0.85 mSv/y. A Nordic study (Christensen *et al.* 1990) describing the variation in exposures from natural sources among the Nordic countries, estimated that it varied between 0.5 and 1.0 mSv/y.

Internal exposures from natural sources

According to UNSCEAR (1993), the average annual effective internal dose from natural radionuclides (mainly ^{40}K and radionuclides from the ^{238}U and ^{232}Th series) is 0.23 mSv/y. In addition, there is an average dose of 1.3 mSv/y received from radon, thoron and their decay products. In regions with elevated natural radiation, these annual internal

doses may be as high as 0.6 and 10 mSv, respectively. For the Arctic regions, the assessment group has used the average internal dose estimated by UNSCEAR of 0.23 + 1.3 = 1.53 mSv/y. Christensen *et al*. (1990) have reported the variation in internal exposure in the Nordic countries to be 0.5-4 mSv/y, due mainly to differences in radon exposures in dwellings. Consumption of reindeer meat and fish contributes to the internal exposure from natural sources because they contain enhanced ^{210}Po concentrations (Tracy *et al*. 1995, Woodhead 1982, Pentreath 1988). Consumers of large amounts of reindeer meat may receive a dose from ^{210}Po in the order of 10 mSv/y.

8.4.2. Radionuclide contamination

To estimate the doses from past and present radioactive contamination in the Arctic, information on population characteristics, including living habits such as occupation, housing and food consumption, for the average populations of the eight Arctic countries have been collated. Furthermore, 'selected' groups have been defined, denoting groups of people in specific Arctic areas expected to receive higher doses from the intake of radiocaesium. It should be stressed here that the various selected groups are not necessarily comparable, some are relatively large whilst others are small, some are based on actual populations whilst that for Greenland is hypothetical. Estimates of external dose have, where appropriate, taken account of the shielding effect of different types of dwellings.

8.4.2.1. Information base for individual dose estimates

The following sections provide information on which the dose estimates for average and selected groups within the Arctic are based. In some instances, the population characteristics differ from those given in the chapter 5. This results from the need to estimate individual doses at an early stage in the preparation of the radioactivity assessment chapter. It should, however, be noted that much of the information used here was specifically collated for the purposes of radiological dose estimation and were reviewed at an earlier stage in the assessment process by the AMAP assessment groups responsible for chapters 5 (*Peoples of the Arctic*) and 12 (*Pollution and Human Health*).

Finnish Lapland

The total number of inhabitants in Finnish Lapland was 202 400 in 1992, and they inhabit an area of 93 057 km^2. Of these 94 700 live in towns and 107 700 in rural areas. Of the rural population, 10.8% depend on agriculture for their livelihood, 23.7% on industry and 62.7% on services.

The reindeer-herding area is approximately 114 000 km^2 and there are 7100 reindeer owners. Approximately 4000 of the reindeer owners are Saami people who depend mainly on reindeer husbandry. The total Finnish reindeer herding area is divided into 56 herding co-operatives, 40 of which are in Lapland. Annually 130 000-150 000 reindeer are slaughtered producing 3 million kg of meat. Reindeer herders and their families consume 130 000 kg of meat themselves. Reindeer herding is economically most significant in the Saami district, where over one third of the herd is managed by about 1500 of the owners.

About 21% of the population are younger than 15 years, 67% are between 15 and 64 years and the remaining 12% are 65 years and older. In rural areas, the houses are mainly single family-houses, largely constructed of wood with some of brick. In urban areas there are also large apartment houses. Of the Lapland population, 20.8% live in large apartment houses and 79.2% in single-family-houses or attached houses. The reindeer herders all live in single-family houses.

The adult Saami reindeer herders (males and females) in the Saami district form the selected group for the Finnish Arctic area for the discussion of dietary intake of radionuclides. The dietary habits of both the average population and the selected group have changed during the past 40 years. In particular, milk and grain consumption has declined and vegetables and fruit consumption has increased in both groups. The consumption of reindeer meat by the selected group has remained fairly constant during the past three decades but is lower than in the early 1960s. Reindeer meat consumption among the average population is small. The consumption of freshwater fish by the selected group has decreased since the 1960s whilst the consumption of marine fish (e.g., salmon from the rivers discharging into the Arctic Ocean) increased during the 1990s (Jokelainen 1965, Hasunen and Möttönen 1976, Hasunen *et al*. 1976, Hälinen and Sikkilä 1993, Laitinen *et al*. 1996, Rissanen unpubl.).

Greenland (Kalaallit Nunaat)

Greenland is one of the three countries within the Kingdom of Denmark. The total number of people in Greenland on 1 January, 1994, amounted to 55 419. Of this population, 87% were born in Greenland and 13% outside, mainly in Denmark. The mean life-expectancy for women is 68 years and 61 for men. About 80% of the population live in the towns, where large apartment houses were built in the 1960s and 1970s. In the villages, where about 20% of the population live, most residences are small single-family houses.

Approximately 20% of the population is dependent on hunting activities, primarily for seal. There are wild as well as domestic reindeer in Greenland (P. Hansen, Grønlands Hjemmestyre, pers. comm. 1996). They are mainly found between 64°N and 69°N on the west coast of Greenland. The number of wild reindeer varies considerably and is presently (1996) very low. The exact number is not known, but it is estimated to be around 15 000-20 000. In earlier years the number approached 100 000 and the annual hunting was 5000-6000 animals. In 1995, only 2000 wild reindeer were shot, with a live weight of about 300 000 kg. Hunting takes place mainly during August and September. The total number of domestic reindeer is about 5000-6000 and approximately 30% of the stock are slaughtered annually, corresponding to about 200 000-300 000 kg live weight. Domestic reindeer are mostly slaughtered from the middle of August to the middle of September, however, winter slaughtering also occurs.

Sheep farming is carried out in southwestern Greenland between 60°N and 62°N. The number of sheep is about 17 900, and about 13 300 lambs (live weight ca. 485 000 kg) are slaughtered each year in September-October. The demand for lamb exceeds local production and lamb is imported from Iceland and New Zealand.

The selected group for Greenland is a hypothetical group that is assumed to consume only reindeer meat instead of imported meat and lamb. Furthermore, it is assumed that the group consumes freshwater fish rather than marine fish and locally collected berries rather than imported fruit.

Northern Canada

The total population of the Canadian Arctic is ca. 70 000 (Yukon 23 075; Northwest Territories 46 000, plus a small number in northern Quebec) (Canadian Encyclopaedia 1985). The indigenous population of the Canadian Arctic is approximately 36 000 (cf. chapter 5). This latter estimate compares

well with the sum of the estimated rural (non-urban) population of the Yukon (8300) and that of the Northwest Territories (24 000) (Canadian Encyclopaedia 1985) allowing for some of the urban population comprising indigenous people.

The dietary intake for the average Canadian Arctic resident is estimated on the basis of a weighted combination of the average (dominantly southern) Canadian consumption pattern obtained to represent the non-indigenous population of the Arctic and the estimated dietary intake for indigenous residents after Coad (1994). The average Canadian consumption figures are based on a 'Nutrition Canada' survey of food consumption patterns (B. Tracy, pers. comm.). The different consumption patterns for individuals in distinct age groups (in the range 12-64) have been averaged to derive these values. The consumption of beverages have been included with drinking water and the consumption of miscellaneous foodstuffs has been ignored. This has been done in full recognition that this community does not represent a group of individuals with common habits analogous to a critical group that might be selected for assessing the acceptability of individual doses arising from a practice. The dietary characteristics of the Old Crow selected group have been determined from recent estimates described in Coad (1994). There is insufficient historical information to estimate any changes in dietary patterns that might influence the retrospective individual dose reconstruction attempted here.

Northern Russia

The total population of the Russian Arctic in the AMAP area numbers about 2 million persons; 1.7 million living in cities, towns and settlements (the urban population) and 0.3 million living in villages and small settlements (the rural population) (see chapter 5). For the purposes of average effective dose estimation, the population of the Russian Arctic is divided into two groups defined on the basis of geographical and social criteria, including dietary habits.

The selected group comprises reindeer-breeders and their families. This group is represented mainly by indigenous peoples and has a population size of about 100 000.

In winter, reindeer are pastured in the forest-tundra zone, where there is an opportunity to find shelter from bad weather and pastures are rich with lichens. In spring, herds move to the northern meadow pastures in coastal areas. In autumn, the reindeer again return to the forest tundra to the south. Along with reindeer-breeding, the indigenous people of the North hunt game and fur-bearing animals, gather berries, mushrooms and fish. The majority of reindeer-breeders live with their families in settlements and take turns in tending the herds. In summer, they move from place to place, together with their families, on the tundra.

About 70% of the reindeer-breeders have permanent winter dwellings which are standard wooden houses. In permafrost areas, houses are mounted on piles with boarding along their perimeter.

In settlements on the coast, indigenous people undertake hunting of sea animals and fish. However, meat producing reindeer-breeding farms also exist. No more than 10-15% of the indigenous people of northern Russia live directly on the coast.

Because of different levels of radioactive contamination of the western and eastern parts of the Russian Arctic, the data on ^{137}Cs and ^{90}Sr activity concentrations in food products, estimates of intake, and internal doses of inhabitants are considered separately for these two regions.

The average population comprises 1700 000 inhabitants of large ports and industrial cities including Murmansk, Archangelsk, and Norilsk, and about 200 000 rural inhabitants of villages and small settlements who are not involved in reindeer breeding . The urban population consumes mainly imported food products. The dietary habits of this segment of the population are, therefore, similar to those of inhabitants of other large Russian cities. The dietary habits of the rural population is more diverse but still includes a relatively large proportion of imported products.

Food habits of the average and selected groups of the Russian Arctic population have been carefully studied since the 1960s to allow internal dose estimation. These studies included specially-devised population surveys combined with wholebody counting (Ramzaev *et al*. 1993). The results of these studies have been used for the calculation of radionuclide intake by inhabitants and subsequent internal dose estimation.

Northern Norway

The total number of inhabitants in Arctic Norway was 379 461 in 1990, and they inhabit an area of 95 489 km^2. At that time, the number of Saami was about 50 000, constituting 13.2% of the total population.

The Saami People in Norway live along the Norwegian–Swedish border from as far south as Engerdal municipality in Hedmark county to the border with Russia in the north. The largest Saami population is found in Troms and Finnmark counties, especially in the municipalities of Karasjok, Kautokeino, Tana, Nesseby and Porsanger. Like other people, the Saami have gradually migrated to the villages and cities but many people still live in the countryside. The numbers of Saami in the three Arctic counties of Norway was recently estimated to be 21 689 in Finnmark, 12 457 in Troms, and 3239 in Nordland.

The general population of Arctic Norway does not differ very much from the population of the rest of Norway. The selected group for Arctic Norway comprises males and females associated with reindeer-breeding.

Alaska

The USA definition of Arctic Alaska includes approximately 51 930 people (Boedeker 1991) and an area of approximately 700 000 km^2.

For rural Alaska as a whole, fish are 59% by weight of the total subsistence harvest; for certain regions, fish comprise over three-quarters of the harvest. Except for the northern and northwestern regions of Alaska, fish represent the majority of the subsistence harvest by weight. Salmon are the most important species, but whitefish, burbot, and trout species are significant as well. Several species of shellfish, including clams and crab, are also important to subsistence harvests.

For coastal communities in Arctic Alaska, marine mammals are a critical and highly valued resource. They are also the reason that many communities are located on the coast, as migrating marine mammals pass within close range. In northern and northwest Alaska (the Arctic Slope, the NANA and Bering Straits regions, i.e., the areas adjacent to the Beaufort, Chukchi and northern Bering Seas) marine mammals account for 42% of the subsistence harvest, or 99 kg/y/cap. In these regions, the primary species taken are bowhead whales, beluga whales, walrus, bearded seals, ringed seals, other species of seals, and polar bear.

For both coastal and inland communities, terrestrial mammals form a significant part of the subsistence harvest. Caribou are the primary species hunted, although moose, Dall sheep, muskox, brown and black bear, and a variety of smaller mammals are also taken. Reindeer herding, introduced in the early 20th century, continues in some areas of Alaska. At

present, caribou populations throughout the Arctic region are high, as are harvest levels. Due to their migrations, caribou are hunted throughout the year in different communities, depending upon their local availability. Moose, sheep, muskox, and smaller mammals are available more consistently, although local preferences and government hunting regulations may restrict harvests.

Iceland

Iceland is the second largest island in Europe, located in the North Atlantic just south of the Arctic Circle. The total area of the country is 103 000 km^2 and the coastline, including fjords and inlets, is about 5960 km long.

Iceland is the most sparsely populated country in Europe with an average of 2.4 inhabitants per km^2. On 1 December, 1994, the total population of Iceland was 266 783, comprising 133 781 males and 133 002 females. Compared with neighboring countries, the Icelandic age distribution is relatively young. Thus, in 1994, around 10.8% of the population was aged 65 years and older, and 24.8% was aged 15 years or younger.

The Icelandic diet is typically western European in most respects. Nevertheless, it retains some characteristics of a subarctic region, making it somewhat unique among European nations. Fish, meat and milk are traditionally the main foods produced in Iceland and this local production affects what people consume. Icelanders consume more fish than any other nation in Europe (73 g/d/cap) and, in general, foods of animal origin constitute a large proportion of the Icelandic diet. Young people consume the least amount of fish while people over fifty consume the most (Steingrimsdottir *et al*. 1991). Grains (with the exception of small amounts of barley and oats) and fruit are not grown in Iceland and vegetable production is mostly limited to potatoes and greenhouse plants. Consequently, Icelanders eat less vegetables than inhabitants of most European countries, even through the economy allows substantial import of such foods. The diet of Icelanders is unusually rich in protein and fat (protein: 17.4%; fat: 41%) but unlike the situation in many other western countries, this is not entirely a modern development since the traditional diet of the nation had some of the same characteristics in earlier times.

Arctic Sweden

The total number of inhabitants in the two northernmost counties in Sweden, Västerbotten (55 401 km^2) and Norrbotten (98 911 km^2), was 259 775 and 267 648, respectively, in 1994. The rural population constitutes 26% and 19% of the total in each county, respectively.

The total Saami population in the main reindeer-herding area of Sweden is about 17 000 (10 000 in Norrbotten, 5000 in Västerbotten, and 2000 in Jämtland). Of these, about 2500 are reindeer owners.

From the mid-1980s, about 80 000 to 100 000 reindeer have been slaughtered annually, producing 2 million kg of meat per year. About 15-20% of the meat production is derived from the slaughter in September, 40-50% during November-December, and 30-40% during January-April. During other periods of the year the slaughter is relatively small.

Cattle, primarily for milk production, constitutes a significant source for the regional supply in both Västerbotten and Norrbotten counties. Beef production, and in Västerbotten also pork production, is substantial. Although lambs are found in these areas, meat production is small and only corresponds to a few percent of the beef production. The contribution from other livestock is of even less importance.

The critical group in northern Sweden belongs to the reindeer-herding population with relatively high consumption of reindeer meat and freshwater fish from the region.

Diet intakes by Arctic populations

Tables 8·11 and 8·12 (next page) show the annual mean consumption rates for the average populations and some selected groups within some of the Arctic countries. The selected groups are generally assumed to be those with the highest consumption of reindeer meat.

8.4.2.2. **External and internal doses to humans**

External exposure from anthropogenic sources

The effective dose commitment due to radionuclides produced in atmospheric nuclear testing is 1 mSv for external exposure in the northern temperate zone (40-50°N) (UNSCEAR 1993). Half of this is due to ^{137}Cs, the other half comes from short lived radionuclides such as ^{95}Zr, ^{106}Ru, ^{54}Mn and ^{95}Nb. The assessment group have assumed that the total external dose is proportional to ^{137}Cs deposition from nuclear weapon testing, which UNSCEAR estimates to be 5.2 kBq ^{137}Cs/m^2 in the 40-50°N latitude belt. The effective dose commitment in the 60-70°N latitude belt, where the majority of the Arctic population lives, assuming that an integrated deposition density of ^{137}Cs in that latitude belt is $\approx$3 kBq/m^2 (UNSCEAR 1993), can then be calculated as $3/5.2 \times 1$ mSv $\approx$ 0.6 mSv. External dose will decrease from south to north due to decreasing amounts of precipitation and thus of global fallout. Furthermore, the regions which received high amounts of Chernobyl fallout will show enhanced external doses. It has been estimated that the external individual dose commitments from Chernobyl to the Norwegian, Swedish and Finnish average populations were 1.0, 0.6 and 1.7 mSv, respectively (Strand *et al*. 1987, Moberg 1991, STUK-A74 1991). The external doses received by the Arctic populations are, in general, less than the national means. The external doses from the Chernobyl accident should be added to the 0.6 mSv from global fallout to obtain the total external exposure from anthropogenic sources. It is assumed that external exposures to members of the Arctic population will be in the range 0.6-1 mSv from all anthropogenic sources.

Internal doses from anthropogenic sources

Internal doses arise from ingestion and inhalation of radionuclides. The inhalation pathway is of minor importance for most radionuclides and will not be considered here. According to UNSCEAR (1993), the most important contributor to internal doses from anthropogenic sources is ^{14}C from ingestion that will deliver an individual effective dose commitment of 2.6 mSv to the average member of the world population over the next many thousands of years. This dose will be the same irrespective of where people live. However, only 10% of the dose will be delivered before the year 2200. For current generations, the most important anthropogenic contributors to internal dose are ^{90}Sr and ^{137}Cs. These radionuclides have been the most intensively studied in the environment.

From 1995 onward, it was assumed that activity concentrations of both radionuclides in the Arctic diet would decrease with an effective ecological half-life of ten years, and the future time integrals were calculated according to this model. However, some measurements (in Canada for instance) suggest that the effective ecological half-life of ^{137}Cs in the human Arctic population may be significantly lower than ten years (Walton 1995). In contrast, Scandinavian studies indicate effective ecological half-lives in species of

Table 8·11. Contemporary annual mean consumption of foodstuffs (kg/y/cap) for average populations of the Arctic.

	Alaska	Finnish Lapland	Greenland	Iceland	Northern Canada	Northern Russia	Norwegian Lapland	Swedish Lapland
Diet – Mainly imported products								
Cow milk[a]	≈80	230	92	176	86	70	206	158
Other milk[a]								
Cow cheese[a]		12	5.8	14		5	9.5	12.7
Other cheese[a]							6.6	0.4
Grain products as flour	≈60	72	57	57	51	100	59	60
Potatoes	≈40	60	28	50	30	70	60	84
Leafy vegetables (e.g. cabbage)		10				20		
Root vegetables (e.g. carrots)	≈50	30	11.6	26	61	10	39	
Fruit (imported)		60	36	57		30	65	55
Pork		60		15		15[b]		
Beef	≈40		36		41	15[b]	40	55
Poultry and eggs		10				15		
Diet – Mainly local products								
Lamb			9.1	24			6	1
Marine fish (incl. fish from fish farms)		14	23	27		10	20	15
Deer							1.5	0.2
Elk or moose		1.5			28	3[c]		1.6
Reindeer or caribou	≈10	1.0	3.8	0.04			0.7	0.2
Freshwater fish (wild)	≈10	4		0.6	12	5	1.2	1
Berries (wild)		10				10	2.6	4.5
Mushrooms (wild)		1.4				5	0.2	2.8
Various game (e.g. birds, hare, etc.)	≈5	0.3		1.1	4.7			0.05
Seal	≈5		30		6.2			
Whale			2.3					
Drinking water	≈700	700	550	600	700	600	600	550

a. Milk and milk products are locally produced in some countries.
b. For ^{90}Sr, intake calculated as 0.25 kg bone.
c. For ^{90}Sr, intake calculated as 0.05 kg bone.
STUK-94A -A62 -A78, Risø Reports 1986-1993, Steingrimsdottir *et al*. 1991, Coad 1994, Tracy *et al*. 1995, Strand *et al*. 1987, Moberg 1991, Johanson and Bergström 1993.

Table 8·12. Contemporary annual mean consumption of foodstuff (kg/y/cap) for selected groups[a].

	Alaska	Finnish Lapland	Greenland	Iceland	Northern Canada	Northern Russia, West	Northern Russia, East	Norwegian Lapland	Swedish Lapland
Diet – Mainly imported products									
Cow milk[b]		150	92					213	158
Other milk[b]						10	10		
Cow cheese[b]		15	5.8					5	
Other cheese[b]									
Grain products as flour		80	57			180	180		50
Potatoes		50	28			15	15		55
Leafy vegetables (e.g. cabbage)									55
Root vegetables (e.g. carrots)		20				5	5		5
Fruit (imported)		10				20	20		55
Pork									
Beef		9						11	3
Poultry and eggs		5				5	5		
Diet – Mainly local products									
Lamb									2
Marine fish (incl. fish from fish farms)		5				40	40	14	
Deer									
Elk or moose		0.1			364	91[c]	91[c]		6
Reindeer or caribou	≈300	70	82					120	60
Freshwater fish (wild)	≈10	20	23		12	70	70	5	15
Berries (wild)		20	10			15	15	12	3
Mushrooms (wild)		1				20	20		
Various game (e.g. birds, hare, etc.)	≈10	1			12	10	10		2
Seal			30						
Whale					0				
Drinking water	≈700	700	550		700	700	700		550

a. These groups are not necessarily directly comparable (see text).
b. Milk and milk products are locally produced in some countries.
c. For ^{90}Sr, intake calculated as 1.5 kg bone.
STUK-94A -A62 -A78, Risø Reports 1986-1993, Steingrimsdottir *et al*. 1991, Coad 1994, Tracy *et al*. 1995, Strand *et al*. 1987, Moberg 1991, Johanson and Bergström 1993.

mushrooms and for moose which approach the physical half-life of ^{137}Cs of 30 years. Although the effective ecological half-life obviously represents an important uncertainty it does not greatly influence the overall dose commitment because the major part of the dose was received between 1960 and 1994.

Future doses, i.e. doses from 1995 and onward, contribute less than 10% to the total dose commitment from ^{137}Cs in Arctic diet. Where data were missing, they were estimated from neighboring periods by interpolation or extrapolation. The composition of the diet may have changed throughout the years and this was taken into account in Arctic Finland and Greenland. Otherwise, a constant composition of the diet throughout the period was assumed. The data used in these calculations are based upon national survey results referred to in the references. However, the national data used

Table 8·13. Estimated dietary intakes of ^{137}Cs by the average population (kBq over period).

Period	Alaska	Arctic Finland	Greenland	Iceland	Northern Canada	Arctic Russia	Arctic Norway	Arctic Sweden
1950-59	–	35	10	–	158	18	38	18
1960-64	–	39	11.1	–	176	20	42	20
1965-69	–	42	13.5	–	175	25	42	20
1970-74	–	22	4.5	–	112	15	19	11
1975-79	–	11.1	3.7	–	91	10.3	34	7
1980-84	–	8.2	4.1	–	63	7.7	5.6	6.5
1985-89	–	10.9	4.0	–	32	10.3	16.3	31
1990-94	–	5.9	1.8	–	21	6.5	13.3	22
1995 → ∞	–	17	4.3	~45[a]	61	19	40	64
1950 → ∞	–	191	57	–	890	132	247	199
mSv	–	2.5	0.74	3.3[b]	11.6	1.7	3.2	2.6

a. From Dahlgaard (1994a).
b. Estimated from Greenland dose assuming proportionality between doses in Iceland and Greenland.

Table 8·14. Estimated dietary intakes of ^{137}Cs by selected groups (kBq over period).

Period	Alaska	Arctic Finland	Green-land	Iceland	Northern Canada	Arctic Russia, West	Arctic Russia, East	Arctic Norway	Arctic Sweden
1950-1959	330	560	142	–	2100	860	240	580	300
1960-1964	370	620	158	–	2300	960	270	640	330
1965-1969	420	1000	192	–	2300	1050	330	1050	630
1970-1974	240	490	64	–	1500	630	240	700	145
1975-1979	58	210	53	–	1200	340	155	460	57
1980-1984	30	149	58	–	820	240	133	260	59
1985-1989	15	195	57	–	420	250	154	560	310
1990-1994	8	146	26	–	270	160	151	270	250
1995 → ∞	23	420	61	–	790	450	440	800	720
1950 → ∞	1390	3800	810	–	11700[a]	4900	2100	5300	2800
mSv	18	49	10.5	–	152	64	27	69	36

a. Individuals in the selected group in Canada eat 13 times more reindeer meat than the average Arctic residents. Hence, the dose is estimated to be 13 times higher than that of the average, because reindeer is the dominating factor for ^{137}Cs in the diet for Arctic Canada.

Table 8·15. Estimated dietary intakes of ^{90}Sr by the average population (kBq over period).

Period	Greenland	Arctic Russia
1950-1959	1.9	2.4
1960-1964	2.1	2.6
1965-1969	1.3	3.6
1970-1974	0.66	1.5
1975-1979	0.41	1.0
1980-1984	0.28	0.54
1985-1989	0.20	0.39
1990-1994	0.14	0.20
1995 → ∞	0.39	0.58
1950 → ∞	7.4	12.8
mSv	0.21	0.36

Table 8·16. Estimated dietary intakes of ^{90}Sr by selected groups (kBq over period).

Period	Green-land	Arctic Russia, West	Arctic Russia, East	Alaska
1950-1959	3.2	31	20	0.5
1960-1964	3.5	35	23	0.6
1965-1969	2.2	47	26	0.6
1970-1974	0.8	18.8	14.5	0.2
1975-1979	0.5	12.3	8.9	0.2
1980-1984	0.3	4.6	3.5	0.1
1985-1989	0.2	2.6	2.2	0.04
1990-1994	0.2	1.0	0.9	0.02
1995 → ∞	0.5	2.8	2.5	0.07
1950 → ∞	11	155	102	2.3
mSv	0.3	4.4	2.8	0.06

to calculate the values shown in the tables have been critically evaluated within the AMAP consultation process. Furthermore, it should be noted that, due to changes in diets in Finland and Greenland, it is not possible to achieve the total intakes of radionuclides shown in Tables 8·13 to 8·16 from simple multiplication of the amounts of food (Tables 8·11 and 8·12) by the activity concentrations in the diet (tables in Annex). In some cases, activity concentrations in the various diet components differ between the average population and the selected group. In such cases, a separate activity concentration table has been provided for the relevant selected group in the annexed tables.

The estimated dose from ^{137}Cs in the diet to the average population varies between 0.74 mSv (Greenland) and 11.6 mSv (Canada), and for the selected groups range from 10.5 mSv (Greenland) to 152 mSv (Canada). The time-integrated concentrations of ^{137}Cs in the human body given in Table 8·53 may be used to estimate doses to members of the selected groups in three of the Arctic countries. According to UNSCEAR (1993) a wholebody time integral of 1 Bq y/kg for ^{137}Cs delivers a dose of 2.4 nSv. Hence, the doses to the selected groups in Arctic Finland, Arctic Norway and Arctic Russia become 21, 32 and 56 mSv, respectively. These doses may be compared with those calculated from diet intakes in Table 8·14. It appears that the doses based on wholebody measurements are about half of those derived from dietary data for Finland and Norway but, in the case of Russia, the two estimates are similar.

Wholebody measurements on Arctic population groups have also been carried out in Sweden and Canada. In Sweden, the dose estimate based on wholebody measurements was a factor of two lower than that obtained from dietary studies, i.e., in agreement with the observations in Finland and Norway. In the case of Canada, wholebody measurements on the selected group from Old Crow, where people are assumed to consume 1 kg reindeer/d/cap did not validate the high dose calculated from the dietary estimate. The wholebody dose was, in this case, an order of magnitude lower than the dietary dose estimate. However, if the Canadian wholebody measurements were considered representative for the average Canadian Arctic population, the agreement between wholebody measurements and dietary estimates of dose was satisfactory.

The relatively good agreement achieved for some populations between assessments of dose based on diet and wholebody measurements may, in some cases, be fortuitous. The deposition of ^{137}Cs associated with the accident at Chernobyl was small in Arctic Finland, Arctic Norway and Arctic Russia. In areas of Norway and Sweden which were more highly contaminated by fallout from the Chernobyl accident, the comparison between wholebody ^{137}Cs content and deposition is influenced by countermeasures. Thus, the weighted averages based on dietary estimates are higher than those obtained from wholebody measurements. Accordingly, considerable overestimation of doses to reindeer herding groups of the Saami population occur in some areas of high deposition of Chernobyl caesium if estimated on the basis of diet. However, the dose estimates match reasonably well in areas where Chernobyl deposition was relatively low. Assessments based on dietary data must, therefore, be made with caution and should consider possible influences of the provision of information and the adoption of countermeasures. Any changes in the use of local food products or changes in the activity concentrations in actually consumed foodstuffs need to be taken into account in dose estimation based on dietary habits. Further procedures are needed to provide a basis for taking account of such effects in dose assessments.

A tendency to overestimate doses when using dietary calculations is often observed. This may be attributed to several

Table 8·17. Estimated dose commitments to the average Arctic resident from anthropogenic releases of radionuclides.

Dose	Radiation type	Dose commitment, mSv
External	^{137}Cs	0.3-1
	Other radionuclides[b]	0.3-1
Internal	^{90}Sr	0.1-0.4
	^{137}Cs	0.7-12
Ingestion and inhalation	^{14}C	2.6[a]
	Other radionuclides[b]	0.2-0.6
Total dose commitment		4-18

a. Infinite dose commitment, which is delivered over several thousands of years. The dose from ^{14}C is minor during the period in which the dose commitment from ^{90}Sr and ^{137}Cs is received.
b. The contribution from other radionuclides were estimated from UNSCEAR (1993).

Table 8·18. Estimated annual dose rate to the average Arctic resident from natural sources.

Source	Annual dose rate, mSv/y[a]
External	0.5-1 (0.85)
Internal	0.5-4 (1.53)
Total annual dose rate	1-5 (2.4)

a. Values in parentheses are the global average annual dose rates according to UNSCEAR (1993).

Table 8·19. Estimated dose commitments to members of selected Arctic population groups associated with previous anthropogenic releases of radionuclides in the environment.

Dose	Radiation type	Dose commitment, mSv
External	^{137}Cs	0.3-1
	Other radionuclides	0.3-1
Internal	^{90}Sr	0.1-4
	^{137}Cs	10-150
Ingestion and inhalation	^{14}C	2.6[a]
	Other radionuclides	≈1
Total dose commitment		14-160

a. Infinite dose commitment.

Table 8·20. Estimated annual dose rate to members of selected Arctic population groups from natural sources.

Source	Annual dose rate, mSv/y[a]
External	0.5-1
Internal (incl. ^{210}Po)	0.5-10
Total annual dose rate	1-11

factors. Probably the most important of these is an overestimate of the amounts of food actually consumed. Other factors are loss of ^{137}Cs during food preparation and a tendency for measurements to represent maximum rather than average activity concentrations. It should also be noted that wholebody measurements can be biased because the individuals participating in wholebody measurements may not be representative of the population group within which they reside.

Doses from ^{90}Sr in diet were calculated for a few countries only. The effective dose commitment from ^{90}Sr varied from 0.21 mSv to 0.36 mSv for the average population, and for selected groups between 0.06 and 4.4 mSv. The doses from ^{90}Sr are thus significantly lower than those from ^{137}Cs, particularly for the selected groups. Reindeer meat is the predominant source of ^{137}Cs in the Arctic diet. ^{90}Sr activity concentrations in reindeer meat are typically two orders of magnitude lower than ^{137}Cs activity concentrations. For intakes of ^{137}Cs by dietary component, see the annex to this chapter.

Doses to average members of Arctic populations from anthropogenic and natural sources are summarized in Tables 8·17 and 8·18. The doses are given as the estimated range between minimum and maximum average values for the eight Arctic countries. The lowest anthropogenic doses are those for Greenland and the highest are for Canada. The lowest doses from natural sources occur in Iceland. Compared with UNSCEAR's data for doses from natural sources, the Arctic anthropogenic doses correspond to 2-7 years additional background radiation.

The dose estimated for the selected groups in the Arctic are shown in Tables 8·19 and 8·20. The highest dose commitments are those received by the selected group in Canada, which was assumed to have an extremely high consumption of reindeer meat (1 kg/d/cap). The Canadian group receives a dose from ^{210}Po in reindeer meat of about 10 mSv/y. The dose commitment from ^{137}Cs is calculated to be approximately 150 mSv for the same selected group, which corresponds to the dose received in 15 years from naturally occurring ^{210}Po in the consumed reindeer meat (Beak 1995). However, it is conceivable that there are other population groups in Arctic areas of other countries consuming similarly large amounts of reindeer/caribou meat and having correspondingly high exposures to natural and anthropogenic radionuclides. Compared to the global average doses received from natural sources, the selected Canadian group receives a dose from natural sources which is four times higher. The dose commitment received from ^{137}Cs by this selected group is 40-50 times higher than that of the average Arctic resident.

The individual mean doses for the average populations (Tables 8·13 and 8·15) in the various Arctic countries have been multiplied by the corresponding population numbers to obtain the collective doses for ^{137}Cs and ^{90}Sr from dietary intake. The estimated collective dose from ^{137}Cs intake is 9000 manSv, and that from ^{90}Sr is estimated, from the Russian and Greenland data, to be 25% of the ^{137}Cs dose, i.e. (2000 manSv. The remaining collective dose is estimated to be 6000 manSv, assuming that the anthropogenic mean external dose and the internal doses from radionuclides other than ^{90}Sr and ^{137}Cs (also excluding ^{14}C) is 1.5 mSv (Table 8·19) and that the total Arctic population is 3.8×10^6. Hence the total collective dose is estimated to be 17 000 manSv. This value may be compared with an estimate for global fallout only of 15 000 manSv (see section 8.5.1.1). These two estimates are in fairly good agreement particularly considering that the dose from Chernobyl is included in the 17 000 manSv. In this calculation, it was assumed that the individual dose commitment to the average population in Alaska from ^{137}Cs is 2 mSv and to the Faeroese population 3.3 mSv. The range of individual dose commitment (4-17 mSv) received by the average Arctic inhabitant exceeds the average dose commitment for individuals living in the north temperate zone (40-50°N) which according to UNSCEAR (1993) is 4.4 mSv.

8.4.3. Intakes of ^{137}Cs through various dietary components

Figures 8·38, 8·40, 8·42, 8·44, 8·47, and 8·49 depict the yearly intakes of ^{137}Cs in various dietary components of the average members of the populations of Arctic Finland, Greenland, Canada, Russia, Norway and Sweden, respectively. Figures 8·39, 8·41, 8·43, 8·45, 8·46, 8·48, and 8·50 depict the yearly intakes of ^{137}Cs in dietary components of the selected populations of Arctic Finland, Greenland, Canada, eastern Russia, western Russia, Norway and Sweden, respectively.

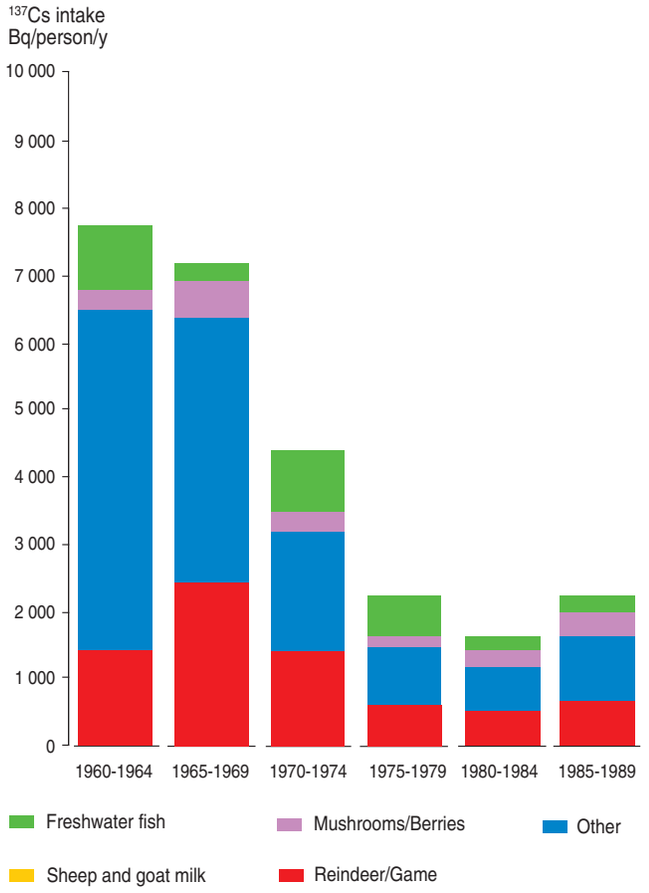


Figure 8·38. Yearly intake of ^{137}Cs from various dietary components by the average population of Arctic Finland.

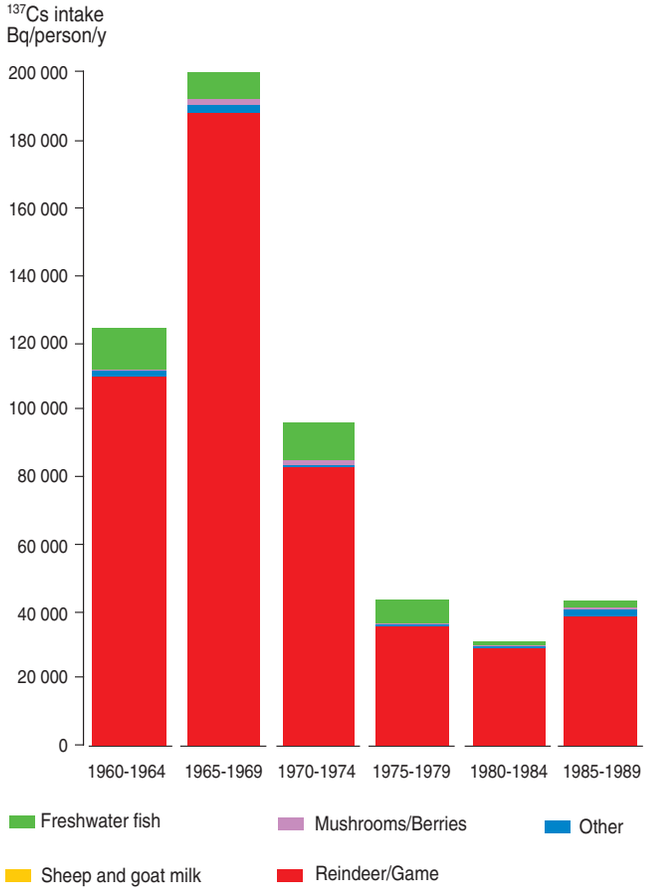


Figure 8·39. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of Arctic Finland.

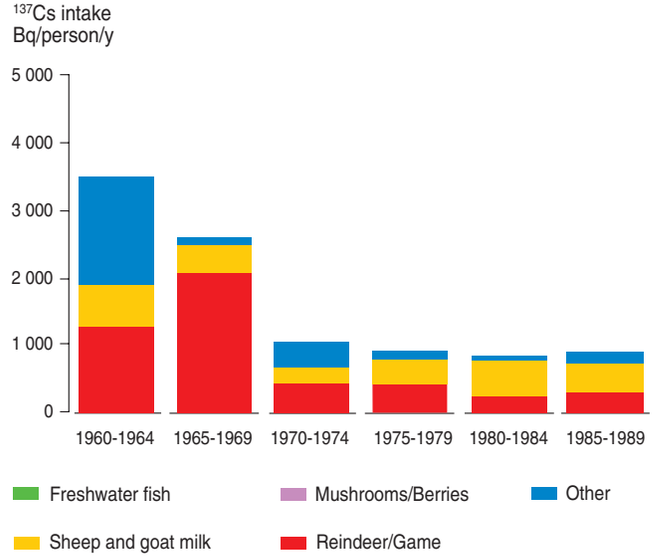


Figure 8·40. Yearly intake of ^{137}Cs from various dietary components by the average population of Greenland.

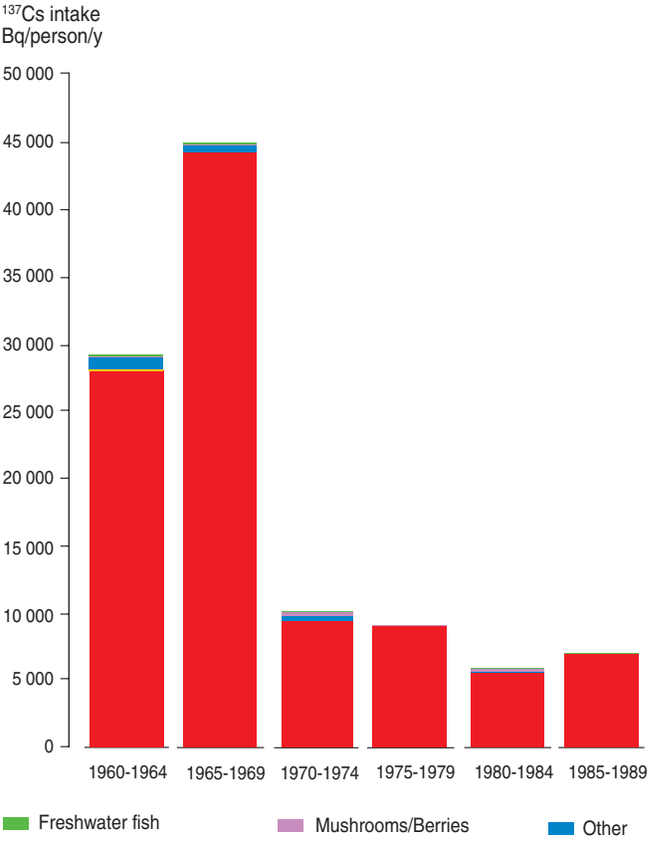


Figure 8·41. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of Greenland.

Variations in sources of ^{137}Cs intake

Figures 8·39, 8·41, 8·43, 8·45, 8·46, 8·48 and 8·50 for the selected Arctic groups show, in all cases, the same feature: reindeer/caribou meat is, for these groups, the dominating source of dietary ^{137}Cs. In general, less than 10% of the ^{137}Cs comes from other dietary components, mostly from freshwater fish.

For average populations, the various dietary components contributing to ^{137}Cs intake vary considerably among Arctic countries. In Arctic Canada (Figure 8·42), reindeer/caribou has been the dominant source of ^{137}Cs in the diet. This is because the percentage of indigenous people

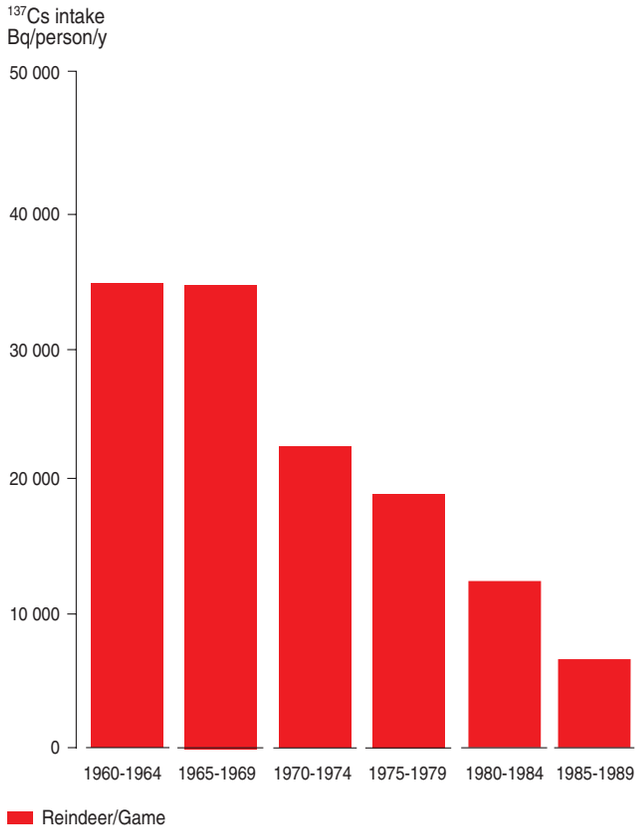


Figure 8·42. Yearly intake of ^{137}Cs from various dietary components by the average population of Arctic Canada.

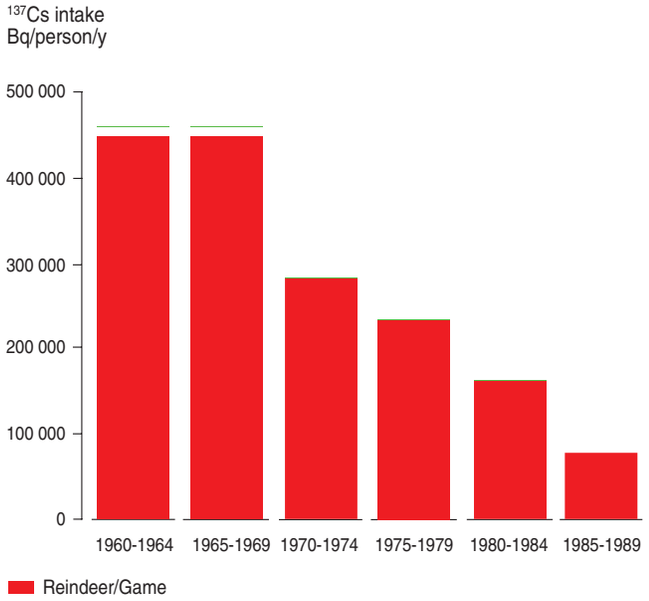


Figure 8·43. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of Arctic Canada.

(with high caribou consumption) in Arctic Canada is higher than in the other Arctic countries. For Arctic Finland (Figure 8·38), Arctic Russia (Figure 8·44) and Arctic Norway (Figure 8·47) agricultural products (i.e., milk, cereals, beef and pork) are important contributors to dietary ^{137}Cs. In Arctic Sweden (Figure 8·49), mushrooms and berries dominate for most years. In Greenland (Figure 8·40), lamb and reindeer are the most important sources of ^{137}Cs in the diet of the average population. Although 87% of the Greenland population is indigenous, reindeer is not as important to ^{137}Cs intake for the average population as it is in Arctic Canada, because Greenlanders prefer marine

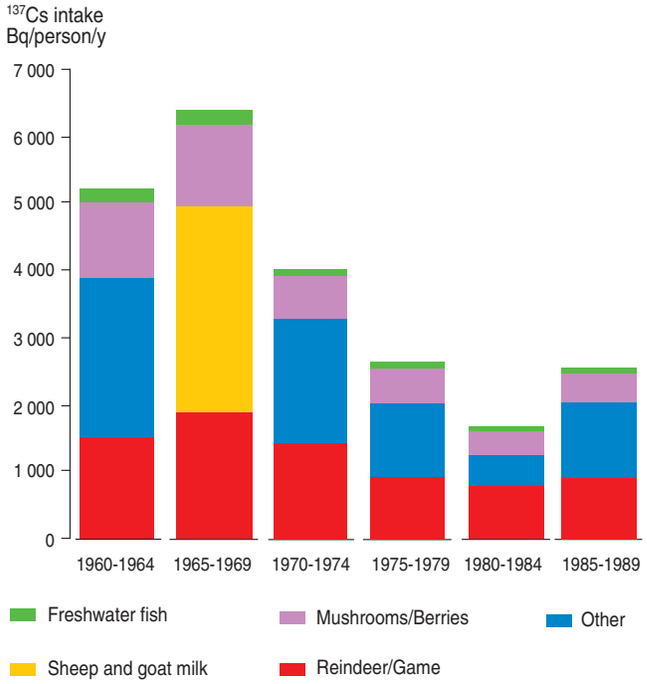


Figure 8·44. Yearly intake of ^{137}Cs from various dietary components by the average population of Arctic Russia.

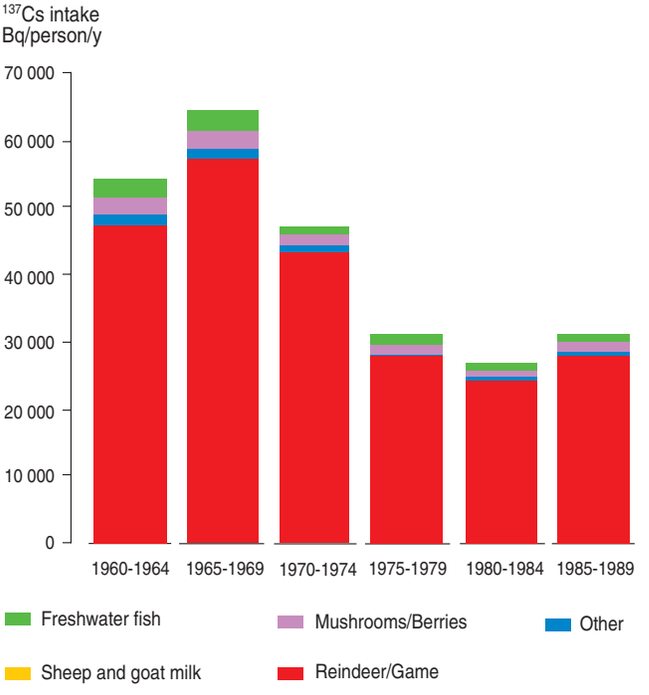


Figure 8·45. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of eastern Arctic Russia.

products such as seals and fish. Contributions of ^{137}Cs from freshwater fish are most important in Arctic Finland (Figure 8·38).

Temporal variations in ^{137}Cs intake

All the calculations give a maximum intake of ^{137}Cs with diet in the 1960s due to the peak in global fallout around 1962-1964. In some cases, the maximum occurred in the first half of the decade, in others in the second half. The Chernobyl signal was evident in 1985-1989 in all Arctic areas included in these calculations except Canada (Figures 8·42 and 8·43). In Greenland (Figures. 8·40 and 8·41) (and probably also Iceland) the signal was weak, but nonetheless present. The strongest response on the Chernobyl

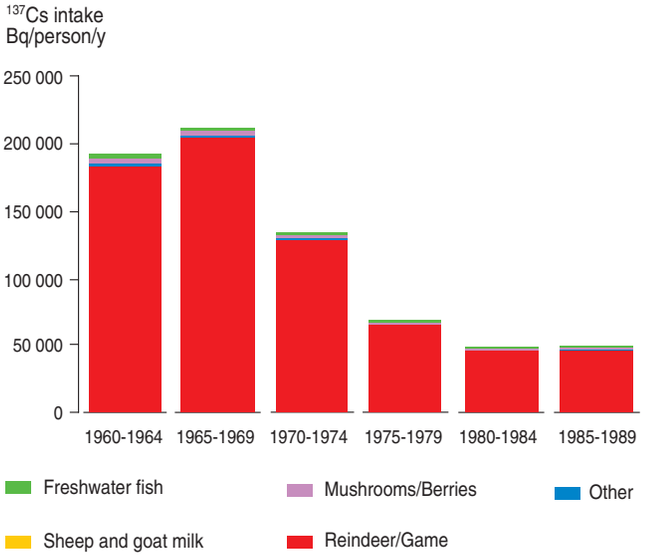


Figure 8·46. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of western Arctic Russia.

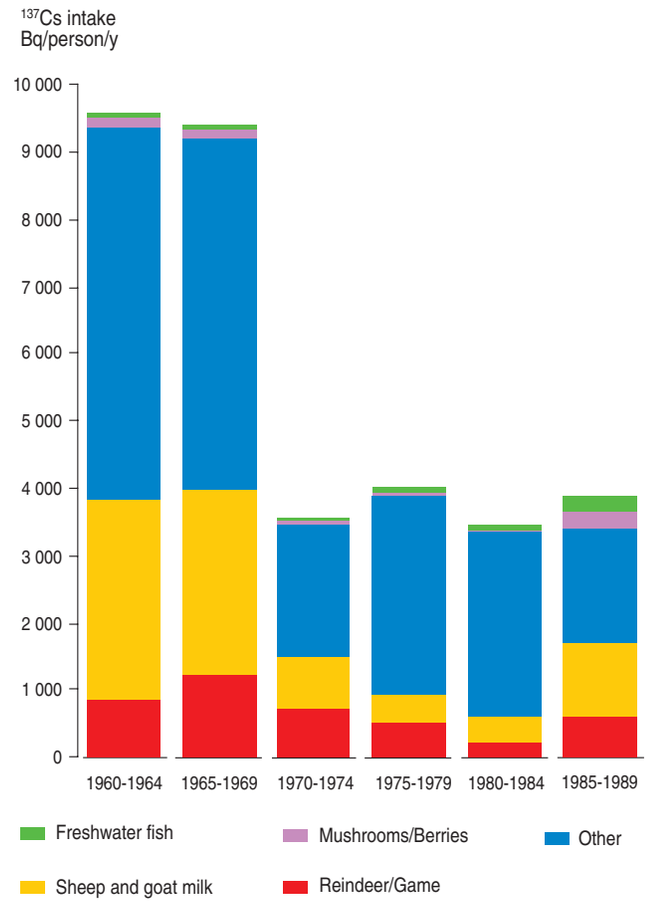


Figure 8·47. Yearly intake of ^{137}Cs from various dietary components by the average population of Arctic Norway.

fallout was seen in Arctic Sweden (Figures 8·49 and 8·50) where the estimated ^{137}Cs intake increased by a factor of 5 from 1980-1984 to 1985-1989.

Changes in the relative importance of dietary components with time

The relative importance of the various diet sources to ^{137}Cs intake by the Arctic average populations showed only minor changes throughout the years. In Arctic Sweden (Figure 8·49), however, the relative importance of mushrooms has been increasing since the 1960s.

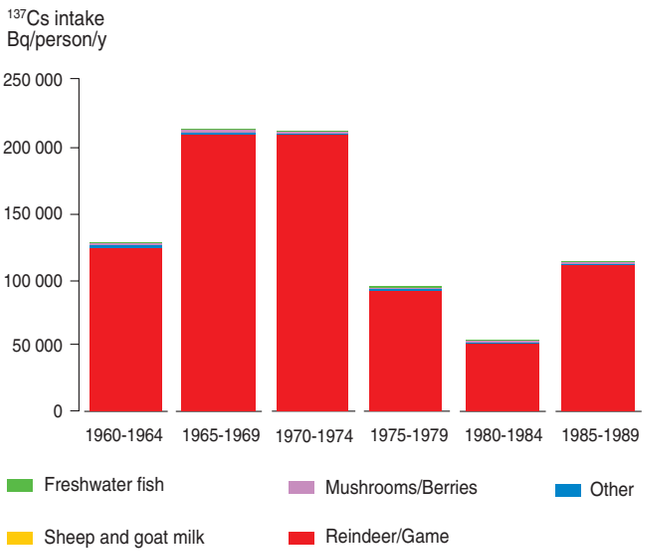


Figure 8·48. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of Arctic Norway.

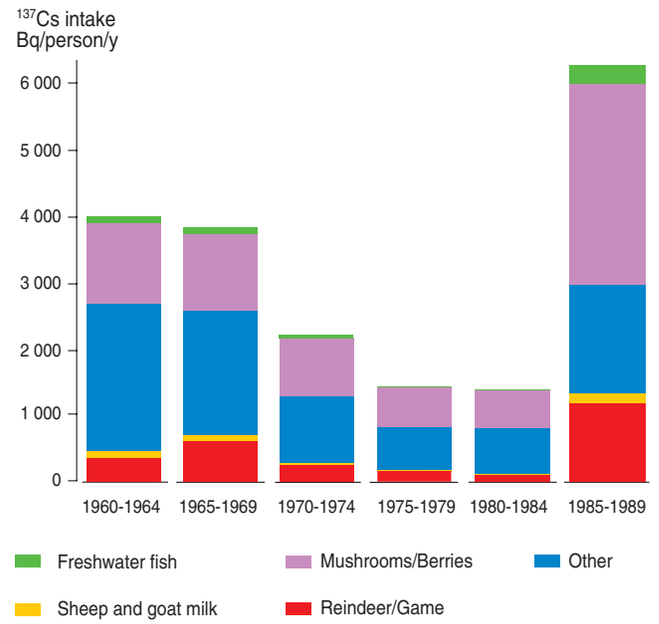


Figure 8·49. Yearly intake of ^{137}Cs from various dietary components by the average population of Arctic Sweden.

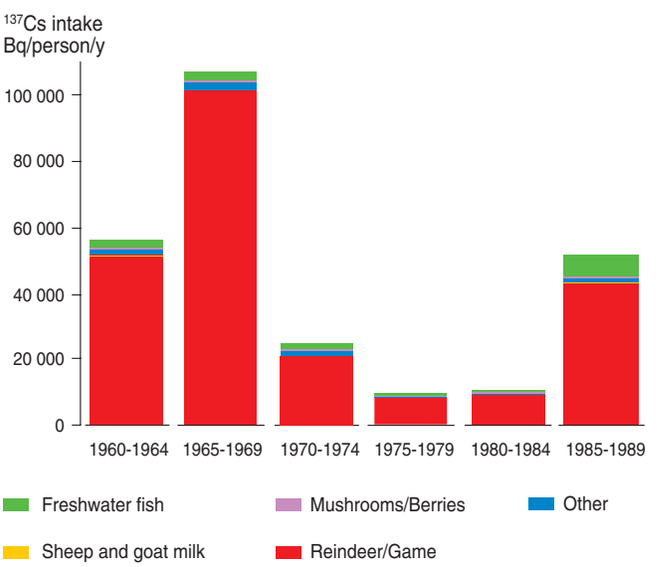


Figure 8·50. Yearly intake of ^{137}Cs from various dietary components by the 'selected' population of Arctic Sweden.

8.4.4. Summary

The doses to Man in the Arctic from natural and anthropogenic radiation derive from both external and internal sources. The major contribution to the average population is the dose from inhalation of naturally-occurring radon; the annual dose from radon is 0.5-4 mSv/y, corresponding to a lifetime dose of 30-300 mSv. Lifetime doses to present generations of the Arctic average population due to anthropogenic radionuclides vary between 2 and 15 mSv or about 5% of the dose from natural sources. The most important contribution to enhanced doses from anthropogenic radionuclides in the Arctic is the consumption of reindeer/caribou meat. Certain specific population groups, for instance those in Arctic Canada having extreme consumption rates of reindeer meat, i.e., of the order of 1 kg/d/cap, may get lifetime doses from ^{137}Cs of the order of 150 mSv. In addition, individuals in such population groups may receive annual doses from naturally-occurring ^{210}Po in reindeer meat of 10 mSv/y. It cannot be ruled out that there are small numbers of individuals within the 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.

This study shows that people eating marine products only, for instance fish and marine mammals, receive doses from anthropogenic radionuclides that are at least an order of magnitude lower than those to people consuming terrestrial products such as reindeer/caribou, freshwater fish and mushrooms. For this reason, individuals predominantly consuming seafood have lower dose rates and lifetime doses than average members of both the Arctic and northern hemispheric populations.

8.5. Source-related assessments of past and present releases

As discussed in section 8.2.3.7, two categories of source-related assessments need to be discussed in this document. These are:

- Previous and continuing releases of radionuclides to the environment from human activities.
- Potential, or possible future, releases of radionuclides to the environment resulting from human activities.

In this section of the document, previous and continuing releases from human activities are discussed. Such releases are of two distinct types: operational and accidental. Operational releases are those authorized in the licensing of practices and are generally routine releases from nuclear fuel cycle operations. Nuclear explosions intended both for the testing of nuclear weapons and for peaceful purposes are also classified for the purposes of this review as operational releases. Accidental releases are unintentional releases of radionuclides to the environment.

8.5.1. Nuclear explosions

The predominant releases of artificial radioactivity on a global scale have been derived from atmospheric testing of nuclear weapons. Thermonuclear weapons tests in the atmosphere have contributed the most to global fallout. Contamination resulting from underground nuclear weapons testing and underground civilian nuclear explosions is, from a global perspective, negligible. However, some underground explosions have resulted in local environmental contamination.

8.5.1.1. Atmospheric nuclear weapons tests

The three major sites for atmospheric testing of thermonuclear weapons have been Novaya Zemlya in the Arctic region of the former Soviet Union (FSU), Bikini and Eniwetok Islands (USA) in the Pacific Ocean and the Nevada test site (Figure 8·51). In addition, the FSU conducted tests at Semipalatinsk in Soviet Central Asia, China at Lop Nor in western China, France at Mururoa in the Pacific Ocean and the United Kingdom in Maralinga and the Monte Bello Islands, Australia, and the Christmas Islands in the Pacific. The USA, UK and FSU all discontinued atmospheric testing by 1962. Since 1980, there has been no atmospheric testing carried out by any country.

A total of 520 atmospheric explosions took place up to 1980 (UNSCEAR 1993). The total explosive yield amounted to 545 Mt (TNT equivalents): 217 Mt from fission explosions and 328 Mt from thermonuclear (fusion) explosions. According to UNSCEAR (1993), of the aggregate releases of radionuclides to the environment, fallout activity deposited close to the test sites accounts for 12%, tropospheric fallout, which is deposited in a band around the Earth at the latitude of the test site, for 10%, and global fallout, which is mainly deposited in the same hemisphere as the test site, for 78%.

As most test explosions have been carried out in the northern hemisphere, (see Figure 8·51), most of the radioactive contamination is found there. Contamination of Arctic regions is, in general, less than that of temperate areas. The AMAP GIS-based estimate of the total integrated deposition of ^{137}Cs on land north of 60°N (section 8.3), derived during the course of this assessment, yields a value of 35 PBq (corresponding to 18 PBq in 1995). The average ratio between ^{137}Cs and ^{90}Sr in global fallout is 1.6. Hence, the present inventory of global fallout of ^{90}Sr in the Arctic (60-90°N), estimated using the GIS approach, is approximately 11 PBq. Local fallout from thermonuclear tests tends to be minor compared with the global fallout because these tests were frequently conducted at high altitude in which much of the debris is injected into the stratosphere and the fireballs created by the explosions did not reach the ground. In the populated part of the Arctic, where the major food production occurs (i.e., the 60-70° N latitude band), the integrated deposition density of ^{90}Sr (Figure 8·52) is 1.7 kBq/m^2 and that of ^{137}Cs is 2.6 kBq/m^2 (UNSCEAR 1982). The equivalent value for deposition on land of ^{137}Cs using the GIS-based approach is 2.2 kBq/m^2. The UNSCEAR estimates of deposition densities have been used for dose estimation for the Arctic population from global fallout.

The transfer coefficients of UNSCEAR (see section 8.2.4) predict the doses to individuals at temperate latitudes from ingestion, inhalation and external exposure pathways, respectively. If the transfer coefficients in Table 8·21 are assumed to be also valid for the Arctic population, and if the integrated deposition densities given above for the Arctic are applied, average individual doses in the Arctic are, by simple multiplication, 0.1 mSv for ^{90}Sr and 0.4 mSv for ^{137}Cs. Taking the total population of the Arctic to be 3.8×10^6, the collective dose from ^{90}Sr and ^{137}Cs is, according to the UNSCEAR model, estimated to be approximately 2000 manSv. This dose would be very unevenly distributed. Population groups consuming local terrestrial products, such as reindeer/caribou meat, would receive significantly higher doses than those people living on imported products from temperate regions. In contrast, people consuming locally-produced marine products, such as fish and sea mammals, would receive lower doses than those consuming largely imported products. For instance, dietary studies in Greenland

Figure 8·51. Sites where atmospheric testing of nuclear weapons has taken place since 1945 (Zander and Araskog 1973).

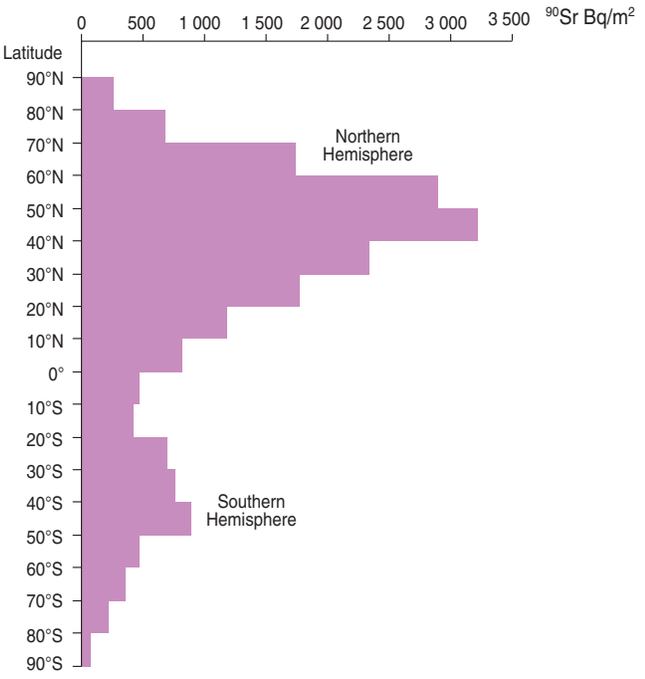


Figure 8·52. Integrated deposition density of ^{90}Sr (UNSCEAR 1993).

have shown that ^{137}Cs intake from the three types of diet: local terrestrial, imported, and local marine, vary in the ratio 20 : 2 : 1 (Aarkrog *et al.* 1963-1995).

The transfer coefficients estimated by UNSCEAR in Table 8·21 do not consider the enhanced intakes of ^{137}Cs from Arctic semi-natural and natural ecosystems. Therefore, the estimated doses calculated above are lower than the actual doses received by Arctic populations because reindeer meat is the main contributor to the ^{137}Cs dose. The total annual production of reindeer meat in the Arctic has been estimated by the assessment group to be 2.8×10^7 kg (Table 8·54). If the integrated transfer coefficient for reindeer meat is equal to the mean of that observed in this study for the Arctic countries of 10 kBq ^{137}Cs kg^{-1} y per kBq ^{137}Cs/m^2 (Table 8·51), the total intake of ^{137}Cs from reindeer becomes $2.8 \times 10^7 \times 10^4 \times 2.6 \approx 7.3 \times 10^{11}$ Bq ^{137}Cs (assuming an integrated deposition density of 2.6 kBq/m^2 of ^{137}Cs from global fallout in the Arctic). The dose intake factor is 13 nSv per Bq ^{137}Cs and the collective dose thus becomes 9500 man Sv. In other words, if we conservatively assume that all reindeer meat produced in the Arctic is consumed by Arctic residents (i.e., a consumption rate of 7.4 kg/y/cap), this would increase the estimated dose by 9500 manSv (i.e., to ≈12 000 manSv). The individual mean dose commitment from ^{137}Cs then becomes 3.2 mSv, which is eight-fold greater than that (0.4 mSv) estimated using the UNSCEAR methodology.

It is unlikely that all reindeer meat produced in the Arctic is consumed by the Arctic people. For instance, if the individual consumption data given in 8.4.2 are multiplied by the respective populations of the Arctic countries, the estimated consumption of reindeer meat by Arctic populations is only 1.5×10^7 kg/y which is approximately half the estimated annual production of reindeer meat of 2.8×10^7 kg/y. However, other terrestrial food products with enhanced ^{137}Cs levels are also consumed, such as freshwater fish, game, mushrooms and berries, and these will also contribute significantly to the dose. On these grounds, the collective dose from ^{90}Sr and ^{137}Cs in global fallout from atmospheric nuclear weapons testing to the Arctic population would more realistically be of the order of 10 000 manSv, i.e. five-fold greater than the UNSCEAR-based estimate.

Table 8·21. Transfer coefficients adopted by UNSCEAR (1993) for global fallout (see Figure 8·1).

		Ingestion		Inhalation		External exposure	
Transfer coefficient	Unit	^{90}Sr	^{137}Cs	^{90}Sr	^{137}Cs	^{137}Cs	Remarks
P_{01}	Bq m^{-3} y (PBq)$^{-1}$			9.3 ×10^{-6}			40-60°N
P_{12}	Bq m^{-2} (Bq m^{-3} y)$^{-1}$			5.56×10^5			Equivalent to 1.8 cm/s
P_{23} (total diet)	Bq kg^{-1} y (kBq m^{-2})$^{-1}$	3.8	8.4	–	–	–	} Mean of Argentina,
P_{34} (total diet)	kg y^{-1} cap^{-1}	500		–	–	–	} Denmark and USA
P_{45}	nSv Bq^{-1}	28	13	350	8.5	–	ICRP
P_{14}	m^3 y^{-1} cap^{-1}	–	–	7300		–	Inhalation rate: 20 m^3/d
P_{24}	Bq cap^{-1} (kBq m^{-2})$^{-1}$	1900	4200	13		–	$P_{23} \times P_{34}$
P_{25}	mSv (kBq m^{-2})$^{-1}$	52	55	4.6	0.11	97	$P_{24} \times P_{45}$ ingest

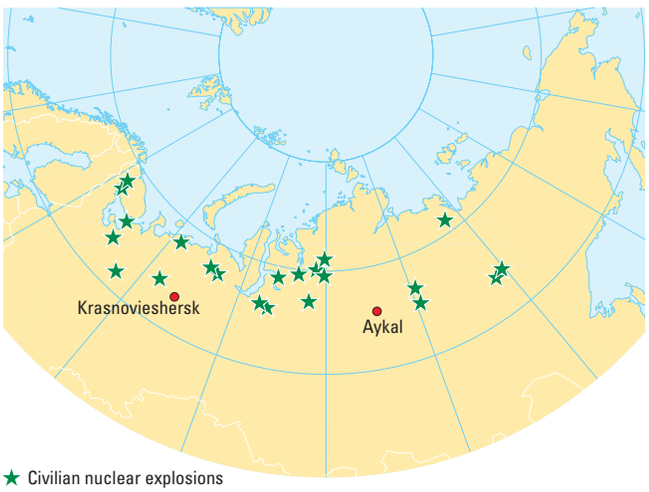
The contribution from other shorter-lived radionuclides present in global fallout, such as ^{3}H, ^{54}Mn, ^{95}Zr, ^{95}Nb, ^{106}Ru, ^{131}I, and ^{144}Ce, will increase the dose to present generations in the Arctic by about 1000 manSv (UNSCEAR 1993). Furthermore, if the doses to be received within the next 50 years from very long-lived radionuclides such as ^{14}C and the transuranic elements (Pu and Am) are included, the dose to Arctic populations from global fallout radionuclides other than ^{90}Sr and ^{137}Cs is estimated to be in the order of 2000 manSv. Hence, the total collective dose commitment over the next 50 years from nuclear weapons testing in the atmosphere to the Arctic population of 3.8×10^6 will be of the order of 13 000 manSv (see section 8.4.3).

8.5.1.2. Underground nuclear explosions

8.5.1.2.1. Underground explosions carried out in the Arctic by the former Soviet Union

Military nuclear explosions carried out by the FSU at underground locations on Novaya Zemlya are described in section 8.3.1.2.1 in the context of localized contamination. The description of these weapons tests is not repeated here.

During the period 1965-1988, a total of 116 peaceful underground nuclear explosions (PUNEs) have been carried out in the FSU (Figure 8·53). Of these, 17 were conducted in regions near the Arctic Circle from 1971 to 1988. Specifically, one was carried out in the Komi Republic, two in the Murmansk region, two in the Tjumen region, four in the Krasnoyarsk region, four in the Sakha-Yakutia Republic, and four in the Archangelsk region (RCRA 1997, Lystsov 1995). A recent Russian report compiling information on nuclear explosions by the FSU (Mikhailov *et al*. 1996) has been used as an additional source of information in this assessment.



8·53. Location of PUNE's in Arctic Russia.

The main application of PUNEs was for mining and construction purposes. PUNEs were also used for emergency extinction of gas-gushers, oil and gas production, creating underground storage cavities, disposal of toxic liquid waste, ore crushing, extinguishing gas fires in coal pits, rock excavation, rock loosening and cratering.

These explosions had yields ranging from less than 0.5-40 kt of TNT equivalent and the total yield was somewhat less than 550 kt of TNT. They were carried out at depths of between 100 and 2860 m. It is difficult to estimate the scales of surface radioactive contamination resulting from these nuclear explosions as information on the condition of the surrounding regions has only recently been made available.

With deep underground explosions, the release of radioactive products is possible only through soil layers or along fissures, and takes a considerable time. Only inert gases and, exceptionally, isotopes of highly volatile elements (e.g., halogens), are released into the atmosphere. Releases of the isotopes ^{133}Xe, ^{135}Xe, ^{137}Xe and ^{138}Xe are the most common.

With shallow ventilated underground explosions, only isotopes of iodine provide unambiguous indicators of venting. However, only insignificant proportions of these isotopes are normally released into the atmosphere. If the release of radioactive products into the atmosphere in case of a camouflet explosion (one that does not result in immediate venting to the atmosphere) commences a few minutes after the explosion, the isotopes of inert gases, whose decay products form aerosols, may enter the atmosphere. Such isotopes of inert gases ($^{89,90}Kr$ and ^{137}Xe) have rather short half-lives and, therefore, their decay products ($^{89,90}Sr$ and ^{137}Cs) are more important. These radionuclides, together with tritium and long-lived isotopes of induced activity (e.g., ^{60}Co), are the dominant contaminants of the environment in cases where releases of radioactive products into the atmosphere actually occur. These radionuclides will be released to the atmosphere in the radioactive cloud in the case of camouflet or excavating explosions, either when the cavity breaks or the rock roof fails due to gas pressure.

A short-range surface plume in the vicinity of the explosion results from the deposition of most of the larger particles from the radioactive cloud and may extend for several hundred meters. The short-range plume may be defined by fallout from the cloud during the first 24-hours following releases from the explosion.

At three PUNE sites there has been significant local contamination:

- At the first site, located 100 km to the north from Krasnovieshersk, an underground explosion took place on 23 March, 1971. Three 15 kt devices were exploded simultaneously at a rather shallow depth of 128 m. The purpose of the explosion was to construct a canal, as part of a major project to alter the direction of some northward flowing rivers to flow to the south. This explosion was the first of 250 planned PUNEs intended to excavate this canal. The 700 m long trench formed by the three devices was significantly contaminated. Information is not available on the types or amounts of this contamination. However, radiation levels in the area were up to 1 mrem/h (10 mSv/h) 15 years after the explosion.
- At the second site, located 90 km to the northeast of Aykal, an explosion took place on 2 October, 1974 at a depth of 98 m. The explosive yield was 1.7 kt and its purpose was the construction of a dam. The explosion resulted in an accidental release of radionuclides, mainly ^{137}Cs, ^{90}Sr, $^{239,240}Pu$, ^{60}Co, ^{125}Sb and ^{241}Am. A total of eight such explosions were planned, but the program was abandoned after this accident. In situ measurements were not performed until 1990-1993. These showed contamination in soil samples of almost 20 kBq/kg of ^{137}Cs and more than 35 kBq/kg of $^{239,240}Pu$. Unfortunately, no information is given on soil sample depth and, therefore, these values cannot be transformed into estimates of areal contamination in Bq/m^2. Neither is it possible, from the information given, to estimate the amounts of specific radionuclides released. However, there is some information about environmental consequences, such as the death of trees (Lystsov 1995).
- At the third site, located 120 km to the southeast of Aykal, an explosion took place on 24 August, 1978, at a depth of 577 m. The explosive yield was 19 kt. Its pur-

pose was seismic sounding of the Earth's crust. After the fifth second following the explosion, a radioactive release was recorded. It is believed that the release took place via an incompletely sealed well. The contaminated cloud moved over the workers camp and the site and about 80 persons were exposed. The dominant radionuclides during the first days following the explosion were ^{131}I, ^{140}Ba and ^{140}La. Currently, the dominant radionuclides are ^{137}Cs, ^{90}Sr, ^{238}Pu, ^{239,240}Pu, ^{60}Co and ^{125}Sb. Large-scale decontamination was carried out in the summer of 1981. From the information provided, it is not possible to estimate the magnitude of the contamination. Soil samples were taken in 1990-1992 and showed concentrations of less than 1 kBq/kg of ^{90}Sr, less than 2 kBq/kg of ^{239,240}Pu and less than 10 kBq/kg of ^{137}Cs. Concentrations of ^{90}Sr in reindeer moss (lichen) and shrubs of up to 44 kBq/kg have been found. At this site, a 'dead forest' (100 hectares) has been reported (Lystsov 1995).

8.5.1.2.2. Underground explosions carried out in the Arctic by the United States

Amchitka Island, Alaska, the southernmost island of the Rat Island Group in the Aleutian Chain, was the site of three high-yield underground nuclear explosions for seismic studies, calibration and warhead development between 1965 and 1971. Amchitka lies approximately 2100 km southwest of Anchorage. The island is about 64 km long and 1.6-6.4 km wide with an area of approximately 30 000 ha. The surface elevation of all three test sites was 41 m above sea level. Unlike the underground explosions in the Russian Arctic, which had explosive yields of less than 50 kilotons, the USA explosions in Amchitka were of much larger yields as itemized below:

- *Long Shot*, October 29, 1965, was detonated at 716 m depth in basalt as part of the *Vela Uniform* program to obtain event measurements relating to the detection of underground nuclear explosions. The yield of *Long Shot* was 80 kt.
- *Milrow*, October 21969, was detonated at 1220 m depth in pillow lava as a seismic calibration test. The yield of *Milrow* was approximately 1 Mt.
- *Cannikin*, November 6, 1971, was detonated at 1790 m depth in basalt as a test of the proposed warhead for the Spartan missile. The yield of *Cannikin* was 5 Mt.

Site deactivation was started by the US Department of Energy in February, 1972, and completed in September, 1973.

No known escape of radionuclides to the surface has occurred as a result of these tests with the exception of trace quantities of tritium which can be detected in surface water and shallow groundwater above the *Long Shot* detonation point. A preliminary assessment conducted by the US Department of Energy (USDOE) under the terms of the Comprehensive Environmental Response, Compensation and Liability Act was submitted to the US Environmental Protection Agency (USEPA) in 1988. This assessment recommended that sampling in the vicinity of the test sites be continued and that the results of migration studies carried out at the Nevada test sites be evaluated for estimating the potential for radionuclide migration at Amchitka.

The three test sites continue to be monitored as part of the Long-Term Hydrological Monitoring Program (LTHMP) carried out by the USEPA. Under this program, samples are collected biennially. Samples of groundwater (23), surface water (19), spring water (1) and rainwater (1) were last collected in 1995.

Environmental contamination that has occurred as a result of these tests is specified as:

- Subsurface contamination surrounding each shot cavity.
- Several seeps containing trace amounts of radionuclides (principally tritium) have been found at, or near, the mud disposal pits at the *Long Shot* ground zero.
- Erosion of buried waste disposal pits has the potential for leakage into the ocean.

An environmental organization issued a report on October 30, 1996, detailing the results of radionuclide sampling conducted on Amchitka Island during the summer of that year. This organization alleges, on the basis of these data, that samples of moss indicate leakage of ^{239,240,241}Pu and ^{241}Am from the cavity created by the *Cannikin* explosion. While it does not claim that these releases constitute a significant risk to human health, it contends that future releases could constitute a significant risk, especially if bioaccumulation occurs.

The USEPA will assess the samples collected by the environmental organization in 1996 to validate the results and USEPA samples to be collected in 1997 will be provided to the environmental organization and other interested groups for cross-checking. Results of the most recent sampling will be made available by September 30, 1997. The biennial sampling program at Amchitka will be expanded to include appropriate biota to address concerns about bioaccumulation of radionuclides in the food chain. The State of Alaska, native groups (including the Aleutian and Pribiloff Islands Association) and other interested parties will participate in the formulation of objectives and protocols for the sampling program. The USDOE will review all available data to determine whether there are any monitoring or other data which would indicate previously unreported releases of radionuclides from the three shot cavities. Finally, relevant classified materials will be reviewed with a view to de-classification.

8.5.2. Operational releases from the nuclear fuel cycle

The term 'nuclear fuel cycle' is used to delineate activities associated with the production of energy from fission reactors encompassing uranium mining, uranium enrichment, fuel fabrication, reactor fuel insertion, nuclear reactor operation, spent fuel removal, reprocessing and the ultimate disposal of wastes from the nuclear power industry. This subsection deals with operational releases from nuclear power plants and nuclear fuel reprocessing plants.

8.5.2.1. Nuclear power plants

There are two nuclear power plants (NPPs) within the Arctic, the Kola NPP near Polyarny Zori on the Kola Peninsula and the Bilibino NPP which is in the Chukchi region of eastern Russia. In addition, there are NPPs in Sweden, Finland and Russia that are within 1000 km of the Arctic Circle.

8.5.2.1.1. Nuclear power plants in the Arctic

The two NPPs operating in the Arctic are: one plant with four VVER (water-cooled and water-moderated energy reactor) units at Kola and one plant with four EGP-6 (light water cooled graphite-moderated) units at Bilibino (Table 8·22, next page) (RCRA 1997).

The criteria (Tsaturov 1996) for critical group exposure used to calculate release limits for normal operation of nuclear power plants in Russia are shown in Table 8·23.

Table 8·22. Russian NPPs in the Arctic.

Unit	Type/model	Installed capacity, MW(e)	Commercial start-up date
Kola 1	VVER - 440/230	440	1973
Kola 2	VVER - 440/230	440	1974
Kola 3	VVER - 440/213	440	1981
Kola 4	VVER - 440/213	440	1984
Bilibino 1	EGP-6	12	1974
Bilibino 2	EGP-6	12	1974
Bilibino 3	EGP-6	12	1975
Bilibino 4	EGP-6	12	1976

Table 8·23. Dose criteria for releases from NPPs in Russia.

Critical organ	Criteria for airborne effluents, mSv/y	Criteria for liquid effluents, mSv/y
Whole body and bone marrow	0.2	0.05
Other organs (except skin and extremities)	0.6	0.15
Skin and extremities	1.2	0.30

Table 8·24. Release limits of radionuclides in airborne effluents for Kola and Bilibino NPPs.

	Release limits, GBq/y (Ci/y)		
NPP	Inert gases	Radioiodine	Long-lived nuclides
Kola	27000000 (730000)	540 (15.0)	810 (22.0)
Bilibino	6700000 (180000)	140 (3.7)	2040 (55.0)

Table 8·25. Annual radioactive releases into the atmosphere from the Kola NPP, 1985-1994.

	Releases, GBq		
Year	Inert gases	^{131}I	Long-lived nuclides
1985	1040000	0.54	0.22
1986	540000	0.77	0.43
1987	560000	1.60	0.91
1988	420000	1.10	0.89
1989	420000	64	113
1990	270000	3.5	85
1991	–	–	–
1992	280000	1.20	2.55
1993	180000	5.6	3.20
1994	82000	3.1	3.0

Table 8·26. Annual radionuclide releases into the atmosphere from the Bilibino NPP, 1991-1994.

	Releases, GBq			
Year	Inert gases	Short-lived radionuclides, half-life < 24 h	Long-lived radionuclides, half-life > 24 h	^{131}I
1991	274100	1.7	Background	n.d.
1992	353800	4.4	Background	n.d.
1993	326300	0.89	Background	n.d.
1994	417100	1.8	Background	n.d.

Table 8·27. Annual liquid releases of some radionuclides from the Kola NPP, 1988-1993.

	Releases, GBq					
Radionuclide	1988	1989	1990	1991	1992	1993
Chromium-51	0.037	0.31	0.54	–	0.093	0.19
Manganese-54	0.11	0.63	0.79	–	0.63	0.82
Cobalt-58	0.1	1.07	0.95	–	–	0.27
Cobalt-60	0.28	1.31	1.14	–	0.99	1.07
Strontium-90	0.014	0.005	0.005	–	–	–
Caesium-134	0.02	0.035	0.072	–	0.16	0.14
Caesium-137	0.083	0.078	0.24	–	–	0.24

Atmospheric releases

The limits for releases of radionuclides in airborne effluents are calculated from the above criteria and are shown in Table 8·24.

Actual releases into the atmosphere from the Kola NPP between 1985 and 1994 are shown in Table 8·25.

Actual radionuclide releases into the atmosphere from the Bilibino NPP for the period 1991-1994 are shown in Table 8·26.

Liquid releases

The liquid release rates of some radionuclides from the Kola NPP for the period 1988 -1993 are shown in Table 8·27.

Other wastes

Disposal sites at the Kola NPP contain 5000 m^3 of solid radioactive wastes having 155×10^{10} Bq (≈42 Ci) of activity and 65 000 m^3 of liquid wastes having 7×10^{14} Bq ($\approx 1.9 \times 10^4$ Ci) of activity.

The associated collective and critical group doses associated with releases from these NPPs were not available to the assessment group.

8.5.2.1.2. **Nuclear power plants in the vicinity of the Arctic**

There are additional NPPs in Russia, Finland and Sweden situated within 1000 km of the Arctic Circle. These may have relevance to Arctic populations, particularly in the event of accidents, and have been included here for completeness.

Russian NPPs

The Leningrad (now St. Petersburg) NPP is situated about 1000 km from the Arctic Circle. It has four units with graphite moderated pressure tube boiling water reactors, a type of reactor which has only been constructed in the former Soviet Union. The graphite consists of blocks that are arranged in the form of columns penetrated by vertical channels that provide locations for the fuel rods, control rods, graphite reflector coolant tubes and instrumentation (NKS 1994). The electrical output of each unit is 1000 MW(e). The plant has been built in two versions. The main differences between these versions are the nature of the emergency cooling systems and containment systems. The four units of the Leningrad NPP were put into operation in the years 1973, 1975, 1979 and 1981, respectively.

The releases as gas and aerosols into air from the Leningrad power plant are given in Table 8·28.

Table 8·28. Annual releases to the atmosphere of radionuclides as gas and aerosols from the Leningrad NPP, 1992-1995, GBq (Ci).

Radio-nuclide	Releases in gases and aerosols to the atmosphere, GBq (Ci, values in parentheses). 1992	1993	1994	1995
Inert radio-active gases	1390000 (37625)	1610000 (43629)	1790000 (48360)	1070000 (29000)
Long-lived nuclides	81 (2.2)	20.9 (0.564)	59.9 (1.62)	35 (0.95)
^{131}I	89 (2.4)	20 (0.53)	50 (1.36)	20 (0.53)
^{51}Cr	14 (0.37)	– –	13.6 (0.367)	5.9 (0.16)
^{54}Mn	0.22 (6×10^{-3})	–	0.31 (8.43×10^{-3})	0.1 (2.6×10^{-3})
^{60}Co	0.67 (18×10^{-3})	– –	0.41 (11.2×10^{-3})	0.14 (3.7×10^{-3})
^{137}Cs	2.6 (69×10^{-3})	0.95 (25.6×10^{-3})	1.88 (50.9×10^{-3})	1.2 (33×10^{-3})
^{89}Sr	0.13 (3.4×10^{-3})	0.10 (2.63×10^{-3})	0.28 (7.6×10^{-3})	0.08 (2.1×10^{-3})
^{90}Sr	0.03 (0.8×10^{-3})	0.10 (2.63×10^{-3})	0.04 (1.1×10^{-3})	0.02 (0.54×10^{-3})

The collective and critical group doses associated with releases from the Leningrad NPP were not available to the AMAP radioactivity assessment group and, thus, could not be included for comparative purposes.

Finnish NPPs

Finland has two nuclear power plants both situated on the Baltic Sea coast: Loviisa on the Gulf of Finland; and Olkiluoto on the Gulf of Bothnia. Two units are in operation at both sites (Table 8·29).

Table 8·29. Finnish NPPs.

Unit (Company)	Reactor type/model	Installed capacity, gross/net, MW(e)	Commercial start-up date
Loviisa 1 (IVO)	VVER 213 (PWR)	465 / 445	1977
Loviisa 2 (IVO)	VVER 213 (PWR)	465 / 445	1981
Olkiluoto 1 (TVO)	BWR	735 / 710	1979
Olkiluoto 2 (TVO)	BWR	735 / 710	1982

PWR: Pressurized water reactor; BWR: Boiling water reactor.

Operation of nuclear power plants is regulated by the national Nuclear Energy Act of 1987 (NEA 1987) and the Nuclear Energy Decree of 1988 (NED 1988). The Council of State has issued General Regulations for the Safety of Nuclear Power Plants (DCS 1991). The Radiation Act (RA 1991) and the Radiation Decree (RD 1991) set forth the general regulations for the limitation of radiation exposure. The Decision of the Council of State stipulates that the limit for the dose commitment to an individual of the population associated with the normal operation of a nuclear power plant in any one year period is 0.1 mSv. Based on this limit, release rate limits for radionuclides during normal operation are defined and specified in the authorization. Adherence to these limits alone does not suffice; in addition, radionuclide releases shall be kept as low as reasonably achievable (ALARA). To limit the overall exposure, the global collective dose commitment to the population, truncated at 500 years, arising from normal operation of a nuclear power plant for any period of one year, is limited to 5 manSv/GW(e) installed net electrical capacity (STUK 1992).

Figure 8·54 shows that the annual radiation doses to critical groups associated with releases from Finnish NPPs are very much lower than the limit of 0.1 mSv/y.

Table 8·30 tabulates the radionuclide releases (GBq/y) for the six radionuclides in liquid effluent contributing most to the individual dose, for each of the two NPPs, for the period 1990-1994. The corresponding releases in gaseous effluent are presented in Table 8·31 (STUK 1994).

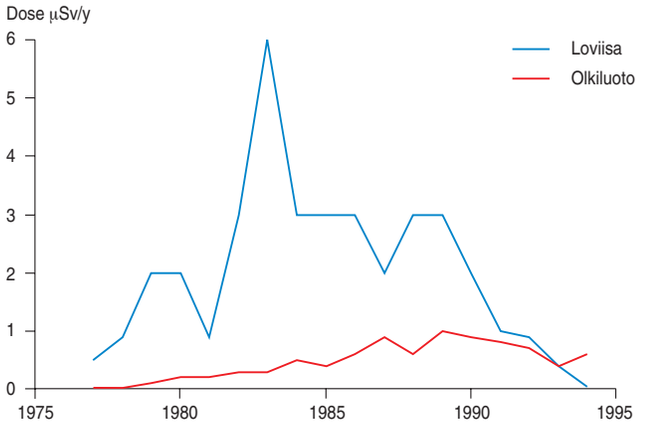


Figure 8·54. Annual radiation doses to critical groups from the Loviisa and Olkiluoto NPP's.

Table 8·30. Annual releases of the most abundant radionuclides in liquid effluents from the Loviisa and Olkiluoto power plants in Finland (GBq), 1990-1994.

	Releases, GBq				
Site/radionuclide	1990	1991	1992	1993	1994
Loviisa					
Tritium	12000	14000	10000	12000	6600
Cobalt-60	2.0	1.3	1.4	0.8	0.1
Silver-110m	3.1	1.6	0.9	0.5	0.02
Antimony-124	7.2	0.1	0.2	0.1	0.2
Caesium-134	1.7	0.4	0.2	0.5	0.001
Caesium-137	3.3	1.5	0.5	0.2	0.002
Olkiluoto					
Tritium	1300	1900	1800	840	2800
Chromium-51	1.5	4.3	3.6	2.1	0.2
Manganese-54	9.0	4.6	2.9	2.9	3.1
Cobalt-58	2.5	1.2	0.9	0.3	0.3
Cobalt-60	16	8.4	7.3	3.6	5.9
Caesium-137	0.6	0.5	0.2	0.04	0.5

Table 8·31. Annual airborne releases from Loviisa and Olkiluoto NPPs (GBq), 1990-1994.

	Releases, GBq				
Site/radionuclide	1990	1991	1992	1993	1994
Loviisa					
Noble gases	1000	1000	1800	1600	1500
Tritium	740	480	230	210	210
Carbon-14	310	320	150	190	180
Aerosols	0.2	0.2	0.3	0.08	0.2
Iodines	0.02	0.2	0.03	0.03	0.0002
Olkiluoto					
Noble gases	22000	43000	29000	9500	3500
Tritium	100	130	350	430	240
Carbon-14	640	640	640	650	470
Aerosols	0.2	0.7	0.3	0.1	0.1
Iodines	0.06	0.3	0.2	0.08	1.1

The annual release limits for liquid effluents from the Loviisa power plant are:

Tritium	150 000 GBq/y
Other nuclides	890 GBq/y

The release limits for liquid effluents from the Olkiluoto power plant are:

Tritium	18 000 GBq/y
Other nuclides	300 GBq/y

The group of noble gases has been calculated as ^{87}Kr-equivalents. The principal noble gas released from Loviisa is ^{41}Ar, and for Olkiluoto xenon isotopes. Among the aerosols, the main releases from Loviisa comprise ^{54}Mn, ^{60}Co, ^{76}As, ^{110m}Ag and ^{124}Sb, and those from Olkiluoto ^{51}Cr, ^{54}Mn, ^{58}Co, ^{60}Co and ^{99m}Tc.

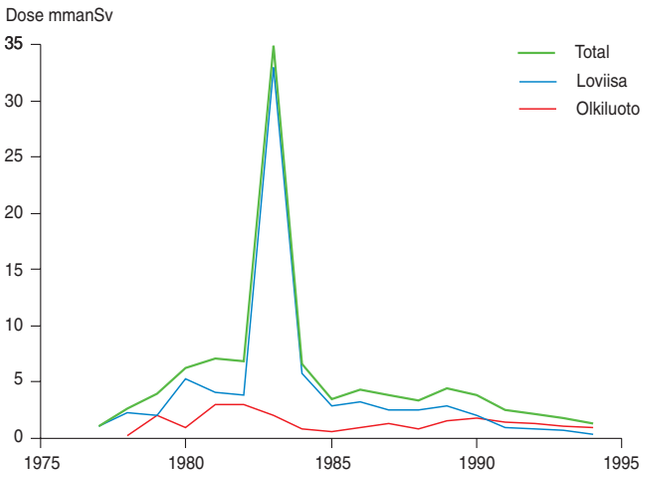


Figure 8·55. Collective doses from the Loviisa and Olkiluoto NPP's.

The annual release limits for gaseous effluents from the Loviisa power plant are:

Noble gases (in ^{87}Kr-equivalents)	22 000 000 GBq/y
Iodines	220 GBq/y

The annual release limits for gaseous effluents from the Olkiluoto power plant are:

Noble gases	18 000 000 GBq/y
Iodines	110 GBq/y

Collective doses calculated on the basis of measured release rates, excluding ^{14}C, from Loviisa and Olkiluoto power plants are depicted in Figure 8·55.

Dose estimates for both individual and collective exposures do not include ^{14}C because ^{14}C releases are estimated from energy output figures and only occasionally checked by measurement. However, since 1992, the ^{14}C release in gaseous effluents from the Loviisa NPP has been measured. The annual effective dose equivalent to the most exposed individual is estimated to be of the order of a few µSv. The estimated global collective doses from annual ^{14}C releases are 4 manSv for Loviisa and 7 manSv for Olkiluoto (assuming 10^{10} global population and 500 years integration time) (STUK 1991).

Swedish NPPs

Sweden has 12 nuclear power units at four sites. Table 8·32 portrays the types, electrical installed capacity and the year of commercial start-up.

Table 8·32. Swedish NPPs.

Plant and unit	Type	Installed capacity, gross / net, MW(e)	Year of commercial start-up
Barsebäck 1	BWR	615 / 600	1975
Barsebäck 2	BWR	615 / 600	1977
Forsmark 1	BWR	1006 / 968	1981
Forsmark 2	BWR	1006 / 969	1981
Forsmark 3	BWR	1197 / 1155	1985
Oskarshamn 1	BWR	462 / 442	1972
Oskarshamn 2	BWR	630 / 605	1974
Oskarshamn 3	BWR	1205 / 1160	1985
Ringhals 1	BWR	825 / 795	1973
Ringhals 2	PWR	905 / 875	1974
Ringhals 3	PWR	960 / 915	1980
Ringhals 4	PWR	960 / 915	1982

PWR: Pressurized water reactor; BWR: Boiling water reactor.

Based on the national Radiation Protection Act (RPA 1988) and Ordinance (RPO 1988) on radiation protection, the Swedish Radiation Protection Institute has stipulated reference values for release limits in terms of radiation dose equivalent to the critical group and global collective dose per GW(e) installed net electrical capacity. The values are 0.1 mSv/y and 5 manSv/GW(e) respectively for one year's releases using a truncation time for the collective dose of 500 years. Reference releases were calculated on the basis of a dose of 0.1 mSv/y to the critical group taking into account all radionuclides and all pathways of exposure. In addition, the ALARA principle is applicable. There are detailed requirements for reporting of situations in which the reference releases are expected to be exceeded and the upper limits for operational releases are defined on the basis of ICRP Dose Limits for members of the public (SSI FS 1991).

Table 8·33 depicts doses to individuals in critical groups based on measured releases. Carbon-14 is listed separately as the values for exposures to this nuclide are based on theoretical calculations and test measurements. Table 8·34 depicts corresponding collective doses (SSI 1994).

Table 8·33 clearly shows that doses to critical group members are lower than the reference dose value of 0.1 mSv/y.

Table 8·33. Annual doses (mSv/y) to critical groups from Swedish NPPs, 1989-1994.

	Annual doses, µSv/y					
NPP	1989	1990	1991	1992	1993	1994
Barseback	1.86	0.29	0.86	0.56	0.15	0.09
(^{14}C)[a]	(0.92)	(0.92)	(0.96)	(0.30)	(0.32)	(0.6)
Forsmark	0.87	0.48	0.50	0.26	0.35	0.10
(^{14}C)[a]	(0.57)	(0.57)	(0.62)	(0.28)	(0.29)	(0.4)
Oskarshamn	2.90	2.24	1.72	0.90	0.75	0.44
(^{14}C)[a]	(0.45)	(0.45)	(0.49)	(0.17)	(0.15)	(0.2)
Ringhals	1.59	2.38	1.39	3.4	19	36
(^{14}C)[a]	(11.0)	(11.0)	(11.4)	(8.5)	(8.6)	(6.7)

a. Calculated ^{14}C contribution.

The purpose of a reference value expressed as collective dose per GW(e) is partly to limit future individual doses from several power plants, all of which may contribute to dose to the same individual, and partly to determine the total risk associated with releases. As can be calculated from the values given in Table 8·34 and the installed electrical capacities of the power plants in Table 8·32, the reference value for collective dose has been exceeded several times. However, it should be noted that the assumed extent of investment in nuclear power used when deriving the reference value was much greater than that actually constructed. Accordingly, there is assurance that the highest future dose to individuals will not exceed 0.1 mSv/y which was the main purpose of placing limits on collective dose per unit installed capacity.

Table 8·34. Collective doses (mmanSv) associated with releases from Swedish NPPs, 1989-1994.

	Collective dose, mmanSv					
NPP	1989	1990	1991	1992	1993	1994
Barsebäck	90	10	30	10	3	0.24
(^{14}C)[a]	(7100)	(7100)	(7400)	(4000)	(4300)	(5700)
Forsmark	140	20	20	10	10	6.1
(^{14}C)[a]	(18000)	(18000)	(19300)	(14900)	(15400)	(17600)
Oskarshamn	40	30	20	20	17	13
(^{14}C)[a]	(13000)	(13000)	(13800)	(8300)	(7500)	(9200)
Ringhals	10	10	10	50	293	510
(^{14}C)[a]	(9800)	(9800)	(10600)	(7000)	(6800)	(8500)

a. Calculated ^{14}C contribution.

As in the case of Finnish NPPs, the radionuclides in liquid effluents contributing most to critical group dose are ^{3}H, ^{51}Cr, ^{54}Mn, ^{58}Co, ^{60}Co, ^{134}Cs, ^{137}Cs, ^{65}Zn and ^{124}Sb. The orders of magnitude of corresponding releases are GBq/y except for tritium which is of the order of TBq/y. Airborne releases comprise mainly krypton and xenon isotopes and these occur at rates in the order of TBq/y.

It is noteworthy that airborne effluent releases of ^{131}I from the Leningrad NPP are reported to vary between 20 and 89 GBq/y. This is a substantial release rate compared with the Finnish effluent releases presented in Table 8·30, but, nevertheless, lower than the Finnish release rate limits. Airborne releases of ^{131}I from Swedish NPPs have varied between 0.006 and 38 GBq/y during the last decade.

Overall, routine discharges from nuclear power plants are small and contribute little to the contamination of the Arctic environment or to the doses of Arctic residents.

8.5.2.2. **Russian civilian nuclear fleet**

Russia currently has seven nuclear-powered civilian vessels in operation of which six are nuclear-powered icebreakers and one is a nuclear-powered container ship. These are:

- Four icebreakers with two reactors each of capacity 171 MW(t) (*Arctica*, *Rossiya*, *Sovietskiy Soyuz* and *Yamal*).

- Two icebreakers with one reactor each of capacity 171 MW(t) (*Taimyr* and *Vaigach*).
- A lighter-container carrier *Sevmorput* with one reactor of 135 MW(t) capacity.

The vessels, all operated by the Murmansk Shipping Company (MSC), operate from their base *Atomflot* located 2 km north of the town of Murmansk on the Kola Peninsula. The nuclear-powered icebreakers have been constructed since the 1950s for facilitating increased shipping along the northern coast of Siberia. They have also been used for scientific expeditions in the Arctic and, since 1989, the carriage of tourists to the North Pole (Filippov 1996).

Spent nuclear fuel was unloaded from two old, partially-decommissioned, icebreakers *Lenin* and *Sibir*. These vessels are anchored and are manned to maintain safety. During the early operational period of the first nuclear-powered icebreaker *Lenin* there were some reactor accidents. As a result of one serious accident, the reactor unit with three reactors was cut out and dumped in 1967 in the Kara Sea near Novaya Zemlya. One of the reactors contained part of the damaged spent fuel with an estimated activity at the time of disposal of about 3.7 PBq (OPRF 1993). Potential radiological dangers associated with this operation are considered in section 8.6.

The Murmansk Shipping Company also operates auxiliary ships for the collection and storage of spent fuel and radioactive waste:

- Two floating technical bases for reloading fuel assemblies and the storage of spent fuel (with 1530 assemblies on the *Imandra* and 4080 assemblies on the *Lotta*).
- A floating technical base *Volodorasky* for the storage of solid radioactive waste of 300 m^3 volume.
- A special tanker *Serebryanka* for the collection and storage of liquid radioactive waste of 1000 m^3 volume.
- Vessels for sanitary treatment of personnel and dosimetric control (e.g., *Rosta 1*).

Four auxiliary ships were out of service and two of them need comprehensive remediation:

- A floating technical base/storage facility for spent nuclear fuel *Lepse*.
- A floating dosimetric control unit (FDCU-5).

A total of about 6000 spent fuel assemblies from the Murmansk Shipping Company are stored on the floating bases *Lepse*, *Imandra* and *Lotta*. Auxiliary ships of the atomic fleet are ice-class vessels with double hulls. The atomic icebreakers and floating bases were designed to withstand collisions with other ships and to remain afloat in the case of flooding of two adjacent compartments. The tanks containing spent nuclear fuel have biological shielding made of steel of thickness 380-450 mm. Prevention of criticality is achieved through the use of a specific geometric arrangement of spent fuel assemblies. The storage tanks have special water-cooling systems independent of those of the vessels. Special tanks for liquid radioactive wastes are made of stainless steel and are placed in the vessel's hull. The tanks are provided with corrosion and biological protection (Filippov 1996).

Spent nuclear fuel is offloaded from ship reactors and stored on the *Imandra* for 0.5-1 year, and then transferred to the *Lotta* where it is stored for a further period of about three years. Subsequently, spent fuel is transported by rail to the reprocessing plant Mayak in the Urals.

The storage facility on the floating base *Lepse* is in a poor state. It contains 28 PBq in 642 damaged fuel assemblies within two tanks. Long-lived transuranic radionuclides comprise 0.6 PBq of this total. The problems of damaged fuel disposal and storage vessel decommissioning is currently being addressed by French, British and Russian experts (Filippov 1996).

In general, the operations and equipment of civilian atomic fleet vessels accord with relevant international and national regulations. This should ensure appropriate levels of safety in the processes of handling and storage of nuclear wastes, including activities on the storage vessels. During the many years of operation of auxiliary ships, there has not been a recorded case of an emergency having negative effects on the environment.

The nuclear fleet vessels are currently stationed, repaired and maintained, along with reactor refueling, at the technologic enterprise *Atomflot* located in the vicinity of Murmansk. Radionuclides can be released to the environment both from ships and the *Atomflot* base during the following operations:

- Refueling.
- Decontamination and repair.
- Handling radioactive waste including transportation and processing.
- Storage of spent nuclear fuel (SNF) on ships.

The main environmental concern relates to releases of radioactive wastes generated as a result of the operation of nuclear-powered ships.

The gas-aerosol releases on nuclear icebreakers comprise mostly radioactive inert gases and vary in the range 40-400 GBq/y, which is two to three orders of magnitude less than the actual releases of these radionuclides from land-based nuclear power plants. During storage of SNF on the service ship *Imandra*, the gas release consisted primarily of ^{85}Kr. During the storage period, not more than 400 GBq is released from a spent core, over 90% of it during the first three months of storage. The air concentration of radionuclides outside the reactors, nuclear ships and service ship *Imandra* during gas releases in all refueling and reactor repair operations does not exceed the derived air concentrations prescribed by the Russian norms for radiation safety.

When burning solid wastes, a mixture of radionuclides is released to the atmosphere containing ^{137}Cs, ^{60}Co, ^{90}Sr, ^{54}Mn, ^{152}Eu and ^{154}Eu. The actual releases of these radionuclides amount to 3×10^{-6} to 1×10^{-2} percent of the permissible level.

In total, during four years of operation of a single reactor core, 130 m^3 of liquid radioactive waste are generated, of which 100 m^3 arises on the nuclear vessel and 30 m^3 on a floating refueling base. The total annual activity of the liquid waste amounts to 4-300 GBq with an average of 20 GBq. Since 1989, a pilot unit for the decontamination of liquid radioactive wastes of 1200 m^3/y capacity has been operating on RTP *Atomflot*. Following decontamination to permissible concentrations and analysis of the remaining artificial radionuclide content, the waste liquid is discharged to Kola Bay.

Altogether, during the operational life of one core, about 32 m^3 of solid radioactive wastes of total activity 2 TBq are generated. Most of the solid radioactive wastes (≈90%) are low-level, of which 50-70% is combustible. Solid radioactive wastes from ships are passed to a special complex of RTP *Atomflot* for compaction and storage. Since 1989, incineration of combustible solid radioactive wastes has been performed in a burning unit having a capacity of 40 kg/h.

The actual levels of radionuclides in air samples over water areas adjacent to RTP *Atomflot* from 1985-1994 are below control levels. Increased levels detected in air and rainfall in May 1986, were due to the Chernobyl accident.

Environmental monitoring during 1992-94, when five refueling operations were carried out, indicates that inputs from refueling constitute 3-32% of the annual release limits

for specific monitoring points. Wholebody measurements on working personnel show that the levels of ^{134}Cs, ^{137}Cs and ^{95}Zr are about 0.01% of permissible levels.

In 1993-94, studies of bottom sediments in Kola Bay in the vicinity of RTP *Atomflot* were conducted. Gamma-spectrometry revealed the presence of ^{137}Cs, ^{60}Co, ^{152}Eu and ^{154}Eu. However, the measured radionuclide concentrations do not provide reason for undue concern about the impact of discharges from this enterprise on the environment (Kolomiets *et al.* 1992, Filippov 1996).

8.5.2.3. The Russian Northern Fleet

8.5.2.3.1. Nuclear-powered vessel operations

Since the 1950s, nuclear-powered submarines and surface ships have been operating in the Northern Fleet of the USSR, now that of the Russian Federation. The Russian Northern Fleet currently includes several tens of nuclear-powered submarines and two nuclear-powered cruisers, operating mainly from nine major bases on the Kola Peninsula extending from Gremikha in the east to the Litsa Fjord in the west. Most of these vessels are equipped with two nuclear reactors of pressurized, light water moderated design with capacity of 70-300 MW(t) (CCMS/CDSM/NATO 1995).

Radiation monitoring during the period 1968-1992 has demonstrated that the main sources of radioactive contamination of base locations of the Northern Fleet were ships with nuclear reactors, depot ships, coastal technical bases and supply ships carrying liquid radioactive wastes. The main sources of radioactive waste generation in the Northern Fleet are: bases at Zapadnaya Litsa Inlet, Olenya Bay, Saida Bay, Ara Bay, Pala Bay, and Iokan'ga; sites of nuclear-powered military vessel deployment (i.e., Polarny and Severodvinsk); sites of interim storage of SNF (i.e., Zapadnaya Litsa and Iokan'ga); floating naval bases for refueling of submarines; shipyards; and the repair plants at Polyarny, Vyuzhny, Severodvinsk. The maximum radioactive releases have occurred in liquid form through disposal of decontamination water, leakages of liquid radioactive wastes during the process of transfer from ships to storage bases and through unauthorized releases. Radioactive contamination of the environment (water and air) in such cases is generally of local scale and decays to pre-existing levels relatively quickly (i.e., within a few hours) (OPRF 1993, Lisovsky *et al.* 1996).

Numerous observations have shown that the radiation situation in nuclear-powered ship bases of the Northern Fleet is only abnormal in the cases of accidents. The concentrations of radionuclides in the environment are normally no higher than background levels (see Table 8·35) (Lisovsky *et al.* 1996). Only in exceptional cases, are traces of radionuclides derived from reactors detected in places where monitoring of reactor servicing activities and radioactive waste storage facilities is conducted. There have been several cases of accidental radioactive contamination of the environment during the period of operation of naval nuclear-powered ships. These cases are summarized in section 8.5.3.5.

Table 8·35. Average radionuclide content in the environment of base locations of the Russian North Fleet in 1987 (Lisovsky *et al.* 1996).

Medium	Unit	Concentration of radionuclides				
		^{90}Sr	^{137}Cs	^{144}Ce	^{60}Co	^{210}Po
Seawater	Bq/L	9×10^{-3}	9×10^{-3}	9×10^{-3}	5×10^{-3}	2×10^{-3}
Drinking water	Bq/L	6×10^{-3}	1×10^{-2}	6×10^{-3}	7×10^{-3}	9×10^{-4}
Atmospheric aerosols	Bq/m^3	2×10^{-5}	3×10^{-5}	2×10^{-5}	7×10^{-5}	1×10^{-4}
Atmosph. depositions	Bq/m^2	1×10^{8}	1×10^{8}	7×10^{7}	1×10^{7}	–
Sea algae	Bq/kg	4	3	4	1.5	0.4
Sea bottom sediments	Bq/kg	10	11	8	11	4
Benthic sea organisms	Bq/kg	2	4	2.5	0.7	2
Soil	Bq/kg	10	7	3	4	9
Surface vegetation	Bq/kg	6	4	4	3	2

8.5.2.3.2. Decommissioning

As a result of disarmament and other technological reasons, nuclear submarines of the Northern Fleet are being decommissioned. By the beginning of 1996, the number of submarines taken out of service was about 90 and is likely to exceed 100 by the year 2000. The procedure for decommissioning of submarines involves unloading of SNF from reactors on the ships, after which decontamination is performed, equipment is removed for further use, and the reactor compartment is cut out and placed in a prepared and environmentally-safe storage or disposal site. However, at present, practices do not fully conform to this scheme due to the lack of technical infrastructure and resources. The most urgent task is unloading of SNF from reactors. By the beginning of 1996, SNF was unloaded from only 25% of the submarines taken out of service in the Northern Fleet. In all, six reactor compartments have been prepared for long-term storage. Other submarines that have been decommissioned, but still contain nuclear fuel, are now moored in harbors on the Kola Peninsula awaiting completion of decommissioning (Petrov 1995, Ecological Safety 1996).

According to one scenario, long-term storage of the unloaded reactor compartments is planned to take place in large man-made rock cavities constructed on the shores of the Kola Peninsula about two decades ago. Unloading of SNF from submarines with damaged cores remains an unsolved problem as there are currently inadequate facilities for removal of fuel rods from damaged reactors. Furthermore, in the next few years, those facilities that are coming to the end of their planned operational lifetime are due to be scrapped, in particular the tenders used for refueling and spent nuclear fuel storage. Some of these do not meet contemporary radiation safety requirements.

8.5.2.3.3. Storage of the spent nuclear fuel and radioactive waste

In recent years, the problem of spent nuclear fuel and radioactive wastes disposal has become more acute for the Russian Northern Fleet. Transport of spent nuclear fuel to reprocessing facilities in the Urals has been slowed and the Russian Navy lacks the necessary processing plants. Furthermore, since 1992, radioactive wastes are no longer dumped at sea. All this has led to a build-up of hazardous materials at locations where the nuclear-powered ships are based and overhauled, with associated negative effects on the radiological and environmental conditions. Also, it is difficult to ensure radiation and environmental safety because of the difficulties associated with implementation of the program for the decommissioning of submarines (OPRF 1993, Petrov 1995).

At present, more than 3000 cells with spent fuel assemblies are stored at facilities of the Northern Fleet. As there are seven assemblies in each cell, the total number of assemblies is over 21 000. The majority of these spent fuel assemblies are in storage on the shore of Andreev Bay (Fjord of Zapadnaya Litsa). Because of accidents at previous 'wet' spent nuclear fuel storage facilities located in the same area in the 1980s, spent nuclear fuel is now kept 'dry' in containers intended primarily for liquid radioactive wastes. Temporary storage facilities were built on a crash basis and do not fully comply with radiation safety and environmental protection requirements. Spent nuclear fuel is stored in coastal

facilities in Andreev Bay (80%), Iokan'ga and on tenders (5%). Part of the spent fuel is situated at the shipyard *Atomflot* of the Murmansk Shipping Company because naval facilities for temporary storage of spent nuclear fuel have been exhausted (OPRF 1993, Petrov 1995).

The technical support system for nuclear reactors on ships that was created in the late 1950s and early 1960s was not designed for decommissioning of a large number of nuclear submarines in a short period. This is the primary reason for the unsatisfactory situation regarding SNF and radioactive wastes in the Russian Navy. There are several factors aggravating the problem: the unloading of spent nuclear fuel cannot keep pace with the decommissioning of nuclear submarines; some storage facilities are in a dangerous condition; and there are no available storage containers.

At present, there are two options for the transport of spent nuclear fuel from the Northern Fleet, in newly-designed TUK-18 transport flasks with improved safety features, from Severodvinsk and from *Atomflot*. This, however, does not resolve the problem. To enable transport of SNF from Andreev Bay, 30 km of railway line needs to be constructed and/or a special transport ship built. The spent nuclear fuel on submarines which suffered accidents is still an unresolved issue. A technology is being developed for the transport of spent fuel from liquid metal cooled reactors. This is planned to be implemented at Iokan'ga in 1998 (Petrov 1995, NEFCO 1996).

As a result of the accident at a spent nuclear fuel storage site in Andreev Inlet, water of the cooling ponds continued to be released into the soil of the adjacent territory and entered a nearby stream for several years. The associated radioactive contamination affected an area of 1300 m^2 together with waters of the bay adjacent to the location where the stream discharge occurs. The storage of spent nuclear fuel on the coast of Andreev Bay continues to be a source of current and potential radioactive contamination of the local environment (Petrov 1995, Lisovsky *et al*. 1996).

Liquid radioactive wastes generated during the operation of nuclear-powered vessels are stored in both coastal and floating containers. The volume of all available containers for the Northern Fleet is 10 000 m^3 but 30% of these are unfit for use. The amount of liquid radioactive wastes from the Northern Fleet is estimated to be 7000 m^3 containing a total activity not exceeding 3.7 TBq. About 2000-2500 m^3 of liquid wastes are generated annually but there are essentially no empty containers in which to store them. Liquid wastes of the Northern Fleet are partly transported for processing at *Atomflot*, Murmansk (Petrov 1995). In 1994-96, the amount of liquid radioactive wastes processed was 1500 m^3.

Solid radioactive wastes are stored in storage facilities and on open temporary storage sites which were largely built in the 1960s and 1970s and do not fully meet the requirements for environmental protection. High-level wastes are stored in special facilities only. The amount of solid radioactive wastes stored in facilities of the Northern Fleet is estimated to be 8000 m^3 with a total activity not exceeding 37 TBq. On average, 1000 m^3 of solid radioactive wastes are generated annually. Considering the increasing rate of submarine decommissioning, the rate of generation of radioactive waste may increase by at least a factor of two. The major storage sites for solid radioactive wastes in the Northern Fleet are Andreev Inlet, Iokan'ga and Polyarny. By volume, 50% of the solid wastes are combustible, 15% are compressible, 35% are non-compressible and 1% are spent ion-exchange filter resins. No processing of solid radioactive wastes currently takes place by the Navy. Numerous plans and projects to build and reconstruct facilities for radioactive waste storage and processing have not yet been completed and/or implemented because of limited financial resources (Petrov 1995, NEFCO 1996).

The chemical and radionuclide composition of the radioactive wastes is complex and changes with time. It depends upon design features of the nuclear propulsion units, leakage rates from fuel rods and the age of the wastes. Isotopes within the wastes include both fission and activation products, mainly ^{54}Mn, ^{60}Co, ^{90}Sr, and ^{137}Cs (RCRA 1997).

A Russian Federal Program, approved by the government in 1995, will significantly improve the handling and disposal of radioactive waste and spent nuclear fuel including decommissioning of naval nuclear reactors between 1996 and 2005. However, measures are being implemented slowly because of contemporary economic problems in Russia (Ecological Safety 1996).

8.5.2.3.4. **Shipyards**

The largest Russian shipyard *Sevmash* is located in Severodvinsk on the coast of the White Sea. The ship repair plant *Zvezdochka* is located in the same area. Nuclear submarines have been built, serviced, repaired and refitted at these shipyards for 35 years and they are now also decommissioned at these locations. During these operations large amount of solid and liquid radioactive wastes are generated which are stored in stationary and floating storage facilities. In the shipyard, there are six floating hull sections containing reactors following spent fuel removal, 3370 m^3 of solid radioactive waste in the stationary storage facility and 1250 m^3 in an interim storage site. Some 950 m^3 liquid radioactive wastes are stored in on-land and floating containers (Koupri 1995).

Also at these shipyards, 15 nuclear submarines with 29 reactors, from which the fuel has been unloaded (more than 7000 fuel assemblies), are being repaired. After unloading, the spent fuel is sent to Mayak for reprocessing. However, the rate of spent fuel accumulation is higher than the rate of reprocessing. As a result, the amount of spent fuel in storage continues to increase.

Monitoring around the facilities in Severodvinsk in the 1990s (Table 8·36) showed that there is no cause for any major concern about environmental contamination (Koupri 1995).

Table 8·36. Radionuclides in environmental media in the vicinity of Severodvinsk shipyards.

Environmental medium	Radionuclide concentration (range)
Atmospheric aerosols	2×10^{-4}-11×10^{-4} Bq/m^3
Atmospheric deposition	2-15 MBq/m^2 per month
Seawater in shipyard areas	4-9 Bq/m^3
Bottom sediments	2-11 Bq/kg dw

The progressively increasing accumulation of spent nuclear fuel and radioactive wastes stored partly in interim storage facilities poses a threat of contamination of both terrestrial and marine environments in the case of accidents (Koupri 1995, NEFCO 1996).

Operational releases from these civilian and military vessels do not appear to have been documented, but are probably relatively small. Finally, it should be noted that military nuclear-powered vessels from other countries such as USA, France, Canada and UK can transit Arctic waters.

8.5.2.4. **European nuclear fuel reprocessing plants**

While there are no nuclear fuel reprocessing plants in the Arctic, radionuclide releases to the marine environment of western Europe can be transported to the Arctic by ocean currents.

Table 8·37. Contributions (%) made by European nuclear fuel reprocessing plants to total discharges within the European Community (Bq), for discharges up to the end of 1984.

Facility	% of total discharge		
	α-emitters	β-emitters	Tritium
Sellafield	95.2	86.9	52.9
La Hague	0.52	5.3	16.4
Dounreay	1.8	6.7	0.33
Total (%)	97.5	98.9	69.6
Total EC discharges, Bq	1.4×10^{15}	1.5×10^{17}	4.6×10^{16}

The principal sources of nuclear waste discharges to the marine environment from western Europe are nuclear fuel reprocessing operations at Sellafield in Cumbria on the west coast of England, at La Hague near Cherbourg, France, and at Dounreay in the northeast of Scotland (see Table 8·37) (after CEC 1990). These plants are all involved in the recovery of plutonium from irradiated nuclear fuel for further use in the electrical generating industry. While there are also reprocessing operations at Marcoule, France, which discharges to the Rhone, and at Karlsruhe in Germany, these are of minimal significance in relation to the impact of European reprocessing operations on the North Atlantic, northern Europe and the adjacent Arctic. Accordingly, the latter two reprocessing plants have not been considered in this summary.

Authorization for releases from such operations are based on limiting radiation exposures to the most exposed individuals for a particular plant. The potentially exposed individuals of primary concern are those in the vicinity of the plant and the area of its releases to the environment. Thus, exposures to distant individuals, such as residents of the Arctic, are not directly relevant to the authorization process for European nuclear fuel reprocessing operations.

Although collective doses to larger populations have not been used in the context of authorizing nuclear fuel reprocessing plant operations and releases, collective dose rates and commitments associated with such plants within western Europe are useful for assessing the large-scale radiological impact of such activities. Thus, collective dose estimates provide a useful framework for considering the extent of radiological consequences and associated risks to Arctic residents resulting from European nuclear fuel reprocessing operations.

It is worth noting, in an Arctic context, that the radionuclide compositions of the discharges from European reprocessing plants have altered significantly during their period of operation. Although Sellafield remains the primary contributor to activity releases among the three plants, the dominant sources of individual radionuclides have, in some cases, altered dramatically. This is particularly well illustrated by the relative contributions made to releases of the long-lived radionuclide ^{129}I by Sellafield and La Hague. While the aggregate discharge of ^{129}I remained relatively constant from 1975-1990, and remains so for Sellafield, the increased fuel reprocessing throughput at La Hague has resulted in substantially increased discharges of this radionuclide making La Hague currently the dominant contributor (90%) to the aggregate ^{129}I discharge (Yiou *et al.* 1995). From a scientific perspective, this is an important observation because of the relatively conservative nature of iodine in seawater and the important contribution of this isotope to Arctic waters through transport into and through the Norwegian Coastal Current into the Arctic making it, in combination with other isotopes, a valuable oceanographic tracer.

This section deals with the history of releases from European nuclear fuel reprocessing plants, the basis under which such releases are authorized, and individual and collective dose reconstruction. This provides a basis for assessing the degree to which these operations might be of concern relative to their effects on the Arctic and its residents. In this discussion, only exposures to humans have been considered. This is because no evidence exists to suggest that doses to organisms resulting from nuclear fuel reprocessing activities in western Europe would be of any concern beyond the vicinity (i.e., the receiving area of releases) of particular reprocessing operations.

8.5.2.4.1. **British nuclear fuels plant at Sellafield, UK**

The nuclear fuel reprocessing plant at Sellafield (formerly Windscale) on the eastern side of the Irish Sea is the largest nuclear complex in the United Kingdom and is operated by British Nuclear Fuels plc (BNFL). Discharges from this site started in 1951 when the facility was initially put into operation by the UK Atomic Energy Authority. The UK government has been assiduous both in maintaining records of the basis of its discharge authorizations (e.g., HMSO 1959) and providing, for public information, the results of its monitoring associated with civil nuclear site activities through the medium of reports issued by the UK Atomic Energy Authority and the Ministry of Agriculture, Fisheries and Food (MAFF 1967). The first of this latter series of reports reflects the commitment to protecting the most exposed individual from exposures exceeding the ICRP dose limits and the commitment to what became known subsequently as the principle of optimisation. These same reports also contain results relevant to estimating potential exposures of residents of the Channel Islands as a result of releases from the La Hague reprocessing plant.

Until 1954, when formal authorisation of such releases occurred under the Atomic Energy Authority Act (UK), discharges took place under controls implemented by the site operators in consultation with the Government Departments (primarily the Ministry of Housing and Local Government). Operation of the facility was largely transferred from the UK Atomic Energy Authority to British Nuclear Fuels in April, 1971. Throughout, authorizations have been based on limiting exposures to individuals in the most exposed group to the prevailing values recommended by the ICRP (Wix *et al.* 1960, Hunt 1995). Discharges from the site have fluctuated considerably over time with a maximum for most radionuclides occurring in the mid- to late 1970s as a consequence of changes in practice and the introduction of additional measures to reduce discharges introduced predominantly in the mid-1970s (Gray *et al.* 1995, Kershaw and Baxter 1993a, 1993b). The nuclides representing the major proportion of the total releases from Sellafield (ca. 130 PBq excluding 3H up to 1986) are the beta-/gamma-emitters ^{137}Cs (30%), ^{106}Ru (21%), ^{241}Pu (16%) and ^{95}Zr/^{95}Nb (18%) (CEC 1990); CEC 1990 reported the ^{95}Zr/^{95}Nb value as 41%, but this number appears inconsistent with the record of discharges from Sellafield, cf. Figure 8·56. Total alpha-emitter releases in the same period were about 1% of the total and are dominated by ^{239}Pu (0.0053%) and ^{241}Am (0.0041%) (CEC 1990). The chronology of releases, both to sea and to the atmosphere, from the Sellafield reprocessing operation has been most recently and comprehensively documented by Gray *et al.* (1995). Figures 8·56 and 8·57 show the rates of different liquid discharges from 1952-1992, reproduced from Gray *et al.* (1995).

Environmental measurements and modeling have been used to confirm the chronological record of liquid discharges from Sellafield with emphasis given to measurements in components of critical pathways of exposure (Gray *et al.* 1995).

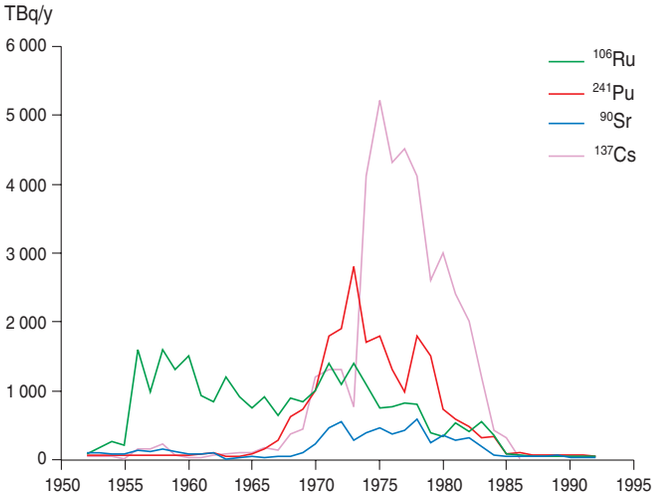


Figure 8·56. Discharges of principle beta-gamma emitters in liquid effluent from Sellafield, 1952-1992 (after Gray *et al.* 1995).

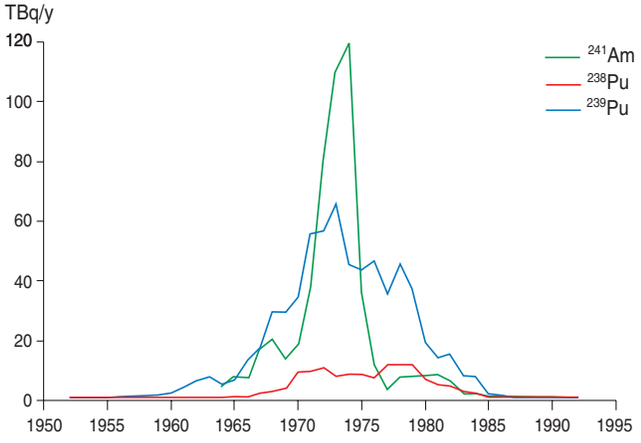


Figure 8·57. Discharges of principle alpha emitters in liquid effluent from Sellafield, 1952-1992 (after Gray *et al.* 1995).

Initial assessments showed that the pathways likely to give rise to the largest exposures were internal exposures associated with the consumption of fish and seaweed and external exposures associated with shoreline occupation. Figure 8·58 presents a comparison of doses to individuals within critical groups over the period of releases from Sellafield up to 1993 after Hunt (1995). It should be appreciated that the application of, and values for, the dose limits applicable to the authorization of practices, as recommended by the ICRP, have changed during the period concerned. Also, changes in the characterization of the local critical group, the rates of consumption of seafood and changes to the dosimetric data for actinides have given rise to changes in the estimates of critical group exposures. Nevertheless, it is clear that the recent doses to individuals within the critical group arising from liquid releases from Sellafield are relatively small (of the order of 200 µSv/y) and well below the currently recommended ICRP individual dose limit of 1 µSv/y and the constraint for doses arising from this category of source. Furthermore, as all of the most exposed individuals are in population groupings situated in the general vicinity of the site, individual doses to more distant members of the public, such as residents of the Arctic, will be much lower.

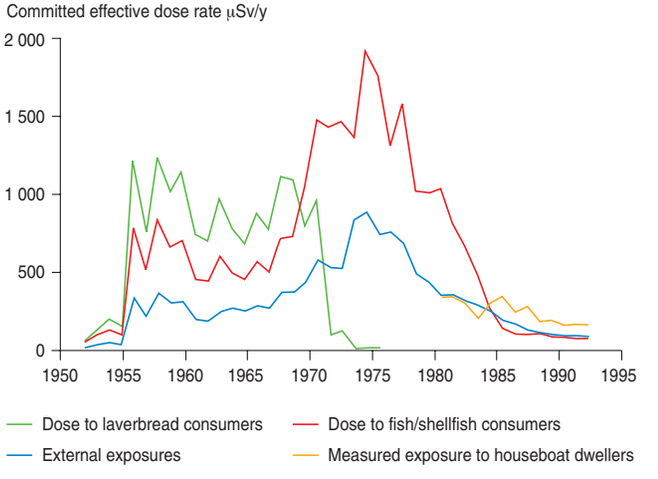


Figure 8·58. Comparison of doses to critical groups for Sellafield discharges from 1952 to 1993 (after Hunt 1995).

8.5.2.4.2. La Hague, France

The reprocessing facility at Cap de la Hague was brought into operation in 1965. Major modifications have been made to process additional types and amounts of spent fuel on two subsequent occasions. The total discharges from this site were, excluding ^{3}H, about 8 PBq up to 1986; much less than those from Sellafield. Releases of individual radionuclides up to 1982 were given by Calmet and Guegueniat (1985) and these were cited and extended to 1986 in the 'Project Marina' report (CEC 1990). In contrast to the contributions made to activity discharges from Sellafield by individual radionuclides, in which ^{137}Cs has been the dominant contributor up to the mid-1980s, for La Hague the dominant contributor to activity releases (55%) is the beta-emitter ^{106}Ru followed by ^{90}Sr rather than ^{137}Cs.

As in the case of Sellafield, authorization of discharges from La Hague is based on limiting the doses to individuals within potentially critically exposed groups to the dose limits for members of the public recommended by the ICRP. A discussion of the identities and habits of the critical groups for doses arising from discharges from La Hague is given in Calmet and Guegueniat (1985), together with calculated proportions of the dose limit for fishermen via consumption and external exposure pathways. It is difficult to assess, from this reference, the quantitative importance of these pathways of individual exposure. However, if it is assumed that, consistent with international recommendations at the time, the individual dose limit for members of the public used in France was 5 mSv/y, the dose to individuals within the critical group resulting from sea discharges was of the order of 0.2 mSv/y. Critical group doses associated with liquid releases from La Hague were reassessed for the period 1982-1986 in 'Project Marina' (CEC 1990). This reassessment yields individual dose rates of 0.03-0.04 mSv/y for the seafood consumption pathway and 0.06-0.27 mSv/y for external exposures over the five-year period.

8.5.2.4.3. Dounreay, UK

Releases from Dounreay are much smaller than those from Sellafield. The aggregate activity discharged up to 1986 was about 10 PBq with $^{95}Zr/^{95}Nb$ representing 55% of this activity; ^{144}Ce, 17%; and ^{106}Ru, 10% (CEC 1990). Discharges of beta-activity was at its highest during the 1960s and early 1970s with small peaks in 1968 and 1973 resulting from plant washout and decontamination procedures. In the early years of operation of the Dounreay reprocessing plant, some of the irradiated fuel, which was derived from the Dounreay Fast Reactor, was of higher specific activity than that processed by any similar plant. In 1980, fuel from the Prototype Fast Reactor began to be reprocessed, and this gave rise to increased discharges of alpha-emitters, mainly ^{238}Pu, ^{239}Pu

and ^{241}Am. The critical groups for individual exposures from Dounreay discharges are fishermen handling fixed salmon nets contaminated by sludge, where dose to hands from beta-emitters is the primary route of exposure, although attention was also given to seafood consumption by fishermen and their families, crustacean consumption by workers in a local canning factory and, potentially, occasional occupants of inlets who might receive external exposures from contaminated sediments (Freke *et al.* 1969).

8.5.2.4.4. Dose reconstruction for releases from Western European reprocessing plants

For the purposes of uniform and consistent comparison, the predominant source used for dose reconstruction information has been 'Project Marina' (CEC 1990). This latter review deals with all sources of radionuclides that give rise to exposures to the European Community population and this further justifies the use of this source for broader consistency in assessing both individual and collective doses within a larger context. 'Project Marina' concluded that 'the quality and detail of environmental monitoring data from nuclear site operators, governments and other sources were more than adequate for assessing critical group doses.' In this context, discharges from reprocessing plants in general, and Sellafield in particular, gave rise to the highest critical group doses arising from nuclear site discharges. During the period 1977-1986, critical group doses, albeit delivered to only a few individuals consuming above average amounts of seafood harvested near to the discharge point, were up to 3.5 mSv/y. Nevertheless, ICRP dose limits have been met as exposures were generally less than 1 mSv/y and exposures exceeding this value did not occur for long enough for lifetime exposure to have exceeded 1 mSv/y on average. Discharges from Sellafield have declined since the mid-1970s and this has led to a decline in critical group doses. In 1986, critical group exposures from Sellafield discharges were about 0.3 mSv. The corresponding doses from the other two European reprocessing plants at La Hague and Dounreay have been substantially less than those arising from Sellafield. The individual effective dose equivalents for La Hague are less than 0.3 mSv/y, and for Dounreay less than 0.05 mSv/y. It is again stressed that these exposures are to members of critical groups resident in the immediate vicinity of the respective reprocessing plant and not to Arctic residents.

Collective dose estimation allows an improved basis to assess the consequences of nuclear fuel reprocessing operations on a larger scale and in an appropriate context to the doses arising from other human activities. The collective dose commitment to the population of the European Community, truncated in the year 2500, resulting from total discharges up to the end of 1984 from nuclear fuel reprocessing operations in western Europe is estimated to be 5150 manSv with the dominant contribution (90% or 4600 manSv) from Sellafield and progressively smaller contributions from La Hague (8% or 430 manSv) and Dounreay (2% or 120 manSv) (CEC 1990) (Figure 8·59). The predominant radionuclide contribution to the collective exposure is ^{137}Cs. About 80% of the collective dose commitment to the European Community population had already been delivered by the mid-1980s. It is also worth noting that the aggregate collective dose from European reprocessing operations essentially accounts for all the collective dose commitment from civilian nuclear activities in the European Community that is estimated, on the same basis, to be 5300 manSv.

The peak collective dose rates for Sellafield, La Hague and Dounreay discharges were of the order of 300 manSv/y

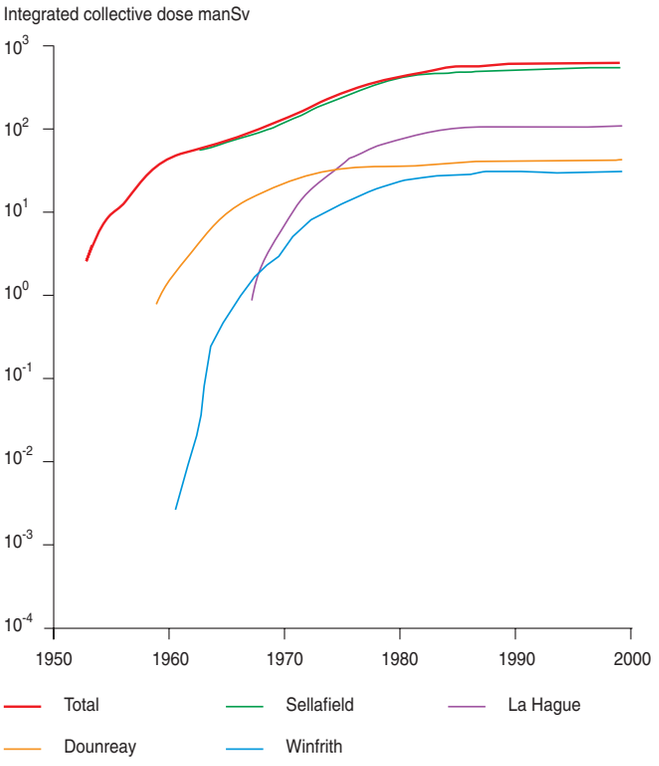


Figure 8·59. Collective dose commitment from nuclear fuel reprocessing activities (after CEC 1990).

(peak in the late 1980s), 40 manSv/y (peak in the mid-1980s) and 7 manSv/y (peak at the beginning of the 1970s), respectively. Collective dose rates have declined substantially since then and can be projected (based on aggregate discharges up to the end of 1986) to be of the order of 20 manSv/y, 0.6 manSv/y and 0.3 manSv/y, respectively, in the year 2000

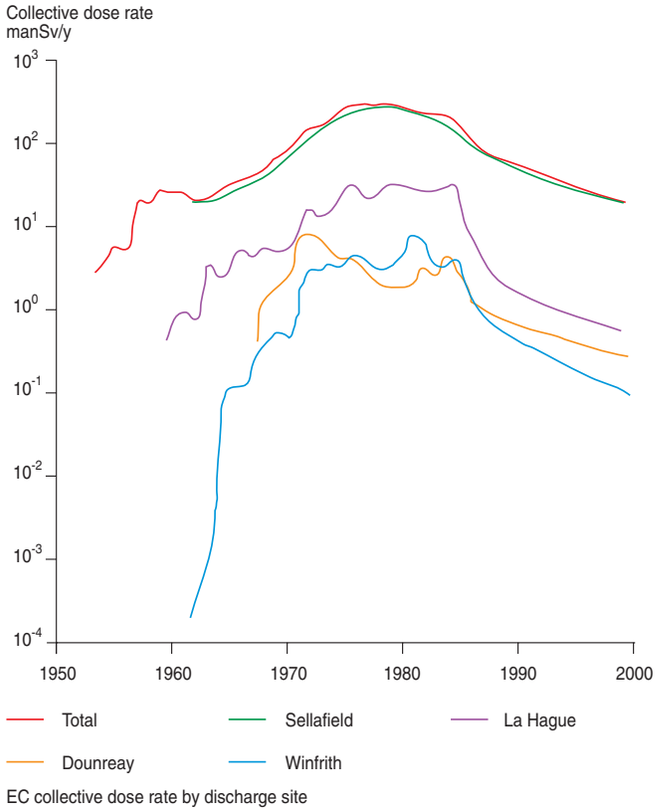


Figure 8·60. Collective dose rates from European nuclear fuel reprocessing plants based on releases up to 1986 (after CEC 1990).

(Figure 8·60) (CEC 1990). Recognising that the major marine contributor to collective dose for the European Community population arises for nuclear fuel reprocessing activities, particularly at Sellafield, and that the seafood consumption pathway is the dominant exposure pathway for the general population, it is worth noting that the collective dose rate to the non-EC population arising from the consumption of seafood caught in EC waters is about an order of magnitude below that to the EC population (CEC 1990).

Given the European Community population of 3.2×10^8 at the time of 'Project Marina' (CEC 1990), the contemporary per capita representation of the total collective dose rate from nuclear fuel reprocessing operations is approximately 1.1 μSv/y. Although this could result in approximately 15 additional cancer deaths within the European Community per year of such exposure, such deaths would not be epidemiologically detected against the normal death rate in the Community. Furthermore, the associated incremental risk of $\approx 5 \times 10^{-8}$ per year of serious health defect (i.e., fatal cancer induction) to an average individual in the European Community per year of practice is relatively trivial (IAEA 1993). The representation of the collective dose rate arising from European nuclear fuel reprocessing operations on a per capita basis for the entire global population yields an incremental individual risk of fatality of about 7×10^{-9} per year (IAEA 1993). This may be a reasonable representation of the average risk to members of the Arctic community, spread as they are throughout northern Europe, Asia and North America, even allowing for the contributions from other, non-European, nuclear fuel reprocessing waste releases.

8.5.2.5. **Russian nuclear fuel reprocessing plants**

In Russia, fuel reprocessing takes place at the Mayak Production Association, Chelyabinsk; and at plants in Krasnoyarsk and Tomsk. Although these plants are not situated in the Arctic, radioactive releases from Mayak and Tomsk enter the drainage area of the Ob River and those from Krasnoyarsk directly into the Yenisey River. Releases to these rivers can be transported downstream, ultimately into the Kara Sea.

8.5.2.5.1. **Mayak**

The nuclear enterprise *Mayak* Production Association (*Mayak* PA) is located outside the town Ozyorsk (until recently called Chelyabinsk-65) between the cities of Yekaterinburg and Chelyabinsk, just east of the Ural mountains. *Mayak* PA was the first plant in the former Soviet Union established for the production of nuclear weapons material and began operation in 1948. The plant had five special nuclear reactors for the production of ^{239}Pu and a facility for the separation of plutonium as weapons material.

Very large quantities of radioactive waste resulted from this operation. The liquid radioactive wastes were released to the River Techa. In all, about 100 PBq were released, including 95 PBq in 1950/51. ^{90}Sr and ^{137}Cs constituted 12% of the activity of the radioactive mixture which included significant amounts of ^{95}Zr/Nb, ruthenium isotopes, and other fission products. The concentration of radionuclides in water of the upper Techa River (reservoir 5) in 1950-1951 reached an average of 1 MBq/L. The population of the villages along the river, numbering 28 000, was subjected to external exposures from water, water-meadows and irrigated kitchen-gardens. The water of the River Techa was used by the population for drinking, watering cattle and fishing, etc. Although a considerable portion of the population along the River Techa was evacuated in 1953-1960, inhabitants of its upper part received average doses to the wholebody of over 1 Gy during 25 years, and some individuals received extreme doses of more than 2 Gy. Specific features of the exposure of inhabitants of the River Techa are the enhanced doses to red bone marrow and bone surface that exceed the wholebody dose by a factor of 1.5-15. High levels of exposure of bone marrow caused tens of cases of chronic radiation sickness in inhabitants of the upper Techa River in the 1950s and a higher incidence of leukaemia over longer time periods (Degteva *et al.* 1992, Kozheurov *et al.* 1994, UNSCEAR 1993).

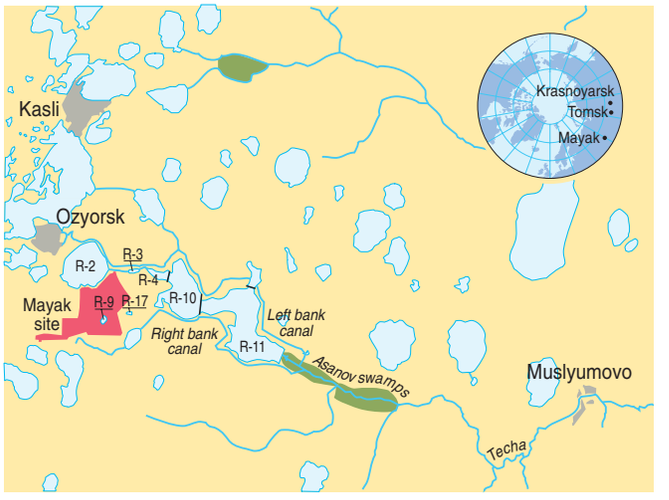


Figure 8·61. The system of dams and drainage channels at Mayak.

Between 1951 and 1966, a system of dams was constructed along the upper part of Techa River, creating several artificial water reservoirs along the old river bed to retain most of the radioactivity (Malyshev 1995). The river water, originating from Lake Irtysh, is today led outside the reservoirs via a canal on the northern side of the dam system (left bank canal). Similarly, the water of the Mishelyak River flows in a right bank canal on the southern side. The two canals join the Techa River about 3 km downstream of dam no. 11, as depicted in Figure 8·61.

It is estimated that the accumulated activity in reservoirs nos 3, 4, 10 and 11 amounts to about 9.8 PBq (0.26 MCi), mainly in the sediments (see Table 8·38). The activity contained in the several reservoirs, including Lake Karachay (reservoir no. 9), are presented in Table 8·38 (Mayak 1993). In the floodplain of the upper Techa River (the Asanov Swamp) about 37 TBq (1000 Ci) of ^{90}Sr and 185 TBq (5000 Ci) of ^{137}Cs is retained in the surface soil (Malyshev 1995, Romanov 1995). The maximum contamination of the floodplain (8 TBq ^{90}Sr/km^2 and 185 TBq ^{137}Cs/km^2) is comparable to the maximum contamination following the *Kyshtym* accident (75 TBq ^{90}Sr/km^2).

Since 1951, radioactive waste has mainly been discharged into Lake Karachay, a small lake of about 150 000 m^3 volume with no outlet, at a total amount of about 4400 PBq (120 MCi) of which about 80% is ^{137}Cs. The annual discharge has now been reduced to about 10 PBq (0.3 MCi)

Table 8·38. Inventories of activity contained in some Mayak storage reservoirs, decay corrected to 1994.

Reservoir no.	Activity contained in reservoir, PBq (kCi)
3	1.6 (44)
4	0.27 (7.3)
10	6.7 (180)
11	1.2 (32)
9 (Karachay)	4400 (1.19×10^5)
17	74 (2000)

(Glagolenko 1995, Malyshev 1995, Romanov 1995). Spring flooding has contaminated large parts of the Asanov Swamp along the river banks of upper Techa River. As described above, the Asanov Swamp has retained much of the discharged activity. However, some of the discharged radionuclides, especially mobile radionuclides such as ^{90}Sr, have probably been carried in aqueous phase more than 2000 km downstream to the Kara Sea.

Within the Asanov Swamp, activity decreases with distance from the dam of reservoir no. 11. Whereas, during the period of direct discharges to the Techa River, the swamp acted as a filter retaining radionuclides, it now acts as a source for slow release. Radiostrontium (^{90}Sr) is remobilized more readily than ^{137}Cs, with a peak concentration in river water during the period of spring flooding. However, the annual release of ^{90}Sr as a proportion of the total inventory in the swamp has been decreasing with time. The annual ^{90}Sr discharge has fallen from 10-15% of the total inventory of the swamp in the early 1960s to 3-4% in the early 1990s. It has been estimated that current releases of ^{90}Sr from the Asanov Swamp are about 20-30 Ci/y (≈0.7-1.1 TBq/y) and future releases are predicted to be ≈15-30 Ci/y (≈0.5-1.1 TBq/y) (NRPA 1997).

Current operational releases are considerable lower following the shut-down, in 1990, of the last of the five uranium-graphite reactors that produced weapons grade plutonium. Current operational releases are about 20 000 m^3/y of waste corresponding to an activity of 370-746 MBq/y entering Lake Karachay.

Large amounts of high-level radioactive waste have been vitrified at the special facility at Mayak. The stored vitrified wastes at Mayak currently contain about 8000 PBq of long-lived radionuclides (Russian Federal Program 1995).

8.5.2.5.2. **Tomsk-7**

The Siberian Chemical Combine at Tomsk-7 or Seversk, is one of the largest nuclear weapons production facilities in the world. The site contains five graphite-uranium plutonium production reactors, a uranium enrichment plant, a reprocessing plant and other plants engaged in the military nuclear materials cycle. Three of the reactors have been shut down. The remaining two are now dual-purpose plants that also provide heating and electricity for the towns of Tomsk and Seversk. Tomsk-7 came to international attention in April, 1993, when a chemical reaction caused an explosion in a tank containing uranium nitrate solution. The plant adjoins the River Tom that ultimately drains into the River Ob. Since 1956, contaminated cooling water has been discharged to the river. It has been estimated that the inventory of radioactivity remaining in the River Tom in 1995 was about 3.7×10^4 Ci (≈1.4 PBq).

A recent report of the Russian Federation Security Council states that the total inventory of radioactive wastes within the industrial zone of the site is estimated to be 44 000 PBq. The majority of this waste is in the form of liquid radioactive waste, part of which was discharged into several reservoirs (estimate 5000 PBq). In addition to surface discharges, Tomsk-7 is one of two sites in Russia where underground injection has been used as a means of disposal for large volumes of waste, estimated in the early 1990s to be about 15 000 PBq (RCRA 1997).

8.5.2.5.3. **Krasnoyarsk-26**

The Krasnoyarsk Mining and Chemical Combine, formerly known as Krasnoyarsk-26 and now renamed Zheleznogorsk, is situated on the Yenisey River. This combine consists of three RBMK-type graphite-moderated reactors, a reprocessing plant for the production of weapons-grade plutonium, and storage facilities for radioactive wastes. Two of the reactors have been shut down since 1992; the third reactor is operated as a dual-purpose reactor, supplying heat and electricity to the region. Construction of a new reprocessing facility was initiated in 1983, but suspended in 1989 as a result of public opposition and economic problems. The President of Russia has since issued a decree calling for the continuation of construction of this plant which, when completed, could treat both domestic and foreign spent fuel.

Most of the radioactive waste originating from the Combine reprocessing activities has been stored in large stainless steel tanks (ca. 8000 PBq) (Lebedev *et al.* 1996) or injected into the ground within the site boundary. A total of between 7×10^8 and 1×10^9 Ci (26 000-37 000 PBq) at time of disposal have been injected. The current activity of this waste is estimated at 4000 PBq (RCRA 1997). The activity has been transported to the injection site via a reportedly leaky pipeline that has spilled an unknown amount of radioactive waste along its path to the injection site (Bradley and Jenquin 1995).

8.5.2.5.4. **Assessment of river transport and associated doses**

River transport of radionuclides occurs through runoff from catchment areas contaminated by global fallout, discharges from nuclear installations and accidental releases. In the Yenisey drainage basin, the main source, apart from global fallout, is previous releases from nuclear installations at Krasnoyarsk. For the Ob, the main sources originate within the following tributary river systems: Techa – Iset – Tobol – Irtysh (contaminated by discharges from Mayak since 1948, the Kyshtym accident in 1957, and airborne release from Lake Karachay in 1967); Karabolka – Sinara – Iset – Tobol – Irtysh (contaminated by the Kyshtym accident in 1957); and Romashka – Tom (contaminated by discharges and accidental releases from the Siberian Chemical Plant, Tomsk-7).

Total fluxes of radionuclides to the Arctic seas through river discharges are difficult to estimate, especially prior to 1961. Data in the open literature is still inconsistent with respect to direct discharges from Mayak to the Techa River during the period 1949-1952. Recent investigations of the vertical distribution of ^{137}Cs and plutonium isotopes in dated sediment profiles from the Ob estuary reflect signals of ^{137}Cs from global fallout only (Panteleyev *et al.* 1995). However, other radionuclides, such as ^{90}Sr, may have been transported from Mayak to the Kara Sea because of their higher aquatic mobility.

During recent years, substantial new information has become available on river transported radionuclides, as well as on past and present sources, especially Mayak PA. Based on annual mean concentrations (Chumichev 1995) about 1.5 PBq ^{90}Sr were estimated to have been transported to the Kara Sea during 1961-1990 by the Ob (0.65 PBq), the Yenisey (0.37 PBq), the Severnaya Dvina (0.10 PBq), the Pechora (0.08 PBq), the Lena (0.29 PBq) and the Indigirka (0.02 PBq) Rivers. Several recent investigations in the tributary system of the Ob River confirm that the highest activity in sediments is found in the upper Techa River; in reservoirs, in the Asanov Swamp and in areas contaminated from the Kyshtym accident (Trapeznikov *et al.* 1995, Christensen *et al.* 1995, Romanov 1995). Radioisotopes of caesium are strongly associated with sediment and soil components, while ^{90}Sr is comparatively mobile (Tronstad *et al.* 1995). There are no available data defining how much radioactive contamination has been transported by rivers into the Arctic

marine environment in the period up to 1961. As a result, it is not possible to reconstruct doses for river-transported radionuclides.

8.5.2.6. **Mining activities**

The only country known to have mining activities in the Arctic of potential radiological significance is Canada that has a single uranium mining operation at Baker Lake north of 60°N. This is an entirely exploratory venture and no application for a license to operate a mine has yet been submitted. Any approval would be contingent on the provision of an Environmental Impact Assessment. The licensing requirements of the Atomic Energy Control Board of Canada impose a dose limit of 5 mSv/y for members of the public. There is no national policy of reducing doses below the dose limit (i.e., the implementation of optimisation) as there is for nuclear power plants.

8.5.3. Accidental releases

There have been several historical radiation accidents that constituted sources of contamination of the Arctic environment with artificial radionuclides. These events began with the Kyshtym accident in 1957, followed by the Lake Karachay accident in 1967, the spillage of plutonium from nuclear weapons at Thule in 1968, the re-entry of the Soviet Union's Cosmos-954 satellite over Canada in 1978, the Chernobyl accident in 1986 and the accident at Tomsk-7 in 1993. There were also a series of accidents involving Soviet nuclear submarines in 1961, 1968, 1978 and 1989. These accidents and their consequences are discussed in the following sections.

8.5.3.1. **The accidents at the Mayak weapons production plant in 1957 and at Lake Karachay in 1967**

In addition to intentional discharges into the River Techa during the period 1949-1951 described in section 8.5.2.5.1, two major accidents have contaminated the areas surrounding the Mayak reprocessing plant.

The Kyshtym accident, 1957

In the *Kyshtym* accident in 1957, a storage tank of highly radioactive material exploded. Of the 740 PBq (20 MCi) in the waste tank, about 90% settled in the immediate vicinity of the site of the explosion. The remaining 74 PBq (2 MCi) was dispersed by the wind and subjected to deposition.

The explosion of the tank containing highly-active wastes at the Mayak plant in the Urals resulted in release of about 74 PBq of a radioactive mixture to the atmosphere to a height of up to 1 km. ^{144}Ce/Pr and ^{95}Zr/Nb were the dominant short-lived radionuclides while the dominant long-lived radionuclide, ^{90}Sr, represented 5.4% of the total activity. The radioactive footprint of the ^{90}Sr distribution in soil had a scale length of 300 km. A population numbering 270 000 was subjected to external exposure by gamma radiation from ^{95}Zr/Nb, internal exposure of the intestinal tract from ^{144}Ce/Pr and of red bone marrow and bone surfaces by ^{90}Sr/Y. Figure 8·62 shows the dynamics of accumulation of the effective dose and of its main components in adult inhabitants, normalized to the density of soil contamination with ^{90}Sr of 1 Ci/km^2 (37 GBq/km^2). The contribution of ^{90}Sr to the effective dose became dominant 1-2 years after the decay of other radionuclides.

About 10 000 persons who received a collective dose of 1300 manSv were evacuated 7-670 days following the accident. The average effective dose in the most exposed group of inhabitants reached 0.5 Sv. A significant portion of ^{90}Sr could have been transported to the Arctic Ocean via the Ob River drainage system but this flux could not be distinguished from other Mayak releases. The present level of radioactivity from this atmospheric release is estimated at 44 000 Ci (≈1.6 PBq) with the main constituent being ^{90}Sr.

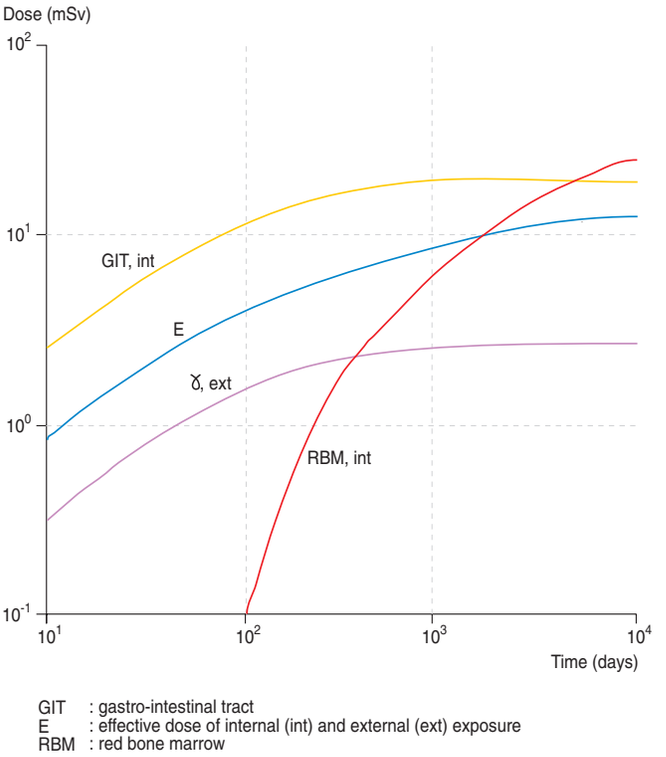


Figure 8·62. Dose accumulation in adult rural residents after the Kyshtym accident in 1957.

Lake Karachay, 1967

In 1967, ten years after the storage tank explosion, Lake Karachay partially dried out during a long summer period without precipitation and about 16 TBq of radioactive dust, mainly contaminated with ^{137}Cs and ^{90}Sr, was spread with the wind, contaminating parts of the same area as the Kyshtim accident. In this case, however, levels of environmental contamination were lower: the area within the isopleth of 4 kBq/m^2 of ^{90}Sr was 1800 km^2 (Malyshev 1995, Romanov 1995).

8.5.3.2. **The Thule nuclear weapons accident in 1968**

On 21 January, 1968, a B-52 aircraft from the US Strategic Air Command (SAC) attempted an emergency landing at Thule Air Base following a fire in the on-board electrical system (Risø-R 213, 1970). All aircraft electrical power was lost and the aircraft crashed on the sea ice of Bylot Sound 11 km west of the Thule Air Base. The aircraft disintegrated on impact and an explosion and fire ensued. The crew bailed out before impact. Six survived but one died. The aircraft carried four nuclear weapons all of which were destroyed by the impact when the conventional high explosive charges in the unarmed bombs detonated. This resulted in dispersion of fissionable material viz. plutonium and, presumably, uranium. Only the aircraft engines were not totally destroyed. There was speculation that parts of the aircraft might have plunged through the ice but this was shown later not to have been the case. Ice cores from an area of approximately 100 m in diameter were contaminated. Outside this zone, conta-

mination was found on the surface only. Contamination from the accident can be divided in four categories:

- Plutonium carried aloft in the cloud from the explosion and fire and dispersed regionally or even globally by the prevailing meteorological conditions.
- Plutonium deposited on the ice and snow surfaces locally.
- Plutonium deposited on aircraft and weapons debris.
- Plutonium in and beneath the ice at the point of impact.

It has been impossible to determine the exact amounts of plutonium in these four categories. However, from a local point of view, it was important to know the amount, form and association of plutonium (and tritium) on the surface in the immediate vicinity of the crash site especially at the point of impact where decontamination operations were technically feasible. Most of the estimated amount of plutonium (3.5 ± 0.7 kg or 8.8 TBq) on and in the ice at the crash site was found within a black teardrop-shaped area of 700×150 m^2 (0.11 km^2). The amounts of plutonium recovered with aircraft and weapons debris is not included in this 3.5 kg. Recent information provided to the Government of Denmark by the U.S Department of Energy disclosed that the Thule weapons contained a total of 6 kg of plutonium. The plutonium contamination was in the form of oxide particles with a very wide size distribution, the median diameter being 2 μm. The plutonium particles were often associated with larger particles of low density inert material.

It was decided to remove as much of the contamination from the sea-ice as possible. First, the weapons and aircraft debris were recovered. A month after the accident, this phase was completed and the weapons parts were returned to the United States while aircraft wreckage was stored for future disposal.

The next decontamination step involved scraping the contaminated area of snow and ice with heavy road machinery. 'Hot spots' were monitored and removed by shoveling. The contaminated snow and ice was placed in sixty-seven 25 000-gallon tanks (total volume: 6300 m^3). In September, 1968, these tanks were shipped to the Savannah River Plant in the USA. The clean-up was very efficient and it was estimated that only approximately 1 TBq (±50%) of plutonium remained on the ice. During the summer months (June-July) the sea ice broke up and started to melt. It was observed that the ice from the crash site drifted northward. However, when and where the ice contamination was released to the sea was not observed.

From measurements of plutonium in marine sediments collected during expeditions to Thule in 1968, 1970, 1974, 1979 and 1984 (Aarkrog 1971, 1977, Aarkrog *et al*. 1984, 1987, Smith *et al*. 1994) it was calculated that about 1 TBq or 0.5 kg of plutonium was deposited on the bottom of Bylot Sound from the Thule accident. The amount of plutonium left on the ice after the decontamination effort in 1968 was estimated to be 1 TBq (±50%). It therefore seems likely that the majority of the plutonium in the sediments comes from the melting of the sea-ice. On the other hand, it is also evident that the highest levels are found beneath the point of impact and, because some of the contaminated ice drifted away before it melted, it seems likely that some debris entered the sea directly through the impact hole in the ice.

The doses from possible consumption of marine biota from Thule were calculated from environmental analysis of plutonium in bivalves, shrimps, fish, seabirds, seals and walrus collected at Thule following the accident. To obtain the integrated ^{239,240}Pu intake over a given period, the time-integrated activity of plutonium (Bq y/kg) was multiplied by the annual individual consumption rate (kg/y) of the product.

Table 8·39. Dose commitments from annual intakes of 1 kg of marine product contaminated by ^{239,240}Pu from the Thule accident (Aarkrog 1995).

Sample	Dose commitments 1968-1995, μSv
Bivalves	7
Shrimps	15
Fish	1
Seabirds	0.2
Seal and walrus	0.4

The number of Bq was then multiplied by the dose factor 0.95 μSv/Bq ^{239,240}Pu and the dose commitment for consumption of the product over the given period was calculated (Table 8·39).

Assuming an annual individual consumption of 5 kg bivalves, 5 kg shrimps, 25 kg fish, 5 kg seabirds, and 30 kg seal and walrus from Bylot Sound, the dose commitment to 1995 becomes 143 μSv, corresponding to the dose received in three weeks from local background radiation, i.e., an insignificant dose from a human health point of view. There appears to be no actual consumption of local marine products from the contaminated area. The doses calculated are therefore entirely hypothetical.

In 1968, the isotopic ratio ^{241}Am/^{239,240}Pu in various media was 0.05 (Aarkrog *et al*. 1984). Since then, the ratio has increased to about 0.15 but the ^{239,240}Pu activity concentrations in biota have typically decreased since 1968 by an order of magnitude. The dose factor for ^{241}Am is 0.98 μSv/Bq (i.e., nearly the same as for ^{239,240}Pu). In bivalves and shrimps, the ^{241}Am/^{239,240}Pu ratio is about twice that in the marine sediments. Hence, if the annual individual consumptions of the various products are the same as suggested above, the dose commitment from ^{241}Am in marine products from Bylot Sound consumed 1968-1995 is of the order of 10 μSv, i.e., an order of magnitude less than the dose from ^{239,240}Pu.

8.5.3.3. **The Cosmos-954 satellite re-entry in 1978**

The disintegration of the Cosmos-954 nuclear-powered satellite containing a fission reactor occurred on 24 January 1978, when the satellite re-entered the Earth's atmosphere over Canada's Northwest Territories. Early search and recovery operations showed that significant quantities of radioactive debris had spread out over a 1000 km path stretching north-east from Great Slave Lake. About 65 kg of large objects were recovered including steel plates, beryllium rods, etc., exhibiting high dose rates. This material represented only a small fraction of the total satellite mass, assumed to be several tonnes. Much material was never recovered.

Air sampling and analyses have revealed radioactive particles consisting mainly of isotopes of Zr, Nb, Ru and Ce. The absence of the volatile fission products ^{137}Cs and ^{131}I on particles showed there was some melting and reformation of core material during re-entry. Environmental measurements did not detect contamination of air, drinking water, soil or food products. Only snow samples taken from the vicinity of the debris path indicated contamination with a mixture of fission and activation products. Increased concentrations of ^{137}Cs (500-800 Bq/kg) in caribou meat were attributable to residual weapons test fallout. Intensive aerial surveys indicated that about 25% of the estimated radionuclide inventory in the satellite reactor was deposited in the form of millimeter-particles over an area of 124 000 km^2. The remaining 75% is suspected to have been volatilized and dispersed as fine dust in the upper atmosphere. This material, containing long-lived ^{90}Sr and ^{137}Cs, descended to the surface over several years. The deposited activity of ^{90}Sr in the northern

hemisphere is estimated to be about 3×10^{12} Bq compared with the total deposition of 1.1×10^{15} Bq in 1980 (UNSCEAR 1982).

The personnel involved in debris recovery obtained individual effective doses up to 5 mSv and a collective dose of about 0.1 manSv among 145 persons. Regarding public exposures, a person spending several hours a day near an unrecovered core fragment could receive an effective external dose of about 5 mSv. Handling a millimeter-size particle for several hours would give a skin dose of about 1 mSv. If ingestion of typical core particle occurred one month after reentry, the effective dose due to GIT (gastro-intestinal tract) irradiation is estimated to be between 4 and 12 mSv. Radioactive decay would cause the dose rate to decrease rapidly with time. UNSCEAR (1993) estimated the collective dose to the population of the northern hemisphere from the Cosmos-954 accident to be about 16 manSv mainly from ^{137}Cs and ^{90}Sr and, over the longer term, from ^{239}Pu.

8.5.3.4. **The Chernobyl accident in 1986**

8.5.3.4.1. **The accident and associated source term**

After nuclear tests in the atmosphere, the accident at Chernobyl in the Ukraine on April 26, 1986, was the most significant large scale source of environmental radioactive contamination. At about 01:23 hours Moscow time, two explosions in quick succession blew the roof off the Unit 4 reactor building of the Chernobyl plant, ejecting concrete, graphite and debris and leaving a gaping hole exposing the reactor core to the outside air. During a ten-day fire, smoke and gases rose to a height of over 1 km into the atmosphere, followed by fragments of uranium fuel. Transuranics and fission product radionuclides from the reactor core, plus activation products and essentially all the noble gases were released to the atmosphere during the first ten days after the accident before the releases could be contained. The heat from the fire increased the release rates of radioiodine, a substantial fraction of the volatile metallic elements, including radiocaesium, and somewhat lesser amounts of other radionuclides normally found in the fuel of a reactor that has been operating for about three years.

According to the best current information regarding the source characteristics and measurements on environmental samples, the total activity of radionuclide mixture released in this accident was about 8000 PBq (200 MCi) (as of 26 April, 1986) or 1900 PBq (50 MCi) (decay-corrected to 6 May,1986). These values do not include the noble gases xenon and krypton which were almost totally released into the atmosphere in an amount of about 7000 PBq (as of 26 April, 1986) (IAEA 1996, NEA 1995). Total releases into the environment of the most radiologically-significant volatile radionuclides ^{131}I, ^{137}Cs and ^{134}Cs have been recently estimated at about 1500 PBq, 85 PBq and 46 PBq, respectively (NEA 1995). About 8 PBq of ^{90}Sr and 0.1 PBq of alpha-emitting plutonium nuclides were also released and mostly deposited in the vicinity of Chernobyl.

8.5.3.4.2. **Radiological consequences at temperate latitudes**

The explosion of a high-power operating reactor and subsequent 10-day fire resulted in radionuclide contamination of Eurasia, especially within the territory of the former Soviet Union. Short-lived radionuclides, including ^{131}I; soil contamination with ^{134,137}Cs and, to a lesser extent with ^{90}Sr; and plutonium isotopes and, later, ^{241}Am, near to the source, were the factors affecting population exposure. More than 140 000 km^2 of the territory of Ukraine, Belarus and Russia was contaminated with ^{137}Cs at levels over 40 kBq/m^2 and more than 7000 km^2 with levels higher than 600 kBq/m^2. The surface contamination of pasture vegetation caused rapid and high contamination of local milk products with ^{131}I, ^{89,90}Sr and ^{134,137}Cs. After the decay of short and medium-lived radionuclides and the removal of the surface contamination in the autumn of 1986, the root transfer of ^{134,137}Cs from soil to vegetation and further along the food chain became the main route of internal exposure of populations. From 1987 to 1991/1993, the effective ecological half-life was 1-2 years for most local agricultural foodstuffs from contaminated regions. However, the high level of contamination of mushrooms and berries in forests of these regions declined much more slowly. Natural food products have, therefore, made a significant contribution to internal exposures of populations that consume them in the last 3-5 years (UNSCEAR 1993, Chernobyl Papers VI 1993, Strand *et al.* 1996).

Following the Chernobyl accident, countermeasures of unprecedented scale, particularly in the context of FSU agriculture, were introduced for the protection of population from long-term internal exposure (UNSCEAR 1993, Chernobyl Papers VI 1993). The average body burden in some villages of the FSU was 0.4-0.6 MBq with individuals having burdens up to 4 MBq. The average internal dose in affected areas during the first year reached 5 mSv, the total dose being 10-30 mSv with individual extremes exceeding 50 mSv. Where countermeasures were not undertaken immediately, the content of ^{134,137}Cs in the bodies of inhabitants of contaminated areas reached its maximum in the summer of 1986. A number of European countries introduced timely countermeasures. Accordingly, in these countries, early incorporation was prevented and the activity peak in body content occurred in the summer of 1987. Decreases in the ^{137}Cs content in the body reflected an effective ecological half-life of 0.3-2 years initially. Compared with the long-term external exposure, the internal dose is insignificant in black-earth regions, is similar in regions of turf-podzolic soils, and predominates in peaty soils areas in Polesye and the Arctic. The contributions of ^{90}Sr, plutonium isotopes and ^{241}Am to internal dose are insignificant.

8.5.3.4.3. **Transport and deposition in the Arctic**

Due to the intensity of the fire, which continued for ten days, and the unique meteorological conditions at the time, parts of the initial radioactive cloud reached altitudes as high as 5-10 km. Thus, although the surface winds were to the west and northwest, parts of the radioactive cloud were more widely dispersed. However, the prevailing transport of radioactive materials after the explosion was westward and then, as the wind gradually turned toward the north, northeast and south. Part of the initial radioactive releases lofted to a height of 1.5-3 km moved first over Sweden, then over Norway and southern Finland, they then continued eastward over the Archangelsk region and the southern part of the Kola Peninsula (28 April), and subsequently over Salekhard in the estuary of the Ob river (28-29 April). In late April and early May, Chernobyl fallout was essentially not observable in the far north of Russia (e.g., on the Franz Joseph Land Archipelago or Cape Zhelanie at the northern end of Novaya Zemlya). In the north of eastern Siberia, in contrast to the European Russian Arctic, radioactive deposition was relatively insignificant.

As the radioactive material from the reactor explosion remained largely in the troposphere, from which aerosols are scavenged fairly quickly, the ^{137}Cs concentration in Arctic air was reduced by nearly one half as early as June, 1986,

Table 8·40. Changes in ^{137}Cs deposition from the atmosphere in different regions of the Russian Arctic in 1986, following the Chernobyl accident.

	^{137}Cs deposition by region, Bq/m^2/month		
Month, 1986	Murmansk	Arkhangelsk	Asian North
May	553.0	187.3	27.5
June	512.3	15.1	8.21
July	3.14	6.85	1.98
August	1.64	4.4	0.91
September	1.27	3.74	0.63
October	0.70	3.11	0.65
November	0.61	2.58	0.50
December	0.46	0.68	0.13

and continued to decrease in the following months. During subsequent years, the ^{137}Cs concentration in the atmosphere in the northern European part of Russia decreased exponentially but 1.5-2 times more slowly than in the mid-latitudes where the effective ecological half-life was 17 months. Table 8·40 shows changes with time in the monthly atmospheric deposition of ^{137}Cs in various regions of the Russian Arctic during 1986. It can be seen that rather high levels of ^{137}Cs in deposition were exhibited during May-June on the Kola Peninsula (the Murmansk region) and the lowest values occurred in the Asian part of the Arctic. High levels of ^{137}Cs deposition in June in the western part of the Russian Arctic are explained by wind resuspension and migration of radioactive dust particles.

Large-scale environmental monitoring programs, started soon after the accident both in the former Soviet Union and other European countries, enabled the production of maps of aerial contamination of Europe with long-lived ^{137}Cs (EU/CIS JSP-6 1996). Figure 8·5 presents such a map covering the northern part of Europe. It can be seen that closest to the Arctic, the significantly contaminated area extends in an east-west direction across the Leningrad region of Russia, southern Finland and Scandinavia. There are no large areas above the Arctic Circle contaminated by more than 10 kBq/m^2. Contamination of northern Fennoscandia is mainly < 2 kBq/m^2. In contrast, most of the European part of the Russian Arctic is more contaminated, but still at a level of less than 10 kBq/m^2, reflecting some additional radionuclide deposition of Chernobyl origin. This observation is based on measurements of Chernobyl-derived ^{134}Cs in deposition. More detailed mapping by Roshydromet has identified some areas with deposition of between 4 and 18 kBq/m^2 on the coast of the White and Pechora Seas (Izrael *et al.* 1990).

Marine transport to the Arctic seas

In addition to the direct atmospheric fallout from Chernobyl, the Arctic seas may also be contaminated by marine transport from the North Sea and the Baltic Sea, the catchments of both of which received considerably more radionuclides from Chernobyl (through direct fallout and runoff) than the Arctic.

Studies of radionuclides in the Baltic Sea have been carried out under the aegis of the Monitoring of Radioactivity in the Sea (MORS) group under the Helsinki Commission. As a result of the atmospheric fallout following the accident, contamination of the Baltic Sea was highly heterogeneous. Nies and Nielsen (1996) have used the extensive MORS data to calculate the inventory of ^{137}Cs in Baltic seawater for the years 1985-1990. These Baltic Sea inventories are shown in Table 8·41.

Table 8·41. Inventory of ^{137}Cs in Baltic seawater (TBq).

Year	Inventory, TBq
1985	325
1986	4260-5000
1987	2700
1988	1790
1989	2320
1990	2060

The concentrations in water declined by a half within the first year following the Chernobyl accident due to sedimentation of ^{137}Cs. Since 1987, the inventory of ^{137}Cs has decayed with an effective environmental half-life in the range 10-15 years compared with a mean residence time of Baltic Sea water of 20-45 years. From these data, it can be estimated that the present (1996) input of ^{137}Cs to the Arctic from the Baltic Sea is of the order of 50 TBq/y and that the total transport of ^{137}Cs derived from the Chernobyl accident to the Arctic has been in the range 0.5-1 PBq.

The ratio of ^{134}Cs/^{137}Cs in Kara Sea water, determined in 1992, was relatively constant (in the range 0.019-0.034) (Strand *et al.* 1993) and systematic geographical variations could not be distinguished. The primary known sources of ^{134}Cs are the discharges from the Sellafield reprocessing plant and Chernobyl fallout. Assuming that the ratio of ^{134}Cs /^{137}Cs in Sellafield discharges has been consistently less than 0.1 since the mid-1970s, the ^{134}Cs /^{137}Cs ratio in the Kara Sea, if Sellafield were the sole source, would be insignificant because of the 2-year physical half-life of ^{134}Cs. It is, therefore, unlikely that the ^{134}Cs observed in the Kara Sea in 1992 originates from Sellafield. Accordingly, assuming that Chernobyl is the only source of ^{134}Cs, approximately 30% of the ^{137}Cs in the Kara Sea is derived from the Chernobyl accident (Strand *et al.* 1993).

8.5.3.4.4. **Food chain and human contamination**

As the ecological food chain lichen → reindeer → humans is the most important in the exposure of members of the Arctic population to environmental contamination with radionuclides, this is considered in relation to the dynamics of ^{137}Cs following the Chernobyl accident. Measurements of the distribution of ^{137}Cs in the environment following the Chernobyl accident are presented in section 8.3.3.

Lichen

A clear increase in ^{137}Cs concentrations in Arctic lichen was observed following the Chernobyl accident. In Arctic Finland, the peak concentration of ^{137}Cs in dried lichens in 1986-1987 was up to 1300 Bq/kg above previous levels. The concentrations decrease with an effective ecological half-life of about 3-4 years. This is faster than that occurring after global fallout because the Chernobyl accident was a single contamination event in contrast to the sustained period of nuclear weapons testing in the 1950s and 1960s. Less pronounced was the ^{137}Cs concentration increase in Arctic lichens collected in the Murmansk region and the Nenets area of northern Russia and in Greenland. This geographical difference can be explained by the heterogeneity of ground deposition, location or sampling sites, and/or variations in snow cover conditions during May-June 1986.

Reindeer

There was an increase in radiocaesium in reindeer meat produced in Finland, Norway, Sweden and Western Russia in 1986. Because the ^{137}Cs content in live reindeer in winter follows the concentration in lichen, it decreased with an effective ecological half-life of 3-4 years following the peak in 1986-1987.

Human body

Long-term radioactive contamination of the Arctic from Chernobyl, mainly with caesium radioisotopes, ^{134}Cs and

^{137}Cs, in 1986 and subsequent years caused significant increases in the wholebody content of these radionuclides in Arctic indigenous people consuming local natural and semi-natural foods (reindeer meat, freshwater fish, mushrooms and berries) that concentrate certain radionuclides. This increase occurred in all European Arctic countries except Iceland and Greenland which were not so significantly contaminated by Chernobyl fallout, see section 8.3.3. A typical wholebody value for ^{137}Cs in Finnish Saami living in the Inari region in the early 1980s was 4 kBq. This had risen to about 9 kBq in 1986-1988 and decreased to the original value during the subsequent seven years (Rahola *et al.* 1993). In both northern and southern Saami areas of Norway, Saami people were contaminated by Chernobyl fallout. Deposition levels varied considerably. The pre-accident body burden of 3 kBq reached 40 kBq in areas most affected by Chernobyl fallout and about 7 kBq in areas which were least affected. For both groups, the consumption of reindeer meat dominated the intake of radiocaesium. Figure 8·63 presents dose estimates based on wholebody measurements for the two groups of Norwegian reindeer-breeding Saami,

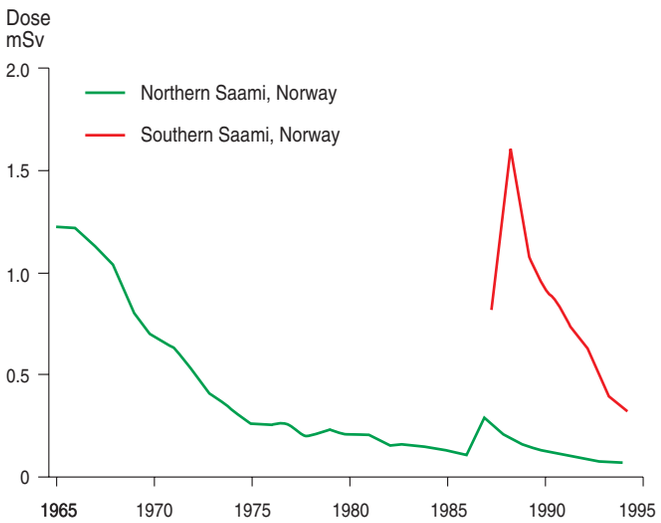


Figure 8·63. Dose estimates based on wholebody measurements (see Figure 8·36) for two groups (northern Saami and southern Saami) of Norwegian reindeer-breeding Saami.

northern Saami and southern Saami. After 1986, there was an increase in both groups but of smaller magnitude in the northern population. The increase in the southern Saami would have been considerably larger but for the implementation of countermeasures. Reindeer-breeding Saami people living in the north of Sweden accumulated an average of about 40 kBq of ^{137}Cs in 1987-88 compared to 5 kBq previously (Johanson and Bergstrom 1993). In Russia, the most detailed information is available for indigenous inhabitants of the Murmansk region (Kola Peninsula). Before the Chernobyl accident, the average ^{137}Cs wholebody content of adults was about 20-30 kBq, the highest values within the Eurasian Arctic (Ramzaev *et al.* 1993). In June, 1986, the average wholebody content remained at a level of 26 kBq, but by July, 1991 it had increased to 33 kBq.

The peak in the ^{137}Cs wholebody content of indigenous people living in different areas and countries depends on the extent of deposition, meteorological conditions, the duration of snow cover in May-June, 1986, and individual food habits. Contamination with both ^{137}Cs and ^{134}Cs of Chernobyl origin occurred. The mean ratio of ^{134}Cs/^{137}Cs, which was 0.54 at the time of the accident, has decreased over time due to the faster decay of ^{134}Cs (physical half-life 2.05 y).

8.5.3.4.5. **Countermeasures**

A wide range of countermeasures were radiologically justified and introduced, especially in Norway and Sweden, following the Chernobyl accident, and some of these are still being applied. Such countermeasures included: ploughing and fertilizing of natural pasture to reduce uptake by plants from soil; feeding of animals with fodder containing lower levels of contamination; use of caesium-binders (saltlick, mixed in concentrate or boli) for freely grazing animals; changing the slaughtertime of wild or semi-wild animals; and dietary advice to consumers. Countermeasures considerably reduced the doses to the population in the affected areas, especially for reindeer-breeding Saami. The dose reduction achieved for this latter group was up to 90% (Strand *et al.* 1990). The countermeasures also had positive effects on social well-being (Strand 1994, Brynhildsen *et al.* 1996, Strand *et al.* 1990).

8.5.3.4.6. **Human dose estimation**

Internal dose estimation was made on the basis of wholebody measurements conducted prior to, and after, the Chernobyl accident. The ^{137}Cs wholebody content was integrated over a period of 5-8 years after 1986 and combined with the appropriate dose conversion factors. It was assumed that 50-70% of the effective internal dose was delivered during the observational period. A further 30-50% is expected to be delivered in the future. For the groups described in the previous paragraph, the committed effective internal dose from caesium radioisotopes of Chernobyl origin ranges from 0.5-10 mSv, depending mainly on ^{137}Cs soil contamination density and food habits. Given an initial deposition of 1 kBq/m^2 of ^{137}Cs (and 0.5 kBq/m^2 of ^{134}Cs), the dose commitment will be in a narrower range of about 0.5-1.5 mSv, which is an order of magnitude more than in temperate latitudes. For comparison, at the same level of initial deposition, the external dose commitment for Nordic indigenous people is about 0.1 mSv. Thus, for people consuming Arctic natural and semi-natural foods, internal exposure from Chernobyl contamination of the environment contributes 80-90% of the total (i.e., external + internal) dose. The collective dose for about 2000 000 persons in the average population and about 100 000 persons in the indigenous population living in the European Arctic, contaminated after the Chernobyl accident at an average level of about 1 kBq/m^2 of ^{137}Cs in Fennoscandia and 2 kBq/m^2 in western Russia, has been preliminarily estimated to be about 500 manSv.

8.5.3.5. **Accidents involving nuclear-powered vessels**

There have been several cases of radioactive contamination of the environment during the period of operation of nuclear-powered ships. In 1961, a submarine with a damaged reactor returned to its base on the Kola Peninsula and local contamination occurred. In 1965, a local release of radioactive materials was reported during an accident with a submarine reactor in the Severodvinsk shipyard. The largest release of liquid radioactive waste (74 TBq) occurred in 1989 from a Northern Fleet submarine in the Ara inlet. This accident led to radioactive contamination of an area of about 1.0 km^2. Two other nuclear submarine accidents occurred in the Norwegian Sea in 1989; one involving failure of the cooling circuit in a Soviet (NATO Echo-class) submarine and the other the loss of the (NATO Mike-class) submarine *Komsomolets*.

During accidents involving submarine nuclear reactors, personnel have been affected by gamma-radiation which can

result in high doses causing development of acute radiation sickness to its highest degree. In addition, beta-radiation from radioactive gases entering the hull of the submarine during major accidents can cause acute radiation damage to skin. Collective doses of gamma-radiation obtained by personnel of nuclear submarines during several previous accidents have been between 17 and 74 manSv, and the average individual doses between 0.2 and 0.6 Sv. The average individual dose to skin was between 2 and 6 Gy and the average individual dose to thyroid, associated with the inhalation of radioiodine, was between 2 and 10 Gy.

In 1961, a submarine, which had suffered an accident, returned to its base. Samples of seawater were subsequently taken at the distances of 5, 50, 100, and 300 m from the submarine hull at the surface and at depths of 5, 10, and 20 m. The highest specific activity in seawater, of 37 Bq/L, was found at a distance of 5 m from the hull at a depth of 5 m. Radioactivity in samples of plants, fish, and sediment taken during ventilation of submarine compartments had background levels. Three months later, the measurements of radionuclides in seawater in the vicinity of the submarine mooring were repeated. The activity concentrations in samples of seawater, taken predominantly in the region of the reactor power compartments, were between 0.1 and 27 Bq/L, which corresponds to background.

In 1968, radioactive contamination of a naval base occurred during decontamination work on a submarine. Increased alpha- and beta-activity was detected in seawater near the mooring location. However, monitoring of sediments, algae and fish at all sites gave values corresponding to background levels.

In 1979, during work on remediating the consequences of another submarine accident, the gamma-radiation dose rate in the sanitary zone of the naval base reached 1.3 µGy/h. The sources of contamination were the submarine which had suffered the accident, the stored radioactive waste removed from it, and releases to the atmosphere during the ventilation of contaminated compartments. The average concentrations of fission products in seawater in the bay adjacent to the base during the period of remediation of consequences varied within the range of background associated with global fallout. Short-term contamination of seawater was observed at the mooring location of the submarine, and was caused by products of decontamination of the submarine hull and by the release of low-level waste from the coastal reservoir to the bay. No significant increases of radioactivity of algae in the region of the submarine mooring, bottom sediments or benthic organisms (starfish) in the bay were detected. Any increased levels could not be attributed to the submarine because similar values were measured prior to the accident. During the period of work on the submarine, the concentrations of short-lived radionuclides in atmospheric air at the location of its mooring did not increase significantly. No contamination of soil and vegetation at the base location was observed during this period.

A Soviet nuclear submarine (NATO Echo-class) with nuclear weapons onboard had problems in the Norwegian Sea in June, 1989. The prime cooling circuit failed and a supply ship had to deliver cooling water (see Figure 8·64). The submarine subsequently sailed back to its base on the Kola Peninsula. During its time off the Norwegian coast there were measurable amounts of radioiodine in the surrounding water and in air samples in the north of Norway. Radioiodine in milk from animals grazing freely in the north of Norway ranged from 0-10 Bq/L. Based on air measurements in Norway and information from the former Soviet Union, the total releases of ^{131}I were of the order of 0.1-10 TBq. The inhaled activity for the people onboard neighboring Norwegian surveillance and rescue vessels were in the range 0-43 000 Bq ^{131}I. The doses to thyroid in the most exposed personnel were 1.6 mGy. Doses to the Norwegian population, assessed on the basis of analysis of air and milk samples, were shown to be insignificant (NRPA 1989).

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Figure 8·64. Burning nuclear submarine.

8.5.3.5.1. Sunken *Komsomolets* submarine

8.5.3.5.1.1. Accident and source term

On 7 April, 1989, the Soviet nuclear submarine *Komsomolets* (K278, NATO Mike-class) caught fire and sank 180 km southwest of Bear Island (Bjørnøya) in the Norwegian Sea. Of the 69 crew members on board, 42 were killed in the accident. The submarine was designed to operate at depths of up to about 1000 m, and its double hull was made of a titanium alloy. At present, the wrecked submarine rests at a depth of ca. 1650 m. The single reactor was shut down in an orderly manner prior to sinking. To estimate causes and consequences of the accident and to develop countermeasures, a number of Russian expeditions to the *Komsomolets* have taken place in recent years.

The sunken submarine contains one nuclear reactor with an inventory of long-lived radionuclides comprising 2.8×10^{15} Bq of ^{90}Sr and 3.1×10^{15} Bq of ^{137}Cs along with other fission and neutron activation products. The change in the reactor nuclide inventory with time is depicted in Figure 8·65 derived from Gladkov *et al.* (1994) and reproduced from the NATO-CCMS Report (CCMS/CDSM/NATO 1995). It can be seen that a decade after the accident long-lived ^{137}Cs and ^{90}Sr will dominate among the fission products. Most activation products are likely to have essentially decayed before major releases through corrosion are liable

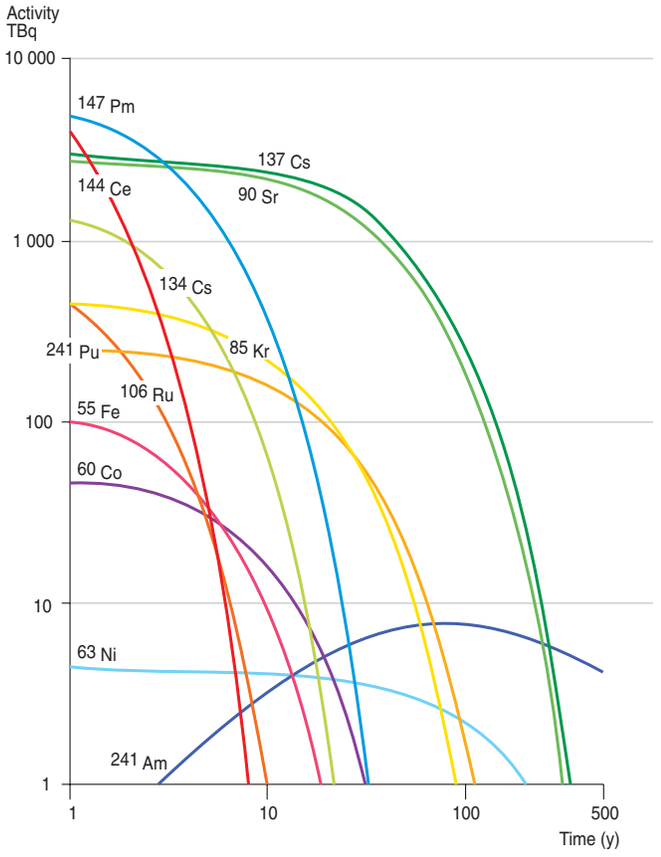

Figure 8·65. The changing inventory of major radionuclides in the *Komsomolets* reactor with time (after Gladkov *et al.* 1994 and CCMS/CDSM/NATO 1995).

to occur. Two nuclear torpedoes with mixed uranium/plutonium warheads, situated in the forepart of the hull contain about 1.6×10^{13} Bq of weapons-grade plutonium.

Minor releases of radionuclides from the reactor compartment have already been detected in the close vicinity of the submarine wreck during Russian expeditions. These surveys indicate radionuclide releases through a reactor ventilation tube, the ^{137}Cs activity concentration in the water inside the tube being of the order 1 MBq/m^3. The annual release of ^{137}Cs from the submarine was estimated to be no more than 0.5 TBq. The likelihood of large-scale releases of radionuclides from the *Komsomolets* submarine in the near future is small. As the containment barriers in the submarine are breached by corrosion, further gradual releases may occur and these will increasingly comprise long-lived fission products from the reactor and uranium and plutonium from the nuclear-tipped weapons. While uranium is relatively soluble and will be mobilized as the structural integrity of the torpedo and warhead casings is breached, the environmental contribution will be essentially insignificant in the context of the natural uranium content of the surrounding environment. Plutonium has limited solubility and a high affinity for particles. Accordingly, most of the plutonium released from the warheads is likely to be retained in sediments within the immediate vicinity of the wreck.

8.5.3.5.1.2. Radiological assessments of the *Komsomolets* accident

There have been two major assessments of the radiological threat posed by the *Komsomolets*. The first of these was carried out by Norwegian experts under the auspices of the NATO sub-Committee on Challenges to Modern Society (CCMS/CDSM/NATO 1995) and the second by experts from the Russian Navy (Lisovsky *et al.* 1996). In addition, a study of the release and transport of radionuclides from the wreck has been carried out by scientists at the Norwegian Institute of Marine Research, Bergen (Blindheim 1994).

The NATO study was intentionally based on a relatively simple model that estimates the corrosion rate of the reactor and torpedoes, considers transport of water-soluble radionuclides by ocean currents but takes no account of radionuclide partitioning between water and particles. The study draws conclusions primarily from the results of the modeled dispersion of soluble ^{137}Cs, the predominant long-lived fission product. Because the modeling is based on an assumption that all radionuclides are completely water soluble, it overestimates the dispersion of particle-reactive nuclides such as ^{60}Co, ^{241}Am and plutonium isotopes.

Barriers to the release of radionuclides within the reactor fuel rods comprise cladding of 5 mm thick stainless steel, the reactor vessel of 150 mm thick carbon steel and the submarine hull of 100 mm titanium alloy. These are expected to prevent corrosion of the reactor fuel for about two thousand years. By that time, only plutonium isotopes and americium will be present in the reactor in significant activities. In the intervening period, the only pathway of radionuclide release from the reactor will be water exchange through the reactor compartment ventilation tube. The rate of ^{137}Cs release can, therefore, be assumed to be comparable with the present rate of about 0.5 TBq/y with appropriate correction for radioactive decay.

Plutonium in the torpedoes is not protected from sea water to the same degree and is expected to be open to corrosion much more quickly than the reactor fuel. Plutonium is highly particle-reactive and insoluble. Accordingly, any plutonium released is likely to be retained in marine sediments close to the point of release as occurred following the Thule nuclear weapons accident.

The transport of radionuclides released from the *Komsomolets* has been modeled by Blindheim (1994) and NATO (CCMS/CDSM/NATO 1995). Both studies have, however, used conservative models that will be more-or-less applicable for nuclides having low particle reactivity (low kds) but be overly conservative (i.e., pessimistic) for reactive nuclides such as ^{60}Co, plutonium and americium. The models predict highest concentrations of conservative radionuclides northwest of Spitsbergen but the contamination is distributed over a large area both in the Arctic and Atlantic Oceans. The Blindheim (1994) study also considers the importance of a 'thermal plume' arising from residual heat in the *Komsomolets* reactor but concludes that the energy is insufficient to create a plume rising more than 500 m from the bottom of the Norwegian Sea.

Assuming that the releases from the submarine occur as suggested above and fish permanently inhabit contaminated water, the contribution of *Komsomolets* releases to radionuclide contamination of the Arctic waters and to the most severely affected fish can be estimated. The NATO study estimate that the increase in ^{137}Cs in seawater was about 0.02 Bq/m^3 in 1991, 0.01 Bq/m^3 in 1995 and is expected to decrease to about 0.001 Bq/m^3 in 2089. These levels should be compared with the concentrations of ^{137}Cs attributable to other sources in surface waters of the open Barents and Kara Seas of between 1 and 10 Bq/m^3, respectively.

The radionuclide concentration in fish is assumed to be proportional to the concentration in surrounding water. *Komsomolets* releases are predicted to contribute concentrations of about 0.005 Bq/kg in 1991, 0.002 Bq/kg in 1995 and 0.00005 Bq/kg in 2089 in the most significantly affected fish. The dominating radionuclides in 1995 are ^{147}Pm, ^{137}Cs

and ^{55}Fe. For comparison, typical Arctic fish generally have a level of about 1 Bq/kg of ^{137}Cs.

This conservative modeling indicates that radionuclide concentrations in seawater and fish caused by past, present and future releases from *Komsomolets* are at least two to three orders of magnitude lower than current concentrations of ^{137}Cs in the same media. Human intake of released radionuclides through seafood consumption should represent an even lower proportion of current exposures because seafood is generally of minor importance as a source of internal dose to humans. Thus, the *Komsomolets* submarine constitutes an insignificant source of existing and projected marine contamination and radiation exposure. The NATO study (CCMS/CDSM/NATO 1995) concludes that the results 'clearly indicate that the sunken submarine represents no significant hazard to man, today or in the future'.

For completeness, a brief presentation of the Russian Navy assessment (Lisovsky *et al*. 1996) of the risks posed by releases from the *Komsomolets* submarine reactor and weapons is given below.

A generalized computer model was constructed for estimating the radioecological consequences of the release of radionuclides to the marine environment. The model embodies the main processes of radionuclide migration and accumulation in environmental compartments and the radiological consequences for humans and marine biota. For estimating the radionuclide releases from the active zone of the reactor, the corrosion rate was assumed to be linearly dependent on temperature. The corrosion rate was chosen on the basis of engineering data on the corrosion resistance of the various barrier materials. The model for mass and heat transfer was based on ordinary differential equations.

Of the number of radionuclides that are formed in the active zone during operation of a reactor, three long-lived radionuclides ^{137}Cs, ^{90}Sr and ^{239}Pu are of primary concern. Calculations were performed only for caesium. The content of plutonium in the active zone of the reactor of the given type is extremely small. For the calculations, it was assumed that the reactor compartment and the active zone of the reactor were damaged at the moment of the vessel's impact on the sea bottom. It was further assumed that the open area of the active zone is 5 cm^2, the area of holes in the guard of the reactor compartment is 2 m^2 and the total area of defects in fuel cladding at the time of the accident was 80 cm^2.

The predicted maximum release rate of ^{137}Cs to the environment is calculated to be 0.05 TBq/y. This is within an order of magnitude of the release rate estimated on the basis of measurements during sea expeditions to the *Komsomolets* wreck site during the period 1992-1994 which is 0.004-0.4 TBq/y. These are relatively small release rates and would not be expected to entail any significant radiological consequences.

About 1.5×10^{13} Bq of weapon ^{239}Pu in the torpedoes of the sunken submarine is assumed to come into direct contact with seawater, resulting in corrosion and plutonium release to the environment. Two release rate scenarios were used in the modeling. The first is based on a release rate that is governed by an exponential law with a half-period of 24 hours; the second is based on an exponential release of half-time of 10 000 hours (i.e., ca. 1 year). The latter variant corresponds to a case of a release rate governed by the corrosion rate alone and yields a constant rate of release.

The results of generalized calculations of plutonium concentration fields are presented in Table 8·42 for the case of the longer-term plutonium release. These indicate that the maximum level of contamination does not exceed 0.27 Bq/L for water and 2.7 Bq/m^2 for sediments. The zone of contamination (i.e., the zone in which concentrations exceed the background contamination by an order of magnitude) has a form close to a symmetrical ellipse with its axis elongated in a northwesterly direction. Under slow release conditions, the contamination zone dimensions do not exceed 200×200 km. The water contamination zone under slow release conditions can persist for about 4-5 years. The bottom contamination zone continuously increases but stabilizes in response to the decline in the level and extent of water contamination. In the slow release scenario (half-period of the release = 10 000 hours), water contamination during the initial period reaches 0.027 Bq/L and a zone with such a ^{239}Pu activity concentration will persist for up to 5000 hours.

Table 8·42. Plutonium-239 distribution in the aqueous marine environment (half-period of release is assumed to be 10 000 hours).

	Time, hours					
	100	500	1000	5000	10000	50000
Activity concentration, Bq/L	Water, km^2					
0.27	2	0	0	0	0	0
0.027	51	48	80	9	0	0
0.0027	460	1200	3000	1400	200	0
0.00027	5400	30000	40000	76000	40000	0
Activity concentration, Bq/m^2	Sediments, km^2					
2.7	0	0	5	5	5	5
0.27	0	80	100	1500	5800	9900

The international expedition conducted in 1994 suggested that some kinds of zoocenosis in the region of the *Komsomolets* wreck could result in the transfer of radionuclides from deep ocean layers to surface layers and serve as food for fishes feeding on plankton. In this respect, Calanus hyperboreus and Themisto olyssorum were regarded as of primary importance, although whether these organisms feed at depth has been questioned. In the absence of data on the dynamics of plutonium accumulation by these organisms, it was conservatively assumed that body content reaches equilibrium in 100-150 hours. The most unfavourable situation will be one in which the release of plutonium from the sunken submarine occurs a short time before the seasonal migration of this species of zooplankton to the upper ocean layers. In such a case, the incorporation of plutonium into fish and consequent exposures to human seafood consumers will be a maximum. Assuming the accumulation factor to be 2600, for the scenario of slow plutonium release, the maximum possible concentration in plankton rising to the upper layers will be 70-700 Bq/kg and the area of contamination will not exceed 80 km^2. The zooplankton species migrating from the near-bottom layers comprise 2-17% of the ration of marketable fish (i.e., the specific content of ^{239}Pu in the biomass that serves as food for marketable fish will range from 1 to 120 Bq/kg). After one to three months, zooplankton dispersion will result in a decreasing plutonium specific activity.

According to conservative modeling of the radionuclide accumulation by hydrobionts, fish having a maximum plutonium content of 0.1-6 Bq/kg in edible parts may appear for some months in an area of about 80 km^2. When individuals leave the contaminated zone as a result of natural migration, the plutonium content in fish will decrease according to a linear or exponential law. Thus, within 25-30 days of the cessation of contaminated plankton consumption, the contamination level in edible tissues will decline to less than 0.1 Bq/kg.

On the basis of the standard maximum consumption of sea products, the hypothetical dose to members of the critical group can be no more than 0.03 mSv/y based on conser-

vative assumptions. Realistically, it should actually be at least an order of magnitude less, namely of the order of 1 μSv/y. The estimated dose rate in zooplankton permanently inhabiting the most contaminated sea area may reach about 1 mGy/h for some months. Fish consuming this zooplankton would receive about 1 μGy/h. This exposure level cannot cause any significant biological effects in marine biota populations. For migrating hydrobionts, the levels of exposure will be one or two orders of magnitude lower.

Thus, based on the results of the studies carried out by NATO and Russian Navy experts, it can be concluded that the threats posed by radionuclides in the wreckage of the *Komsomolets* submarine are minor.

8.5.4. Summary

The major contribution (≈15 000 manSv) to the collective dose to Arctic populations results from fallout from nuclear weapons testing. The range of individual dose commitments to members of the Arctic population resulting from fallout is 1-150 mSv. The next most important contribution (≈500 manSv) to collective dose within the Arctic derives from the Chernobyl accident. Individual dose commitments associated with releases from Chernobyl were in the range 1-50 mSv. Individual annual doses to the most exposed residents of the Arctic from Chernobyl releases were, however, about 10-20 mSv/y. Countermeasures introduced by several countries following the Chernobyl accident and, in some cases, being maintained to the present day, resulted in individual doses and dose commitments being much smaller than they would have been if no intervention measures had been introduced. The effectiveness of these measures can be judged by the 90% reductions in dose to native people that were achieved in Fennoscandia. The next most important collective dose contribution (≈50 manSv) results from releases from the Sellafield fuel reprocessing plant but the contribution of this source to individual dose has been relatively small (i.e., in the range 0-0.05 mSv) (UNSCEAR 1993).

Smaller-scale releases from accidents in military operations, such those in northern Russia, the plutonium spill at Thule and the loss of the *Komsomolets* submarine in the Norwegian Sea, have resulted in no significant increases in radiation exposures within Arctic populations. However, a limited number of military personnel have been exposed to significant doses in connection with accidents on nuclear vessels. It has been estimated that the collective dose to personnel operating Russian nuclear vessels was in the range 17-74 manSv with the highest individual doses in the range 0.2-0.6 Sv.

For some previous releases to the environment, it has been difficult to carry out an assessment of the collective or individual doses specifically to Arctic populations. Such cases include releases from the Mayak plant and releases from Sellafield for which an assessment in relation to the Arctic area was based on a global assessment.

8.6. Source-related assessments of potential releases

This section discusses potential releases that may occur in the future: as a result of accidents within the nuclear power and weapons industries; from contained sources of radionuclides within the marine and terrestrial environments; as a result of failures of containment structures for radionuclides stored in, or previously released to, restricted areas of the environment; and from accidents involving nuclear weapons.

8.6.1. Nuclear power plant reactor accidents

Prevention of nuclear reactor accidents has been the main objective of nuclear safety since the beginning of the nuclear era. International and, in most countries, national nuclear safety efforts were improved following the Chernobyl reactor accident. The first step in national nuclear safety administration is the development of a regulatory infrastructure, laws and regulations to specify the criteria against which safety is judged.

To achieve and maintain a high level of nuclear safety worldwide through the enhancement of national measures, an International Convention on Nuclear Safety was signed under IAEA auspices in 1994 (IAEA 1994). National regulations understandably vary in detail from country to country. The Nuclear Safety Convention stipulates the minimum requirements for the operation of a nuclear installation; for example, it prohibits the operation of a nuclear installation without a license and it requires comprehensive and systematic safety assessments to be carried out prior to licensing and throughout the lifetime of the installation.

The Convention also requires that, when necessary, a Contracting Party shall ensure that all reasonably practicable improvements are made as a matter of urgency to upgrade the safety of an existing nuclear installation. If such upgrading cannot be achieved, plans should be implemented to shut down the installation as soon as practically feasible.

The primary risk posed by a nuclear reactor relates to the large amount of radioactive material, primarily fission products, that are generated during its operation. The release of only a small fraction of these to the environment could cause severe harm to humans and to the environment. Therefore, one of the main aims of nuclear safety is to prevent the release of radioactive fission products into the environment.

To prevent such releases, a number of barriers are placed between the primary risk source (i.e., the fuel) and the environment. In addition, the nuclear chain-reaction in the reactor core should be self-controlling or inherently stable, so that small perturbations in operating conditions always cause the reactor to return to normal conditions by itself. To cope with abnormal conditions, the reactor should possess effective and expedient shut-down capabilities.

Safety criteria

Comparisons between some Western and Eastern safety criteria have been made – for example, by the Nordic Nuclear Safety Research (NKS) program (NKS 1994). The following observations have resulted from these studies:

- In the West, safety design has often been demonstrated through tests and experiments in pilot plants. These demonstrations have shown the functioning of the system in question and helped to verify computer codes developed for analysing the safety of nuclear power plants.
- In contrast, in the former Soviet Union, safety design was often based on calculations rather than experiments. However, the systems were often designed conservatively, such that pipe dimensions, the number of pumps, the size of vessels, etc., were larger than necessary. This compensated for some of the uncertainties in the calculations and the lack of experiments and verification of codes.

Differences between Eastern and Western practices regarding some basic safety criteria are described in more detail below.

Western safety concepts give priority to measures for accident mitigation and accident management as well as automatic actions of safety systems. To relieve operators and to

reduce the response time of protection systems, a progressive concept of protection by automatic control is applied.

The barriers present between the primary risk source and the environment differ in the West and the East as follows:

West	*East*
Fuel matrix	Fuel matrix
Fuel cladding	Fuel cladding
Pressure boundary of primary coolant system including reactor vessel	Pressure boundary of primary coolant system including reactor vessel
Reactor containment	Confinement
Filter	—

According to the relevant safety criteria, protective measures are realized in the West at four, and in East at three, different safety levels:

West	*East*
Normal operation	Normal operation
Transient conditions	Upset conditions
Design basis accident	Design basis accident
Incidents beyond design basis accidents	—

The principal aim of all western safety considerations is to ensure that the radioactive materials present in a nuclear power plant are confined at all times. In other words, a nuclear power plant must be designed and operated in such a way that, at all times, during specified normal and upset operation and during the so-called design basis accidents, the following design goals must be fulfilled:

- The reactor can be safely shut down and be kept shut down;
- The residual heat can be removed;
- The radiation exposure of personnel and radioactive releases to the environment must be kept as low as possible.

To achieve these design goals, the safety precaution principles were set up with a multiple level safety concept as follows:

- Assurance of normal operation with least possible occurrence of abnormal operating conditions;
- Control of abnormal operating conditions that might occur through the application of engineered safety features; and
- Assurance that design basis accidents stay within given limits with assurance of dose minimization by means of engineered safety features.

Furthermore, the so-called 'single failure criteria' must be fulfilled; that is, the safety systems must comply with the design criteria even under the assumption of a single component failure in one of the safety systems.

When analysing emergency conditions, the following criteria are applied, with the differences between Western and Eastern practice indicated in parentheses:

- With the reactor at rated power, (a maximum diameter pipe break (West); a break of a pipe with a diameter of 500 mm (East)) with a two-way free outflow of coolant (a so-called guillotine break) is postulated to be the design basis accident. (Note: for some of the oldest Russian reactors, the pipe break diameter was limited to 32 mm. The pipes then had flow-reducing orifices).
- Regarding fuel, the following design limits are applied to normal operation:

  The number of failed fuel rods with gas leakage must be less than 1.0% of the total number of fuel rods (West and East).

  The number of failed fuel rods resulting in direct contact between fuel and coolant must be less than 0.1% of the total number of fuel rods (West and East).
- During accident conditions the following limits are applied:

  The maximum cladding temperature must be less than 1200°C (West and East).

  Local depth of oxidation of fuel cladding must be less than 17% (West) or 18% (East) of the original cladding thickness.

The NKS comparison among the respective western and eastern safety criteria indicates that little difference exists between them. Overall, the largest difference is the lack of full containment in many of the reactors of the former Soviet Union.

The values stipulated in Finland are presented as examples of the dose and activity levels related to the technical requirements at various safety levels. In a design basis accident, the limit for the dose to an individual in the population arising from external radiation in any period of one year and associated radioactive materials intake is 5 mSv. In the case of a beyond-design basis accident, the limit for the release of radioactive materials arising from such an accident is that which causes neither acute harmful health effects to the population in the vicinity of the NPP nor any long-term restrictions on the use of extensive areas of land and water. For satisfying the requirement relating to long-term effects, the limit for atmospheric releases of ^{137}Cs is 100 TBq. The combined fallout comprising nuclides other than caesium-isotopes shall not cause, in the long-term (starting three months from the accident), a hazard (dose equivalent) greater than that arising from a radiocaesium release corresponding to the specified limit. The probability that, as a result of a severe accident, the above mentioned requirement is not met, shall be 'extremely small' (STUK 1992).

After the Chernobyl accident, most countries introduced enhanced safety measures, such as the introduction of filters to reduce accidental releases of radionuclides in particulate/aerosol form. In Sweden, requirements for such additional safety measures were formulated by limiting releases, even in very severe accidents, to noble gases and to less than 0.1% of the core inventory of long-lived radionuclides such as ^{137}Cs (Sweden 1982). The implementation of such measures took place between 1985 and 1989.

8.6.1.2. **Probabilistic safety assessment (PSA)**

Previous safety analyses that needed to be performed before licensing and throughout the life of a NPP were mostly deterministic and did not address the probabilities of events. Compliance with safety requirements was checked by deterministic analyses that used pessimistic assumptions to ensure that the results of the assessments were conservative (i.e., 'on the safe side'). However, the lack of events of safety significance in the operating experience of NPPs does not preclude the existence of underlying safety deficiencies. Safety experts considered that a gap existed between the bulk of operating experience and events of safety significance that could only be bridged by theoretical analyses. PSA is a systematic approach to performing such analyses (IAEA 1992). Thus, currently, both deterministic and probabilistic safety analyses are used to complement each other in nuclear safety.

In practice, PSA aims at:

- Identifying and delineating the combinations of events that may lead to a severe accident.
- Assessing the expected probability of occurrence for each combination.
- Evaluating the consequences.

There are three levels of PSA:

- *Level 1* provides a review of plant design and operation, focusing on sequences that could lead to core damage.
- *Level 2* addresses, in addition to the analyses on Level 1, the phenomenon of a core damage accident, the response of the containment to the expected loads, and the transport of radioactive material from the damaged core to the environment.
- *Level 3* addresses, in addition to the requirements of levels 1 and 2, the dispersion of radionuclides in the surrounding environment and potential environmental and health effects.

Levels 1 and 2 PSAs are fairly well-established tools for nuclear safety, although not carried out for all NPPs relevant to this assessment. PSA Level 1 studies are available for all Finnish and Swedish reactors. Level 2 studies for these reactors are underway and have been partially reviewed by safety authorities. Level 3 PSAs were not available to the assessment group for any of the NPPs relevant to effects in the Arctic. Typical official or unofficial criteria used in some of the western Arctic countries are 1.0×10^{-4} to 1.0×10^{-5} (core damage probability/year) for Level 1 and 1.0×10^{-6} to 1.0×10^{-7} (for large off-site releases/year) for Level 2 (NEA 1994). The Russian legal criterion for 'equipment failure or active reactor zone melting' is 1.0×10^{-5} per reactor year (Tsaturov 1996).

Most of the studies in the literature dealing with the consequences of nuclear accidents do not take into account the probabilities of abnormal events. They also use varying assumptions of source terms, meteorological conditions, seasons, combinations of extreme conditions, etc., and are therefore difficult to compare. However, they are still useful for emergency preparedness planning. Emergency response preparedness is, in practice, also an important consideration when considering the ultimate consequences of an accident. Some studies take the efficiency of protective measures into account when predicting accident consequences.

As an example, we can qualitatively compare the Loviisa NPP in Finland and the Kola NPP in Russia as they are both of the VVER-440 design (Rantalainen 1995). Kola NPP Units 1 and 2 are of model 230 which does not have an accident localization system. The design basis accident is a break of a primary circuit pipe with a diameter of 200 mm. Kola NPP Units 3 and 4 are of model 213 having accident localization systems. The design basis accident is a break of a pipe having a diameter of 500 mm. The accident localization system is not leak-tight like a containment, but consists of rooms, suppression pools and sprays to delay and reduce any radioactive releases. The Loviisa NPP has two units of VVER design model 213. The design basis accident is a double-ended break of a 500 mm diameter primary circuit pipe. Both units have free-standing steel containments surrounded by concrete walls with internal and external spray systems. The Kola and Loviisa NPPs have several positive common features, e.g., the water volumes in the primary and secondary circuits are large compared with those in typical western PWRs making energy densities low and fuel cooling effective. Despite its positive features, the Kola plant has some special risk characteristics, namely:

- Two reactors are in the same building.
- There is no leak-tight full-scale containment.
- Emergency cooling systems are more limited.
- The axes of the turbo-generators are parallel to the reactors, causing a higher missile risk to the control room and reactor cooling systems in the event of turbine disintegration.
- Fire risks are relatively high.
- Redundancy of the safety systems is lower as they are not designed for large leaks or severe accidents.
- During an accident, the high ventilation stack will be closed and the radionuclides will be blown, via openings, to the inner yard of the plant site.
- In accidents where the radionuclides are released straight above the reactor, no system has been designed to control releases or to mitigate the consequences.
- In a core melt accident, there is apparently a higher probability of larger releases than in units fitted with containment.

Because of the similar thermal-hydraulic features in the Loviisa and Kola NPPs, the accident analyses made for the Loviisa plant can also be used to illustrate the behavior of the Kola NPP. However, the probability of a core damage accident in the Kola NPP cannot be assessed from the results of the PSA made for Loviisa NPP. A systematic PSA for Kola NPP would be the only way to estimate the core damage risk for the plant and such an assessment has not been made available to the AMAP radioactivity assessment group. However, the assessment group was informed that the present technical and protection devices are not adequate to retain the radioactive products inside the plant in cases of severe low-probability accidents. Accordingly, the consequences would exceed those of the design basis accident (Tsaturov 1996). The core damage frequency for the Loviisa plant is of the order of 1.0×10^{-4} per year, which is slightly higher than the value for the other Finnish plant at Olkiluoto comprising BWR-type reactors (STUK 1991). The present PSA Level 1 studies for all Swedish reactors result in an estimated core damage frequency of 1.0×10^{-5} per year (Swedish Institute for Radiation Protection and Swedish Nuclear Power Inspectorate 1995).

8.6.1.3. Studies to assess the consequences of major reactor accidents

There are considerable shortcomings in the analyses available to the AMAP radioactivity assessment group that allow conclusions to be drawn about the probability and consequences of potential accidents in nuclear power plants in the Arctic. Despite these limitations, some general conclusions can be drawn from consequence analyses which focus on short-period consequences and external doses. It should be stressed, however, that without associated probability values, consequence analyses are a less than satisfactory basis for such an assessment. Consequence analyses have been made for most NPPs of relevance to the Arctic and provide some idea of the order of magnitude of consequences that can be expected if a severe accident occurs.

Studies on two different types of reactor will be given as examples. In both cases, the scenario can be considered to represent a severe, but not worst-case, accident. Neither scenario, however, represents an extremely unlikely combination of situations although no calculated probability for these cases can be given. The first example is a study of one 1000 MW RBMK unit of the four at the St. Petersburg (Leningrad) NPP (Ilvonen *et al.* 1994). This is the same type of reactor as the one at Chernobyl that suffered an extremely serious accident in 1986.

Improvements have been made at the St. Petersburg plant since the Chernobyl accident. Therefore, the release for the study is assumed to be smaller. It corresponds to 100% of the noble gases, 10% of the more volatile radionuclides, such as radioiodine, radiocaesium, etc., and 1% of the other

fission product radionuclides. A 3-hour release at heights of 20 and 100 m is assumed.

The calculations give the following orders of magnitude for dose rates and doses for outdoors external and inhalation pathways north of the Arctic Circle:

- External dose rate from cloud gamma is 0.01-0.1 μSv/h 30 hours after the release.
- Dose received from cloud gamma within 96 hours to a person outdoors is 0.1 to 10 μSv.
- Dose commitment from radionuclides inhaled outdoors during 96 hours is 10-100 μSv.
- Dose from deposited gamma in the first year is 0.01-1 mSv.

These values are averages of about 3000 different dispersion situations, representing expectation values with no presumptions regarding the meteorological conditions at the time of the release.

The food pathways are not described because they are highly dependent on the season in which the accident occurs and on local habits, which were not taken into account sufficiently in the study. However, the doses from local foodstuffs in the first year would be higher than from surface deposited gamma-emitting radionuclides (assuming both pathways are without protective measures), with the magnitude depending on food habits.

It was concluded by the AMAP radioactivity assessment group that there is no risk to Arctic residents of deterministic health effects from releases at NPPs situated further than about 1000 km from the Arctic Circle. Nevertheless, because of the particular ecological conditions in the Arctic (cf. section 8.7), it is likely that contamination of some food pathways could impose a requirement for protective measures.

The second study given as an example deals with the Kola NPP (Rantalainen 1995) which is situated within the Arctic. It assumes a hypothetical accidental release, at a height of 100 m, of 100% of the noble gases, 10% of the easily volatile radionuclides, and 1% of the less volatile radionuclides of the radionuclide inventory of the core of a 445 MW NPP after a long period of steady operation at full power. The calculations show that, with high likelihood, the doses are less than 1 Sv at distances greater than 5 km from the NPP during the first 24 h, and smaller than 0.1 Sv at distances greater than 30 km. This means that, even without protective measures, acute health effects are not expected at distances greater than 5-10 km. A similar result was obtained by Amosov *et al.* (1995).

The magnitude of external and inhalation doses in the Kola area would be similar to those at any other site but contamination has more severe consequences in terms of dose commitment via food pathways than at lower latitudes because of the ecological characteristics of the region. There is no information available of the probability of accidents of this magnitude. Levels 1 and 2 PSAs would be required to obtain such information. If such an accident were to occur, measures would be needed to ensure adequate protection of the local population close to the NPP against acute health (i.e., deterministic) effects and in the population in the large scale (over thousands of km^2) against late health (i.e., stochastic) effects

8.6.2. Potential accidental releases from nuclear vessels and nuclear storage sites

There are significant risks of accidents during the routine operation of nuclear-powered vessels both military (i.e., those in the navies of China, France, Russia, the United Kingdom and the United States) and civilian (i.e., the ice-

Figure 8·66. Major Russian naval bases along the Kola Peninsula and White Sea.

breaker fleet of the Murmansk Shipping Company). The experience of previous radiation accidents on Russian nuclear-powered vessels (see section 8.5) indicates that accidents in the open sea, both underwater and on the surface, pose neither threats of serious radioactive contamination of the terrestrial environment nor significant public exposures. In these events, however, vessel crews are usually subjected to significant radiation exposure. In contrast, accidents involving nuclear-powered vessels located in harbors and coastal bases, which may occur during fueling operations (refueling, unloading of nuclear reactors, etc.), can pose significant radiation risks to the population of surrounding territories. The following discussion of potential accidents occurring with nuclear-powered vessel reactors is based entirely on information provided by the Russian Federation.

Because of the presence of the Russian Northern Fleet bases (Figure 8·66), northwest Russia contains the highest concentration of nuclear-powered vessels and nuclear reactors in the world. In addition, the civilian Russian icebreaker fleet is based in Murmansk. For comparative purposes, the number of nuclear reactors operating in northern Russia in military and civilian categories with their aggregate inventories of ^{90}Sr plus ^{137}Cs are shown in Table 8·43.

Table 8·43. Nuclear reactors operating in northwest Russia.

Type of reactor	Number of reactors	Approx. aggregate inventory, (^{137}Cs + ^{90}Sr), PBq
Military naval	120[a]	2000
Civilian naval	11	200
Nuclear power plant	4	300

a. Approximate number (NEFCO 1996).

Disposal of decommissioned nuclear-powered submarines represents a complex technical, scientific, ecological and economic challenge. About 99% of the total radionuclide inventory in submarine reactors is located in the spent nuclear fuel (SNF). Accordingly, it is important to focus on safe defueling procedures and on the short and long term storage and disposal of spent nuclear fuel. At present, spent nuclear fuel with an activity of about 2000 PBq (^{137}Cs + ^{90}Sr) is stored on land (40%) and on submarines taken out of service (60%). The Murmansk Shipping Company also has an estimated inventory of 500 PBq (^{137}Cs + ^{90}Sr) in a range of wastes stored on ships at its base.

At the moment, a complete assessment of the risks and impacts in relation to possible accidents with Russian spent nuclear fuel does not exist. However, some information is available, the most relevant of which pertains to potential

accidents associated with nuclear submarine decommissioning which has been provided to the AMAP radioactivity assessment group by the Russian Federation (RCRA 1997). The results of radiation and safety analysis for submarine refueling operations at the *Sevmash* enterprise in Severodvinsk (on the coast of the White Sea) are presented in this report. These analyses demonstrate that, in the case of design basis accidents, no inadmissible concentrations occur outside the sanitary zone designated for refueling operations. Radiation risks to the public are associated with beyond-design basis accidents and the report presents the results of the analysis of such accident scenarios involving chain reactions.

The most serious design-basis accident considered was a loss of integrity of the primary cooling circuit as a submarine approaches the shipyard location for refueling. An assumption was made of a rupture of a primary circuit pipe with an aperture of 10 mm. About 90% of the gaseous fission products and 8% of the other radionuclides in the coolant would be released to the reactor compartment. It was further assumed that the reactor compartment loses its integrity and all such releases enter the atmosphere. However, wind transport is not predicted to result in radionuclide activity concentrations outside the restricted or 'sanitary' zone exceeding the values prescribed for areas around nuclear power stations. On the basis of this analysis, it is concluded that design-basis accidents do not result in undue radiological dangers to the surrounding population in the urban area of Severodvinsk.

The largest beyond-design accident scenario that has been considered during submarine reactor refueling operations is the ejection of two or more control rods from the core of the reactor when either the primary circuit is pressure tested without the control rod retainers being in place (as a result of an operator error) or through capsizing of the vessel without the control rod retainers in place. Such an accident would result in a criticality event with 0.5×10^{20}-1.5×10^{20} fissions. This is sufficient to melt the reactor core. The core would achieve a temperature sufficient to melt the fuel rods within three seconds following the ejection of the control rods. It is assumed that all of the fuel rods lose integrity as a result of such an accident. Four other specific scenarios for beyond-design basis accidents were considered:

Type 1: Chain reaction in the fresh core to be loaded (not relevant to decommissioning).
Type 2: Chain reaction in the spent core with a cooling time of 30 days.
Type 3: Chain reaction in the spent core with an cooling time of 90 days.
Type 4: Chain reaction in the new core following dockside trials (the reactor is assumed to have worked for a month at 30% of the nominal power rating with a subsequent cooling time of 1 day) (not relevant to decommissioning).

It was assumed that the radionuclides are released into the atmosphere at a height of 20 m and that the contaminant cloud is transported and dispersed as a result of turbulent diffusion and advection. The exposure of individuals after the accident occurs through four pathways:

- External exposure from the cloud.
- Internal exposure due to inhalation.
- External exposures from contaminated surfaces.
- Internal exposures from the long-term consumption of contaminated agricultural and fisheries products.

The individual doses can be segregated into three categories:

- Emergency dose in the first few hours after an accident.
- Short-term dose up to ten days after an accident.
- Medium-term dose for a period of up to a year after an accident.

In these contexts, particular significance is given to the thyroid dose.

Under the basic criteria and requirements for ensuring safety in the siting of nuclear power plants set down by the State Atomic Inspection Authority (Gosatomnadzor), total and thyroid doses of less than 5 mSv and 50 mSv, respectively, do not require protective measures following an accident. The foregoing analysis showed that, while the public beyond the shipyard would not be at great risk, protective measures would clearly be required in relation to radioiodine and inhalation doses for those within the sanitary zone. The cited document then deals with the requirements for protective actions for this latter group of individuals.

It still remains for the probabilities and doses associated with accidents during the transport, storage and disposal of spent fuel from submarine reactors to be evaluated.

The summary conclusions of the Russian assessment are as follows:

- The beyond-design accident during dockside trials with a newly loaded core does not lead to significant radiological consequences to the near-field population. Exposures to individuals in the population outside the shipyard do not exceed 5 mSv and no special protective measures are required. Within the shipyard, protective measures against radioiodine ingestion and protection of respiratory organs and skin may be required.
- The major hazard in the cases of accidents involving a chain reaction in spent cores is associated with the release of ^{137}Cs and subsequent surface contamination. In such cases, decontamination of the shipyard and, possibly, parts of the territory beyond the shipyard, would be required.
- The dose to workers in the shipyard (excluding the personnel servicing the nuclear propulsion plant at the time of the accident) within two hours of a reactivity accident in a spent core can be as high as several sieverts which can result in some deaths. The implementation of an emergency contingency plan could ameliorate these doses significantly.
- In the case of an accident near to the slip docks under adverse wind conditions, a short-term dose to about 5000 city dwellers in Yagry could be higher than 0.5 Sv which would require immediate evacuation of this portion of the population.
- The short-term dose to another 30 000 to 35 000 of the city's population would range between 0.05 and 0.5 Sv.
- If a criticality accident in a new core during dockside trials were to occur, total and thyroid doses could lead to deaths among workers at the shipyard. However, the much reduced radiocaesium isotope release, compared to accidents involving spent cores, means that the consequences for the external population are much less severe.

This is the only study of its type made available to the AMAP radioactivity assessment group and, although its validity cannot be judged in isolation, it appears to be the kind of safety assessment that would be warranted for evaluating the probability and consequences of reactor refueling accidents.

8.6.3. Potential releases from reprocessing plants

The primary issue of concern here is the possibility of accidents in nuclear fuel reprocessing plants that could result in major releases of radionuclides to the environment. In an Arctic context, accidents in western European (i.e., at Sellafield and Cap de la Hague) and Russian (e.g., Mayak) reprocessing plants would be of primary concern. Unfortunately, no evaluations of the potential for accidents and associated consequences have been provided to the AMAP radioactivity assessment group on which to base an analysis.

Also of relevance here are accidental releases from environmental reservoirs into which discharges from nuclear fuel reprocessing plants have previously taken place. These reservoirs include those associated with Russian nuclear fuel reprocessing operations in the drainage basins of the Ob and Yenisey Rivers and coastal marine sediment areas close to the discharges from nuclear fuel processing operations in western Europe. Considering that the remobilization of radionuclides from western European coastal reservoirs is likely to be of limited importance to the Arctic, emphasis here has been devoted to the remobilization of radionuclides derived from Russian reprocessing operations in the Ob and Yenisey drainage basins.

There has been widespread recognition that nuclear facilities in central Siberia have the potential to release significant quantities of radionuclides to tributaries of the Ob and Yenisey drainage basins, which ultimately discharge into the Arctic Basin. Several open reservoirs of radionuclides from previous releases from nuclear fuel reprocessing activities exist in the drainage basins of these major rivers. Concerns regarding potential releases in the present context relate to the behavior, movement and effects of radionuclides currently retained within fluvial systems and the consequences of possible future accidental releases from fuel reprocessing operations. In view of the considerable distance of central Siberia from the Arctic, consideration is given here only to the future mobilisation and the effects of radionuclides currently retained in environmental reservoirs as a result of previous operational releases.

8.6.3.1. Mobilisation of radionuclides released to the terrestrial environment

Modeling has focused particularly on ^{90}Sr because of its relatively high aqueous mobility (i.e., low particle reactivity). The models have conservatively assumed no interaction with sediments or biota. At Salekhard (the Ob river/estuary interface), the current flux of ^{90}Sr is assumed to be dominated by surface runoff of atmospheric fallout with, on average, an annual discharge from the Ob River of around 10-40 TBq. This flux has been used as a benchmark against which the effects of scenarios of additional release upriver can be compared using fluvial transport models.

A model analysing available data relating to the movement of radiostrontium, radiocaesium and plutonium through the Ob River system has been developed by Paluszkiewicz *et al.* (1995). Transport of radioactive contamination downstream is governed by meteorological/hydrological forcing, decay, sediment partitioning and radionuclide release conditions. Based on an estimate of the current activity concentration of ^{90}Sr in the River Ob at Salekhard of 25-100 Bq/m^3, these authors have calculated scenarios for releases from the storage reservoirs at Mayak, both on a scenario of dam failure, assuming total release of the inventory within a year, and on scenarios of a steady flux into the river through groundwater. They calculate that failure of ponds 10 and 11, the most likely ponds to be subject to such an event, could raise ^{90}Sr activity concentrations at Salekhard to 1000-2500 Bq/m^3. The scenario of release into groundwater has also assumed release of the complete inventory of ponds over a time determined from information on current fluxes in groundwater. The estimates obtained, which represent a highly conservative worst-case scenario, suggest a flux at Salekhard of 11 000 Bq/m^3 from pond 9 (Karachay), and 10-20 Bq/m^3 from ponds 3 and 10.

The final scenario considered is that of release of activity from the Asanov Swamps. Remobilization of the total inventory (a worst-case scenario) would involve drying out of the marsh and flora and subsequent flood events washing organic matter down the River Ob. This scenario would involve the release of an additional 327 TBq of ^{90}Sr within one year.

The movement of radioactive contaminants in the Yenisey River system has been studied by Vakulovsky *et al.* (1993). They found that, of the principal contaminants, ^{24}Na and ^{51}Cr were transported in dissolved form, whereas ^{46}Sc, ^{54}Mn, $^{58/60}$Co, ^{59}Fe and ^{65}Zn were transported predominantly by suspended particles. Transport of ^{32}P and ^{137}Cs was more evenly distributed among solution and sediment phases. These differences were reflected in the activity concentrations of the radionuclides in water samples. The authors concluded that, of the long-lived radionuclides, only ^{137}Cs could reach the Kara Sea, with an average discharge of between 0.8 and 2.8 TBq/y. Panteleyev *et al.* (1995) concluded that most of the ^{137}Cs and ^{239,240}Pu in sediments of the Ob delta was derived from nuclear weapons testing and that the contribution from central-Siberian reprocessing plants was small by comparison.

The model estimates for the discharge of ^{90}Sr assume complete mobility. As the transport of radionuclides will involve both the dissolved fraction and transport of contaminated sediments, in reality the rate of diffusion toward the Kara Sea will depend on interactions with soils, sediments and biota. For instance, Trapeznikov *et al.* (1995) found that, in the River Techa, 250 miles downstream of Mayak, the activity concentration of ^{90}Sr in water was roughly halved, ^{239,240}Pu decreased fourfold and ^{137}Cs decreased by an order of magnitude, reflecting the relative particle reactivities of these radionuclides. The rivers carry the greatest sediment load at times of greatest flow during the late spring-melt.

The rate of transport downriver will therefore be influenced both by the distribution of a contaminant among soluble and particulate phases and its affinity for different particle size fractions. Tronstad *et al.* (1995) found that ^{90}Sr was remobilized from Asanov Swamp sediments more readily than ^{137}Cs or ^{239,240}Pu. They concluded that the Asanov Swamp area could act as a long-term source of contamination of the River Techa.

The sorption of a radionuclide is often modeled using an equilibrium distribution or partition coefficient, K_d. A lower K_d indicates enhanced mobility, i.e., a greater proportion of the radionuclide is in the soluble phase. In an open system, such as a river, changes in physical, chemical and biological parameters can influence the K_d value. Radionuclides can be transferred between sediment and water phases by chemical, physical or biological processes. Physical-chemical changes in water characteristics can change the equilibrium partitioning conditions resulting in sorption or release of radionuclides. Chemical mobilisation includes ion-exchange, leaching and dissolution; biological processes can effect both

chemical and physical mobilisation. In addition, physical transport can occur due to natural or anthropogenic resuspension of sediments.

Changes in ionic composition of water can affect K_d significantly. K_d values for both ^{90}Sr and ^{137}Cs in river water and ice are higher than those in seawater and sea ice. The mobilisation of sediment-associated ^{137}Cs and ^{90}Sr may be caused by changes in pH, ionic strength, salinity and/or concentrations of exchangeable elements.

Ice may be an important mechanism for the transport of sediment-bound radionuclides in the Arctic Sea. The extent to which sea ice could have a role in transporting sediments deposited in estuaries is not well known. It appears that contaminated sediments can be transported considerable distances in the sea ice mass. The direction of transport is much dependent on the drift of ice within the polar pack. Meese *et al.* (1995) suggest that sea ice is a primary transport mechanism by which contaminated sediments are redistributed throughout the Arctic Ocean.

8.6.3.2. **Mayak**

The Mayak facility is situated around the headwaters of the River Techa, which ultimately drains into the Kara Sea via the Ob River. The system of waste management has been dependent on a series of natural and artificial reservoirs and drainage canals. A total of 4000 PBq (decay corrected to 1994) comprising mainly ^{137}Cs and ^{90}Sr has been released to this restricted system (NRPA 1997). The principal risks of release are from leakage through the walls of dams and conduits and from drainage through lakebeds into groundwater. This latter scenario is relevant to the situation around Lake Karachay. A further source is existing contamination in the Asanov Swamps that has accumulated from accidental and routine discharges during the period of operation of the plant.

Failures in the waste containment in this system could result in discharges of radionuclides to the Techa River. Such failures include partial or total dam failure, leakage from the reservoirs into the neighboring diversionary conduits that discharge into the Techa River below the lowest reservoir (reservoir no. 11), and overflow of reservoirs as they approach capacity. Total failure of any of the dams, but particularly of dam no. 11, would potentially release a great volume of contaminated water and sediment into the Techa River and Asanov Swamps below the dam. Reservoir no. 11 has a volume of 216×10^6 m^3.

Further release scenarios identified include filtration of contaminated water through the dams of reservoirs nos 10 and 11, and leakage through the beds of the reservoirs into the aquifers beneath. Leakage is monitored through a system of boreholes. The potential importance of the former pathway may increase as it is thought that the sorption capacity of the rock strata in the vicinity of reservoir no. 11 is almost exhausted and contamination will enter the River Techa more readily than previously. Contamination of groundwater from the smaller storage reservoirs nos 3 and 4 has not been determined.

Leakage of waste from reservoirs nos 10 and 11 has been identified in the diversionary conduits that discharge into the Techa River below the reservoirs as well as in water seeping through the dam of reservoir no. 11. The rate of seepage increases with increasing volume in the reservoir. The discharge of ^{90}Sr into the Techa River after seepage through the dam of reservoir no. 11 has increased steadily during the last 15 years. Despite some remedial action in the period 1990-1993, a peak discharge of 27 GBq was recorded in 1993. This was due to an increase in reservoir volume after an unusually wet period. Similarly, discharge of ^{90}Sr through the right- and left-bank conduits reached 14 GBq and 100 GBq, respectively, in 1993 (NRPA 1997).

The risk of resuspension of contaminated spray and sediments has been mitigated by a project to fill in the lake by depositing concrete sections on the lakebed. However, the potential remains for contamination of the water table through expansion of a lens of contaminated groundwater resulting from diffusion of radionuclides through the lakebed. Currently, the lens of contaminated groundwater extends over an area of about 10 km^2 beneath Lake Karachay to a depth of around 100 m. The lens straddles a subterranean watershed with complex hydrogeology. Depending on the nature, capacity and replenishment of the aquifers, expansion of the lens can be northward toward the area of the storage ponds or southward toward the River Mishelyak, a minor tributary of the Techa River. Lateral expansion of the lens in this area may be enhanced by the use of groundwater as a source of potable water by some local villages. The primary radiological concern arises because of the mobility of ^{90}Sr, which may be enhanced by the presence of dissolved organic compounds in the discharged waste. Caesium-137 is less mobile and is largely sorbed to lake sediments.

Severe flooding of the Asanov Swamps, perhaps in connection with failure of dams in the pond chain above, could result in substantial releases of ^{90}Sr and ^{137}Cs to the Techa River system.

8.6.3.3. **Tomsk**

Storage ponds at the site are believed to be contaminated to a similar degree as Lake Karachay. They contain an estimated 1.3×10^8 Ci (≈4800 PBq). As at Mayak, there are concerns about potential contamination of groundwater, compounded by the proximity of the Tom River and the relatively high groundwater level, the water table of which is only 20 m beneath the surface.

8.6.3.4. **Krasnoyarsk**

The storage ponds at Krasnoyarsk are believed to contain an inventory of about 5×10^4 Ci (≈2 PBq). As at the other sites, there is a risk of contaminated groundwater migrating into rivers, in this case the Yenisey River.

8.6.4. **Radioactive wastes dumped at sea**

The former Soviet Union dumped high, intermediate and low level radioactive waste in the Arctic seas during the years 1959-1991. In spring 1993, the Russian Federation published a report, the so-called 'White Book' (Office of the President of the Russian Federation, OPRF 1993) that included information on sea dumping operations. According to this report, the total amount of radioactivity dumped in the Arctic Seas was approximately 90 PBq at the time of dumping. The items dumped included six nuclear submarine reactors and a shielding assembly from an icebreaker reactor each containing spent fuel with an aggregate inventory of 85 PBq; ten reactors (without fuel) containing 3.7 PBq; liquid low-level waste containing 0.9 PBq; and solid intermediate and low-level waste containing 0.6 PBq. The packaged and unpackaged solid waste and the nuclear reactors were dumped in the fjords of Novaya Zemlya at depths of between 12 and 135 m, and in the Novaya Zem-

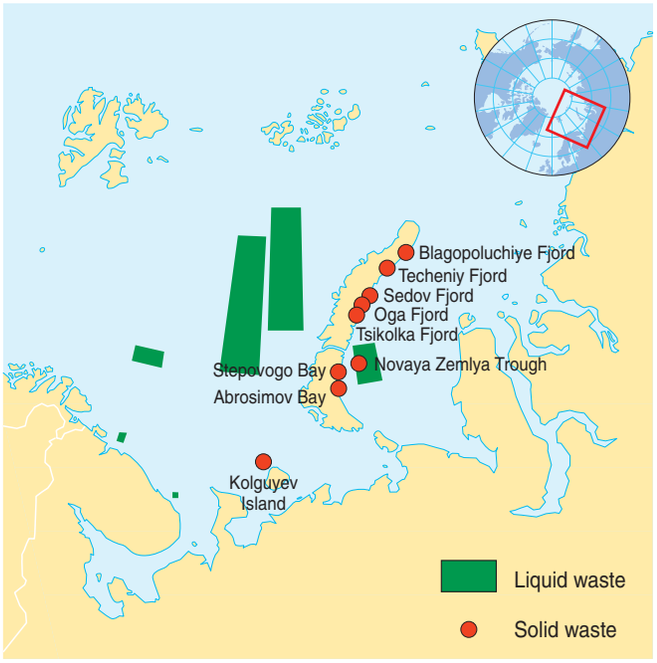

Figure 8·67. Locations of sea dumping of radioactive waste in the Russian Arctic.

lya Trough at a depth of 380 m. The liquid low-level waste was discharged in the open Barents and Kara Seas. Figure 8·67 shows the locations of these dumping sites. Tables 8·44, 8·45 and 8·46 list types, locations and radioactivity of objects containing spent nuclear fuel (SNF), objects devoid of SNF, and solid low and intermediate level wastes, respectively, dumped in Arctic marine areas as given in the 'White Book' (OPRF 1993).

8.6.4.1. Surveys of dumped objects

A Joint Norwegian-Russian Expert Group was established in 1992 to investigate radioactive contamination of northern areas as a result of the dumping of nuclear waste in the Barents and Kara Seas. Exploratory cruises to the dumping areas were conducted in 1992, 1993 and 1994. Using high-frequency side-scan sonar and a remotely operated vehicle (ROV) equipped with an underwater video camera, a NaI-detector and a sediment sampler, attempts were made to identify and examine the dumped wastes. All of the four sites where spent nuclear fuel was dumped were visited, but only some of the objects containing high level waste were successfully located. However, dumped vessels and numerous containers of low level waste were also located.

The Norwegian-Russian Expert Group also took sediment, seawater and biota samples, both in the immediately vicinity of the dumped objects and in the surrounding area. The levels of radionuclides in waters, sediments and biota in the Kara Sea are very low compared to other marine systems, e.g., the Irish, Baltic and North Seas (NRPA 1996). Nevertheless, levels in the immediate vicinity of dumped low level waste containers indicate that some leakage has occurred.

8.6.4.2. International Arctic Seas Assessment Project (IASAP)

While it appears that no significant global or regional effects have yet resulted from the dumping of radionuclide waste in the Arctic, there is concern about the gradual deterioration of the waste containments that could lead to releases of radionuclides in the future. This could result in contamination of the marine food chain and increased radiation exposures of human consumers of fish and other marine foodstuffs.

Table 8·44. Dumped objects containing spent nuclear fuel in Arctic seas (OPRF 1993).

Dumped objects	Year	Location	Depth, m	Total activity, PBq[a]	Radionuclide content
Compartment of nuclear submarine no. 285 with two reactors, one containing SNF	1965	71°56'2"N 55°18'5"E Abrosimov Fjord	20	29.6	Fission products
Compartment of nuclear submarine no. 91 with two reactors containing SNF	1965	71°56'2"N 55°18'9"E Abrosimov Fjord	20	14.8	Fission products
Shielding assembly of reactor from OK-150 unit of icebreaker *Lenin* containing residual SNF (60% of fuel complement	1967	74°22'1"N 58°42'2"E Tsivolka Fjord	49	3.7	^{137}Cs (≈1.8 GBq), ^{90}Sr (≈1.8 GBq), {^{238}Pu, ^{241}Am, ^{244}Cm} (≈70 TBq)
Reactor from nuclear submarine no. 421 containing SNF	1972	72°40'N 58°10'E Novaya Zemlya Trough	300	29.6	Fission products
Nuclear submarine no. 601 with two reactors containing SNF	1981	72°31'15"N 55°30'15"E Stepovogo Fjord	50	7.4	Fission products
Total (5 objects with 7 reactors containing SNF)	1965-1981			85.1	

a. Expert estimates were made at the time of dumping.

Table 8·45. Objects devoid of spent nuclear fuel dumped in Arctic seas (OPRF 1993).

Dumped objects	Year	Location	Depth, m	Total activity	Radionuclide content
Reactor compartment of nuclear submarine no. 254, containing two reactor assemblies	1965	71°55'13"N 55°32'32"E Abrosimov Fjord	20	Requires special analysis	
Reactor compartment of nuclear submarine no. 260, containing two reactor assemblies	1966	72°56'2"N 55°8'5"E Abrosimov Fjord	20	Requires special analysis	
Nuclear power plant of icebreaker *Lenin* containing three OK-150 reactors with primary coolant lines	1967	74°26'4"N 58°37'3"E Tsivolka Fjord	50	≈1.9 PBq	Mainly ^{60}Co
Two reactors from nuclear submarine no. 538	1972	73°59'N 66°18'E Techneniye Fjord	35-40 35-40	Requires special analysis	
Total (5 objects with 9 reactors without SNF)	1965-1972			<3.7 PBq at dumping	

Table 8·46. Low and intermediate level radioactive waste dumped in the Kara and Barents Seas (OPRF 1993).

	Number of disposals	Years dumped	Activity, TBq
Solid wastes			
Novaya Zemlya Trough	22	1967-1991	123
Sedov Fjord	8	1982-1984	126
Oga Fjord	8	1968-1983	75
Tsivolka Fjord	8	1964-1978	99
Stepovoy Fjord	7	1968-1975	47
Abrosimov Fjord	7	1966-1981	25
Blagopoluchiye Fjord	1	1972	8
Techeniye Fjord	3	1982-1988	68
Off Kolguyev Island	1	1978	1.5
Zornaya Bay (Novaya Zemlya)	1	1991	11
Barents Sea	1	?	>4
Liquid wastes (Volume >189634 m^3)	About 100 operations at 5 sites plus 6 accidental releases elsewhere		900
Total			≈1500

Because the wastes are lying in shallow waters, the possibility of radiation exposure by other routes, such as direct exposures following the movement and transport of the waste packages by natural events (ice or storm action) or human actions cannot be totally ruled out. The International Arctic Seas Assessment Project (IASAP) was established by the IAEA in 1993 to address these and other related issues partly at the request of the London Convention 1972.

The objectives of the IASAP study were:

- To assess the risks to human health and to the environment associated with solid radioactive waste dumped in the Kara and Barents Seas; and
- To examine possible remedial actions related to the dumped solid waste and to advise on whether they are necessary and justified.

The IASAP project was carried out by a multidisciplinary team of scientists from several countries within the normal procedures of the IAEA. The following approach was adopted:

- Examination of the current radiological situation in Arctic waters to assess evidence for releases from the dumped waste;
- Prediction of potential releases from the dumped wastes concentrating on the solid high-level waste objects containing the majority of the radionuclide inventory;
- Modeling of the environmental transport of released radionuclides and assessing the associated radiological impact on humans and biota;
- Examination of the feasibility, costs and benefits of possible remedial measures applied to a selected high-level waste object.

The study was divided into a series of five working areas: 1) source term reconstruction, 2) existing environmental concentrations, 3) transfer mechanisms and modeling, 4) radiological impact assessment, and 5) assessment of remedial measures.

8.6.4.2.1. **Source term reconstruction**

The information needed about the dumped radioactive wastes for the purposes of assessing the radiological impact and evaluating the need for remedial measures was acquired and evaluated by a special working group of IASAP. The work involved obtaining knowledge of the characteristics of the steam generating installations and nuclear fuel, data on reactor operating history, detailed inventories of the radionuclide composition of the wastes and the likely behavior of protective barriers with time. Attention was focused on the high level waste, i.e., the dumped reactors and the container of spent fuel from the icebreaker *Lenin* which obviously pose the highest potential risks.

During the IASAP project, general information about the actual dumping operations was obtained (Yefimov 1994, IAEA 1996). Fuel had been removed from ten of the reactors prior to dumping. Those dumped with spent fuel (six reactors) had usually suffered an accident prior to dumping, in which the fuel had been damaged. The dumping of the reactors took place by four methods: 1) Most of the submarine reactors were dumped within their respective reactor compartments. 2) In some cases, the reactors were taken out of the compartment and placed in a special metal box prior to dumping. 3) In the case of the lead-bismuth cooled reactors, the submarine compartment was filled with bitumen and the entire submarine was dumped (in this case, the solidified liquid metal coolant forms an additional protective barrier). 4) The dumped components of the nuclear icebreaker include a reactor compartment with three reactor vessels from which the fuel was removed. About 60% of the fuel from one of the reactors was dumped in a separate metal lined concrete box. All of the reactors containing nuclear fuel and the icebreaker fuel box were filled with a special polymer mixture, Furfurol™.

The total activity of the dumped reactors (with and without nuclear fuel) at the time of dumping was re-estimated by IASAP to be 37 PBq. This may be compared with the first estimate of ca. 89 PBq provided in spring 1993 by the Russian Federation (see Tables 8·44 and 8·45). The total inventory of the dumped reactors had, by 1994, declined through radioactive decay to an estimated 4.7 PBq (Sivintsev 1994a, 1994b, Yefimov 1994, IAEA 1996). The peak of radioactivity of the dumped material, 25 PBq, was reached in 1967, when spent fuel from one of the reactors of the icebreaker was dumped (Figure 8·68) (IAEA 1996, Sivintsev 1995, NRPA 1996).

Construction details of the steam generation installations were analysed to determine the likely ingress routes for sea water and the time after dumping at which this occurs. The evaluation of radionuclide release rates was based on an analysis of the weak points of the protective barriers. Release was assumed to be controlled by corrosion of the ma-

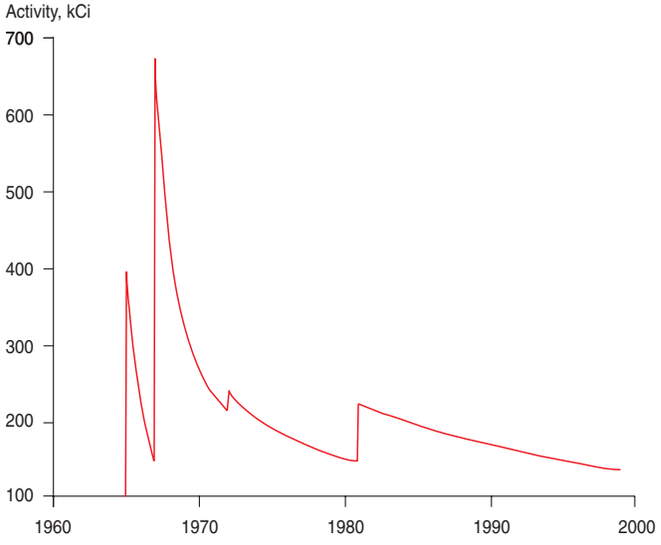

Figure 8·68. Predicted release rates of different radionuclide groups from the submarine reactor dumped in the Novaya Zemlya Trough (best estimate scenario).

terials forming the reactor structure and nuclear fuel. The best available predictions for corrosion rates in an Arctic environment were derived from simple computer models of containment failure.

It was assumed that all corroded material is immediately released to the environment. This is a highly conservative assumption, as most of the corroded matter will be both heavy and insoluble and will be retained in the hull or reactor pressure vessel until other barrier corrosion is well advanced.

The following release scenarios were used for impact assessment calculations:

(A) A ***best estimate scenario*** – release occurs through gradual corrosion of the barriers, waste containers and the fuel itself.
(B) A ***plausible worst case scenario*** – normal gradual corrosion followed by catastrophic disruption of two sources at a single dump site (the fuel container and the reactor compartment of the icebreaker) in the year 2050 followed by accelerated release of the remaining radionuclide inventory of these sources.
(C) A ***climate change scenario*** – corrosion up to the year 3000 followed by instantaneous release, due to glacial scouring, of the radionuclide inventory remaining in all sources.

The release rates with time resulting from each of these three scenarios constitute the respective input terms for the modeling of environmental transport and exposure pathways.

8.6.4.2.2. Consideration of possible criticality

A preliminary analysis of the possibilities of criticality in the dumped reactor assemblies concludes that run-away criticality ($k \gg 1$) is so extremely unlikely, given the moderating conditions, as to be ruled out. Nevertheless, the probability of some criticality in some of the reactor assemblies is somewhat higher and this might accelerate corrosion of barriers and lead to enhanced release of radionuclides. If this occurred *in situ*, it is unlikely to be of major concern, especially as the inventory of the reactor assemblies declines with time. If, however, it occurred during recovery of one of the assemblies, it could involve significant risks of direct radiation exposure for those involved in recovery operations. It is therefore suggested that, before any decision to undertake remedial action is taken, a thorough criticality study should be conducted.

8.6.4.2.3. Pathway modeling and radiological assessment

When the IASAP project was launched, there were practically no radiological assessment models dealing with Arctic marine areas. In addition, very little information was available on the oceanographic, sedimentological and biological conditions in the Kara Sea. Thus, the first task toward the radiological assessment of the impact of the dumped waste to human health and environment was to develop realistic marine environmental transport models for the region. The approach taken within the IASAP project was to involve several national modeling groups, each of them extending their models to the target area or creating completely new models. At the same time, through the work of the Norwegian-Russian Expert Group and several Russian and international institutes, improved environmental information was gradually acquired (Strand *et al.* 1997, Pavlov 1994, Ivanov 1994, Sazykina and Kryshev 1994, IAEA 1996).

The IASAP models differ, *inter alia*, in their spatial and temporal resolutions. Thus, the results from different models were used for different specific endpoints. For example, the results from models with greater spatial resolution were used for critical group calculations. For global collective dose calculations, which involve long time scales, radiological compartment models were used.

For individual dose estimation, three critical groups were considered:

- A group living in the Ob and Yenisey estuaries and on the Taimyr and Yamal Peninsulas whose subsistence is heavily dependent on the consumption of locally-caught Kara Sea fish, marine mammals, and seabirds and their eggs, and who spend 250 hours/y on the seashore. These habits are also typical of subsistence fishing communities in other countries bordering the Arctic.
- A hypothetical group of military personnel patrolling the foreshore of the fjords of Novaya Zemlya containing dumped radioactive wastes, for assumed periods of 100 hours/y. The exposure pathways considered include external radiation and the inhalation of seaspray and resuspended sediment.
- A group of seafood consumers considered representative of the northern Russian population situated on the Kola Peninsula eating fish, mollusks and crustaceans harvested from the Barents Sea. No consideration was given to the consumption of seaweed or marine mammals, nor to external radiation.

The total annual individual doses in the critical groups of seafood consumers (Groups 1 and 3) for all three scenarios are small and very much less than variations in natural background doses. For scenario A, the maximum individual dose rates to members of the critical groups in the Ob and Yenisey estuaries, the Yamal and Taimyr Peninsulas and the Kola Peninsula are less than 10^{-7} Sv/y with the doses on the Kola Peninsula being the lowest. These low doses were mainly delivered through fish consumption, with ^{137}Cs, ^{239}Pu and ^{90}Sr being the dominant radionuclides. The maximum dose rates for the other two release scenarios (B and C) are about one order of magnitude higher than those of scenario A. These maximum doses to the general public are essentially trivial. For members of the hypothetical critical group 2, comprising military personnel on Novaya Zemlya, the maximum dose is estimated to be ≈3 mSv/y (for the plausible accident scenario B) derived primarily from external exposure and inhalation. This latter dose is high enough to warrant continued restrictions on the occupancy of the foreshores of the fjords of Novaya Zemlya in which wastes have been dumped.

Collective doses were estimated only for the best estimate release rate scenario (A). The collective dose to the world population arising from the dispersion of radionuclides in the world's oceans (for nuclides other than ^{14}C and ^{129}I) were calculated for two time periods: 1) up to the year 2050 to provide information on the collective dose to the current generation, and 2) over the next 1000 years, a time period which covers the estimated peak releases. The estimated collective doses are 0.01 and 1 manSv, respectively. Appropriate global models were used to calculate the collective doses associated with ^{14}C and ^{129}I, which are very long-lived nuclides and circulate globally in the aquatic, atmospheric and terrestrial environments. Assuming that the entire ^{14}C inventory of the wastes is released around the year 2000, integrating the dose to the world's population over 1000 years into the future (i.e., to the year 3000) yields a collective dose of about 8 manSv. The correspond-

ing value for ^{129}I is much lower at 0.0001 manSv. Thus, the total collective dose over the next 1000 years to the world's population from all radionuclides in the dumped waste is of the order of 10 manSv. In contrast, the annual collective dose to the world's population from naturally-occurring ^{210}Po in the ocean is estimated to be about three orders of magnitude higher and the collective dose from previous sea dumping of low-level radioactive waste in the Northeast Atlantic Ocean is 1 manSv over 50 years and 3000 manSv over 1000 years.

8.6.4.2.4. **Effects on marine organisms**

The IASAP study also included an evaluation of doses to marine organisms arising from the dumped radioactive waste objects and the likelihood of their having effects on populations of such organisms. The doses to marine organisms are orders of magnitude below those at which detrimental effects on populations might be expected to occur. Furthermore, these doses are delivered to only small proportions of the resident populations.

8.6.4.2.5. **Remediation**

The Contracting Parties to the London Convention 1972 specifically requested the IAEA to study possible remedial actions and to advise on whether they are necessary and justified.

A preliminary engineering feasibility and cost study was conducted as a case study for the container of spent fuel from the icebreaker. It was chosen because it has the highest inventory of any of the dumped containers (2200 TBq) and is the best documented in terms of construction and mode of disposal. Furthermore, there is little risk of criticality occurring during remedial measures.

A group of salvage experts defined two broad categories of option for remedial measures that would warrant initial evaluation from the perspectives of engineering feasibility and cost:

- Capping *in situ* with concrete or other suitable material to encapsulate the material.
- Recovery for land disposal.

Both options were deemed to be technically feasible but the costs (US$ 6-10 million) would be very high relative to the potentially avertable dose. The radioactive waste sources in the Barents and Kara Seas are predicted to give rise to future annual doses of less than 1 μSv to individuals in population groups bordering these seas. The risk of fatal cancer induction from a dose of this magnitude is about 5×10^{-8} which is a wholly trivial risk. The collective dose to the world's population over the next 1000 years from the dumped wastes in the Barents and Kara Seas is of the order of 10 manSv. A simplified approach to considering collective dose in a decision-making framework is to assign a monetary value to the health detriment that would be prevented if remedial action was implemented. Such a scoping approach indicates that remedial measures applied to the single largest source (the dumped spent fuel package from the nuclear icebreaker) costing in excess of US $ 200 000 would not appear to offer sufficient benefit to be warranted. If, on the other hand, remedial actions were to be taken for reasons other than radiological ones, it appears that doses to those involved in remedial actions would not be large.

8.6.4.2.6. **Conclusions of IASAP**

The IASAP study concludes that the radiological risks posed to human health and the environment by the radioactive wastes dumped in the western bays of Novaya Zemlya and the Kara Sea are minor. It further concludes that remedial actions are not warranted on radiological grounds. The existing restrictions on the occupations and habits of military personnel on Novaya Zemlya are sufficient to prevent significant radiation exposures to humans occurring. It is, however, recommended that attempts be made to locate and identify all dumped high level waste objects.

8.6.5. Nuclear weapons

History confirms that measurable risks exist of releases of radioactive material following accidents involving nuclear weapons (see, for example, section 8.5.3.2.). Platforms (ships, aircraft, ground vehicles) carrying nuclear weapons can be involved in accidents such as aircraft crashes, fires onboard vessels and loss of vessels or aircraft at sea. There are two main categories of concern in this context: the risk of a nuclear explosion; and the risk of releases of radioactive constituents of weapons to the environment.

Weapons designers have used various features to minimize the risks of nuclear explosion and release of fissile material from weapons deployed in the field. These include:

- Permissive Action Links (PAL), believed to be used on all weapons. PAL is a mechanism that prevents a weapon being armed before an authorized code is introduced to an electronic arming system. PAL is intended to prevent unauthorized ignition outside the chain of command.
- The effectiveness of the PAL system can be easily evaluated in the laboratory.
- Enhanced Nuclear Detonation Safety (ENDS), a mechanism to reduce the probability of a warhead detonation in an accident. However, ENDS is probably not employed in all nuclear weapons.
- One Point Safety, intended to eliminate the possibility of criticality being attained if the conventional explosive ignites at a single point. Implosion weapons can only be properly initiated if a precise sequence of a series of conventional explosive detonations occurs. One Point Safety prevents the correct detonation sequence being initiated if a single conventional explosive component is detonated. One Point Safety is believed to have been introduced in all weapons.
- Insensitive High Explosives (IHE), these are chemical high explosives that are less easily detonated by impact or fire than are conventional high explosives. A large proportion of the USA stockpile of nuclear weapons does not employ conventional charges containing IHE and it is not known whether similarly insensitive chemical explosives are used in weapons manufactured by other nuclear countries.
- Fire-Resistant Pits (FRP), these are pits (i.e., Uranium-Plutonium weapon cores) that are covered with high-melting-point metal shells that reduce the possibility of dispersing plutonium through fire following an accident. FRP has only been introduced in a few USA weapon designs and is not known to be used by other weapons manufacturing countries.

Thus, PAL is intended to prevent unauthorized detonation. One Point Safety, IHE and ENDS are used to reduce the probability of an unintended nuclear explosion by ensuring

that the conventional charges do not ignite in a manner that initiates a nuclear explosion in the event the weapon is exposed to fire, mechanical shocks or fragment/bullet penetration. FRP is used to reduce the possibility of the dispersion of nuclear material from the core of a weapon in the event of fire.

The problem remains that not all weapons employ all these safety features. Modern weapons design is extremely complicated and no precautions can entirely eliminate the possibility of an unintended nuclear detonation. Nevertheless, an inadvertent nuclear weapons detonation has never been known to occur and the risk is considered low.

In contrast, there have been several cases of releases of radioactive material from nuclear weapons as a consequence of accidents involving weapons platforms. The Thule B-52 aircraft crash (see section 8.5.3.2.) and an incident at Palomares, Spain, in 1966 are merely documented examples of such events. The greatest radiological risks posed by such accidents are the ejection of fine particulate plutonium into air that may be inhaled. By comparison, the radiological risks associated with marine-deposited weapons plutonium and uranium following accidents are relatively minor.

On January 17, 1966, four nuclear weapons were inadvertently dropped from a USA aircraft in the vicinity of the small village of Palomares in Spain. The conventional explosive in one of the bombs detonated and scattered plutonium but no nuclear explosion occurred. A recently-released report from Los Alamos, however, revealed that '*by good fortune*' the weapons involved had just previously been modified after small-scale nuclear tests had uncovered a safety problem. If the weapons had not been modified to reduce the risk, the latter report states that '*the chance of a significant nuclear explosion would have been more than a thousand times greater*'. This was at a time when the USA had experience of more than 430 weapon tests. To date, France has only conducted half as many tests and China about 10%. There may, therefore, be a risk that some of the weapons in the stocks of these countries have less developed safety systems and may pose a greater risk of nuclear detonation in the case of accidents. Altogether, the USA has conducted 1032 nuclear tests and the experience has largely benefited both the United States and the United Kingdom. The former USSR has conducted 715 nuclear tests and this should imply that a relatively sophisticated level of safety design features has been incorporated into Russian weapons.

The most serious concern relates to a possible critical event in a weapon – i.e., an accidental nuclear explosion. This has been prevented to date but, clearly, the fail-safety of nuclear weapons is an area in which public reassurance is warranted. If some accidental ignition of a part of the conventional explosive within a weapon occurred, a full-scale nuclear detonation is unlikely. However, partial criticality from a very small yield to almost full yield cannot be ruled out. The most serious such accident would be one occurring at ground level. The neutron flux from a partial criticality at ground level would also result in the formation of a larger quantity of activation products in the vicinity of the detonation.

Clearly, the risk of releases of radioactive material following accidents involving nuclear weapons is high. Although little information has been made available regarding safety precautions during the handling, storage and deployment of nuclear weapons by either the Russian Federation or the United States, there are reasons for concern about possible accidents involving fires and spillage of fissile material in such operations. There is a need for enhanced information on such activities and assurance that adequate surveillance and safety measures are being implemented.

The dangers associated with weapons are clearly related to concerns about nuclear proliferation. New nuclear nations may not be able to make safe weapons and any use of nuclear material by irresponsible factions or organizations to construct weapons, even crude ones having virtually no incorporated safety features or deployment controls, would immeasurably enhance concerns about nuclear weapons accidents.

8.6.6. Radionuclide thermoelectric generators

Radionuclide Thermoelectric Generators (RTGs) have been used as power sources in remote areas because of their ruggedness and reliability. RTGs are deployed in both Alaska and Russia. The first USA RTG was installed in 1973, and nine additional units were installed in 1985. They were established at remote sites to provide reliable electricity for the seismic stations established for nuclear treaty verification. Electricity is generated by the Seebeck thermoelectric effect in which an electrical potential is generated across a thermocouple exposed to a temperature gradient. The temperature gradient is provided by the radioactive decay of a ^{90}Sr source. The ^{90}Sr capsule is made of strontium titanate, a ceramic material. Depending on the unit, this may weigh between 0.5 and 1.8 kg (i.e., between 1.2 and 3.9 pounds). This material is fire resistant with a melting point above 1100°C (2000°F). It also has a very low solubility in water. There are 10 RTGs remaining in Alaska, located at the US Air Force seismic observatory at Burnt Mountain, Alaska (67°25'N, 144°36'W). As of April, 1994, the total inventory for the 10 RTGs at Burnt Mountain was approximately 26 PBq (700 000 Ci). Each RTG weighs between one and two tonnes including radiation shielding, insulation, thermocouples and housing. The units are designed to ensure that external exposure at a distance of 1 m does not exceed 0.1 mSv/h.

No accidental releases of radioactive material have been associated with the units installed at Burnt Mountain, Alaska. A 1994 evaluation of these units by the US Office of Technology Assessment (OTA 1995) found that, even in the event of a forest fire impacting the observatory site and burning one of the buildings housing a RTG, there is little risk of a release of ^{90}Sr . In the event of a release from the radioactive capsule, the main risk would be associated with radiation exposures to people cleaning up the site. These radiological risks would most likely be posed by external exposure to ^{90}Sr because the strontium titanate is not readily converted into fine particles that are likely to be inhaled. In this form, ^{90}Sr is not readily biologically available to either plants or animals. These RTGs, therefore, pose little risk to the general public and worker exposures are minimized by the use of appropriate safety procedures.

In 1994, there were approximately 155 Russian RTG-powered lighthouses in service in northern areas. Each of these RTGs contains up to 13 PBq of ^{90}Sr (i.e., a total of (2000 PBq). Only one accident involving RTGs has been reported. A helicopter carrying a RTG source crashed off the east coast of Sakhalin Island in 1987 and the source was lost. No enhanced levels of radioactivity have been detected in the area.

Some satellites also employ RTGs as power sources. A US Navy navigation satellite (Transit 5BN) containing a SNAP 9A RTG source fueled by ^{238}Pu, burned up in the

atmosphere over the Indian Ocean in April 1964. As a result, a considerable quantity of ^{238}Pu was released to the upper atmosphere and this has resulted in altered relationships between ^{238}Pu and other plutonium isotopes in atmospheric fallout.

8.6.7. **Summary**

The greatest threats 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 is the potential for accidents in nuclear power plant reactors, during the handling and storage of nuclear weapons, in the decommissioning of nuclear submarines and the disposal of spent nuclear fuel from vessels. The risks posed by radioactive wastes dumped in the marine environment of the Russian Arctic and by radioelectric thermal generators deployed in the Arctic environment are relatively minor.

It is not possible to judge the risks posed by the remobilization of radionuclides previously released from nuclear reprocessing activities currently residing in storage basins of northern Russian river systems because of the limited understanding of the rates and modes of transport of radionuclides within terrestrial environments, especially the effects of episodic events. Unfortunately, neither has it been possible to assess with confidence and quantitatively the risks posed by potential reactor accidents in the Arctic, both military and civil, potential accidents in nuclear reprocessing operations and potential accidents in the handling and storage of nuclear weapons, either because of the limited relevant information made available to the AMAP radioactivity assessment group or because of shortcomings in contemporary safety assessments of these practices.

It is concluded that there is a need for more detailed probabilistic safety assessments of civilian nuclear power plant installations. It is also essential that account is taken of medium and long-term internal doses via terrestrial Arctic pathways. In addition, continuing attention should be paid to the modes and rates of radionuclide mobilisation in major river catchments of Siberia, including groundwater.

8.7. Spatial analysis of vulnerability of Arctic ecosystems

Radioactive doses to populations are derived via two major exposure routes, external and internal; the latter including both inhalation and ingestion pathways. For collective external exposure, distance from source and population density are the major factors which determine vulnerability to a radionuclide release. In the Arctic, specific environmental and morphological factors, such as extent of forest and urban areas or presence of snow cover, could also influence vulnerability to external dose; but in general, assessments of potential external dose can be made from considering potential sources. Internal doses are, however, readily affected by environmental influences, including biological, physical, climatic and socio-economic factors. For this reason, this study will concentrate on internal doses, in particular the ingestion pathway, aiming to identify areas or processes which would be *vulnerable* to deposition from a possible future radioactive release.

Arctic ecosystems vary in their land cover, land usage and soil type. Furthermore, the Arctic environment is exploited by humans in different ways, according to climate, resource availability and socio-economic factors. Variations in ecological, social and other associated factors (e.g., food production) can lead to spatial differences in: 1) radiocaesium intake by humans, and 2) total radiocaesium output (or net flux) from different Arctic areas. This section considers the extent to which contamination of Arctic food products may vary spatially considering, primarily, radiocaesium, but also radiostrontium and radioiodine.

During the initial phase (days to weeks) external radiation, short-lived radionuclides and direct contamination of foodstuffs are of major concern. Fluxes, on the other hand, are mainly relevant to long-term considerations. By using fluxes, it is possible, beforehand, to identify areas and foodstuffs most vulnerable to radioactive contamination. After a nuclear accident, there will probably be an uneven distribution of radioactive contamination and, accordingly, there will be a need to map and characterize the fallout distribution. The priority should be given to areas with the greatest vulnerability, identified from proximity to the source and previously calculated fluxes.

8.7.1. **Sources of radionuclide intake by humans**

The radionuclide intake of an individual depends on both the rate of consumption and the contamination level of each dietary component. Information on dietary composition is available for the average population and, to some extent, for indigenous groups, the latter groups constituting, in varying ways, the specially selected groups for each country given in the individual dose assessment in section 8.4.

The accuracy of such estimates of intake for different population groups is dependent on the accuracy or representativeness of the information on consumption and contamination of foodstuffs provided by participating countries (cf. also discussions in chapter 5). Furthermore, the approach does not intrinsically consider the proportions of the foodstuffs produced locally or imported from outside the study area, but does allocate different foodstuffs as either imported or local. Additionally, the respondents to dietary surveys may have a tendency to overestimate their consumption of some foodstuffs and there is also likely to be considerable variation within the groups surveyed. Nevertheless, this analysis provides an indication of which products are most important in determining ^{137}Cs intake and identifies the foodstuffs for which the collection of spatial data would be most appropriate, such as reindeer meat, lamb, milk, mushrooms, berries and freshwater fish.

Using the values for radiocaesium activity concentration in dietary components provided in section 8.4, the relative importance of different groups of dietary components to ^{137}Cs intake in various Arctic countries has been plotted for the most recent five-year data series available (1990-94), and is shown for both the average (Figure 8·69) and specially selected populations (Figure 8·70). These figures have necessarily been derived from data (which for some foodstuffs/countries is rather limited) supplied by each country on the basis of measurements from the Arctic area.

For the average populations (with the notable exception of Canada, where intake is dominated by the consumption of caribou meat), a range of different products contribute to the total ^{137}Cs intake. The lowest estimated ^{137}Cs intake by an average population is for Greenland, where a large proportion of the diet is derived from marine ecosystems, the products of which (fish, seal, whale) generally contain low ^{137}Cs activity concentrations. The highest estimate of ^{137}Cs intake by the average population is for Canada. Also, intake in Fennoscandia and western Russia is enhanced by recent additional contamination from Chernobyl.

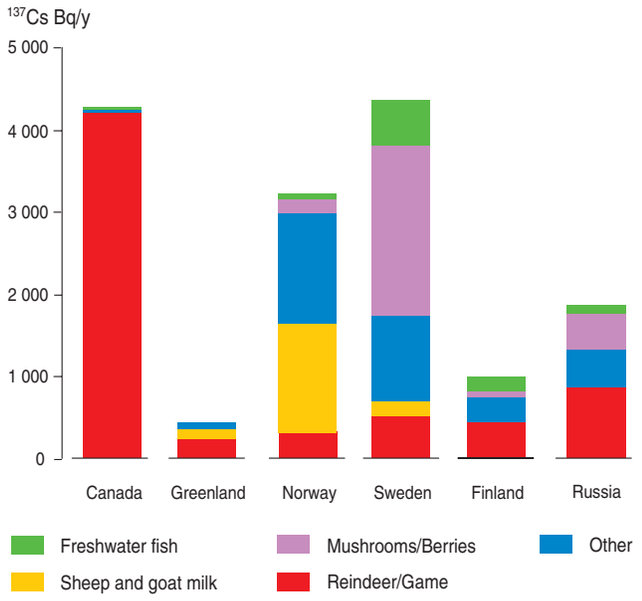

Figure 8·69. The relative contribution to the ^{137}Cs content of the diet of the 'average' population of various Arctic areas for the period 1990 to 1994.

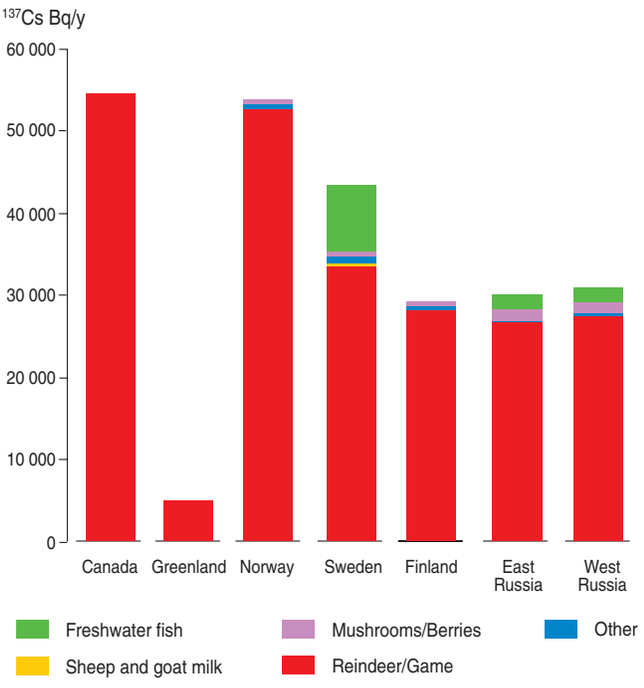

Figure 8·70. The relative contribution to the ^{137}Cs content of the diet of the 'selected' population of various Arctic areas for the period 1990 to 1994.

In Figures 8·69 and 8·70 the effects of regional dietary preferences are clear. For example, goat cheese is an important source of dietary radiocaesium in Norway only; mushrooms and berries appear to be a more important source of ^{137}Cs intake in Fennoscandia and Russia than elsewhere. However, information is sparse on exploitation of natural food products in other areas.

In contrast to the average populations, ^{137}Cs intake by the specially selected groups is consistently dominated by reindeer/caribou consumption throughout the Arctic area, although in Sweden consumption of freshwater fish also contributes significantly. The highest annual estimates of intake for 1990-1994 (ca. 50 kBq) were obtained for Canada and Norway, with the smallest for Greenland (ca. 5 kBq). However, the results must be interpreted with some caution as the Greenland selected populations is hypothetical and other selected groups are not representative of all indigenous people in that country (e.g., Canada). Where hypothetical or critical groups have been used to calculate intakes by 'the indigenous population' as in Canada and Greenland, calculated intakes will not be applicable to the indigenous population as a whole. For instance, estimates of ^{137}Cs intake by the Canadian selected groups are based on consumption of foodstuffs by an inland Inuit group who subsist almost entirely on caribou and eat no marine fish. This is not representative of all Canadian Inuit, many of whom live closer to the sea and consume larger quantities of seafood, or of the Indian or Metis indigenous peoples. Indeed, the Canadian group could be considered representative of a critical population for the whole Arctic, namely people who consume extremely large quantities of reindeer meat. It is likely that such groups could be found in other areas of the Arctic, such as in Fennoscandia or Russia. Under such circumstances, the intake by hypothetical individuals consuming reindeer meat at similar rates to the Canadian group during the 1990-94 period have been calculated (using the ^{137}Cs activity concentrations in reindeer meat for each country for 1980-84 in the Annex tables) and are shown in Table 8·47.

Table 8·47. Annual intake of ^{137}Cs by individuals consuming large quantities of reindeer/caribou meat.

Country/ region	Estimated annual ^{137}Cs intake (kBq) from consumption of reindeer/caribou meat at a rate of 1 kg/d	
	1980-1984 (pre-Chernobyl)	1990-1994 (post-Chernobyl)
Arctic Canada	164	55
Arctic Sweden	88	290
Arctic Norway	157	160
Arctic Finland	164	144
Arctic Russia (West)	182	108
Arctic Russia (East)	95	104

Table 8·47 demonstrates the importance of food intake from such semi-natural products. A comparison of pre-Chernobyl estimates of intake given in Table 8·47 with the ^{137}Cs deposition in each country (Table 8·3) shows some discrepancies. These probably arise from variation in location and representativeness of the ^{137}Cs data for reindeer. In particular, few data are available for ^{137}Cs in reindeer meat in Canada, relative to its area.

8.7.2. Spatial distribution of Arctic communities

The likelihood and extent to which different Arctic populations are affected by radionuclide deposition will be influenced by the size, ethnic and spatial compositions of communities. The selected populations in section 8.4 (comprising mostly indigenous people) have higher dietary intakes of radiocaesium than average populations. Therefore, the larger the proportion of indigenous people in an area, the higher will be the net transfer of radiocaesium. However, caution must be taken in some areas where indigenous people form the bulk of the population and also, therefore, dominate the average population.

In some of the areas being considered, assessments of typical consumption and intake rates may be complicated by the diversity of indigenous groups, with different dietary preferences. For instance, Arctic Russia is populated by many discrete groups, from Saami in the west to Chukchi and Yupiks in the extreme east. Similarly, Arctic Canada and Alaska are populated by various different indigenous groups, including Inuit, Aleut, Athabascans, Indians and Metis with similar variations in culture and habits.

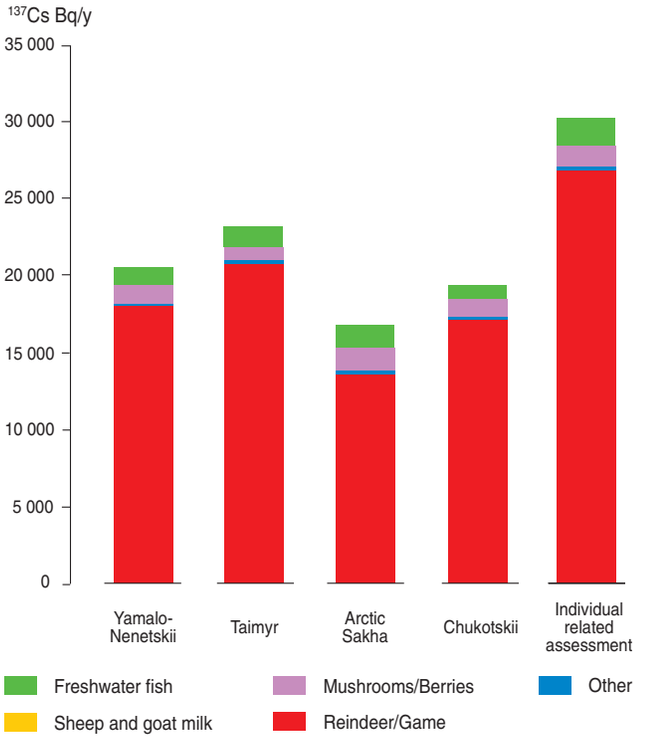

Figure 8·71. Estimated source of dietary intake of ^{137}Cs by various indigenous population groups in different regions of Arctic Russia in 1993, and the AMAP region as a whole.

From Figure 8·71, it is clear that, throughout Asian Arctic Russia, intake of ^{137}Cs by all groups is dominated by reindeer consumption, although it is also possible to identify regional differences in consumption of freshwater fish and mushrooms and berries that have an influence on total ^{137}Cs intake. The above approach used varying values for deposition to each indigenous autonomous area, calculated using the GIS-based method described in section 8.3. This has been combined with a mean value for aggregated transfer to reindeer meat of 0.3 kg/m^2 across eastern Arctic Russia. This mean value has been used because measurements of reindeer in eastern Russia were too few to derive representative spatial or temporal values relevant to each indigenous group. For additional comparison, the rates of consumption used in the dose assessment (section 8.4) to calculate the intake of the selected population have also been calculated.

The distribution of indigenous peoples and their contribution to the populations of different Arctic countries as a whole are shown in Table 8·48. These values are derived from chapter 5 of the AMAP assessment.

There is considerable variation in the proportion of indigenous people in each country's Arctic area, with the highest proportions in Greenland and Canada. Such differences will obviously affect the interpretation of the 'average group' estimates given previously.

Table 8·48. Populations of different AMAP Arctic regions.

Country/region	Total	Indigenous	% Indigenous
Alaska	481000	73200	15.2
Arctic Canada	93000	47400	51
Greenland	55400	48000	86.6
Iceland	255700	–	0
Arctic Finland	200700	4000	2
Arctic Norway	379500	37400	9.9
Arctic Russia	1999700	67200	3.4
Arctic Sweden	263700	6100	2.3
Faeroe Islands	43700	–	0
Svalbard	3200	–	0
Total	3775600	283300	7.5

8.7.3. Spatial differences in transfer through pathways

The rate of transfer of radionuclides through major food-chain pathways can vary considerably within relatively small areas due to differences in factors such as the form of contamination, soil type and nature/intensity of land use. Additionally, the relative importance of different pathways varies among radionuclides. By considering important pathways and the factors that influence transfer through them, the layers of information necessary for spatial modeling can be identified and developed.

In most terrestrial ecosystems, the major transport pathway for dietary intake will be that from soil → plant → animal → humans, for instance via milk and meat. The extent of contamination will be influenced by the soil type. In Arctic ecosystems, however, the pathway lichen → reindeer → humans is of great importance. As lichen have no rooting systems, this pathway is not influenced by soil type. Arctic lichen are contaminated via direct aerial contamination after which the deposited radionuclides are retained by the lichen and made available for subsequent translocation to fresh growth. Both ^{90}Sr and ^{137}Cs cycle readily within lichen. Hence, comparatively high levels of radioactive contamination are maintained over long periods. Ultimately, however, ^{90}Sr is washed out more rapidly than ^{137}Cs (Nevstrueva *et al.* 1967). During the winter, reindeer may graze almost exclusively on lichen, especially the species *Cladonia alpestris* and *Cladonia rangiferina*, and transfer can then be expected to be independent of soil type. In contrast, during the summer, reindeer graze on herbaceous vegetation, at which time their ^{137}Cs content will be derived from soil via the pathway soil → plant → reindeer. The ^{137}Cs content of reindeer, therefore, follows a seasonal cycle with peak activity concentrations in the winter months.

As reindeer are traditionally culled in winter, less spatial variation in the rate of transfer of radiocaesium might be expected than for other food products. Generally, in most Arctic areas, this appears to be the case. However, there are exceptions; Icelandic reindeer consume herbaceous vegetation and a species of lichen (*Cetaria islandica*) which is less efficient at trapping and retaining radiocaesium than those species preferred by reindeer in most other regions (Palsson *et al.* 1994). Consequently, transfer rates to Icelandic reindeer are much lower than those observed elsewhere and there is little seasonal variation in the level of ^{137}Cs contamination. Similar observations have been made in the Yakut region of Siberia. Transfer may also be influenced by such factors as grazing pressure; intensive grazing of lichen could reduce the pool of radiocaesium available for ingestion by reindeer.

For other foodstuffs, differences in soil type can profoundly influence transfer of some radionuclides. Spatial variation in soil type will be reflected in the rate of transfer to the products derived from the soils. Generally, radiocaesium is fixed strongly by mineral soils but can remain mobile in soils with low clay mineral content, such as highly organic and sandy soils. In contrast, radiostrontium is often less mobile in organic soils. Generally, improved soils usually fix ^{137}Cs and the transfer to arable crops and animal products is relatively low, whereas transfer to milk and meat from animals grazing on unimproved pasture over poorer quality soils is higher. The spatial distribution of different soil types will, therefore, be an important factor in determining vulnerability.

In most Arctic soils, cold wet conditions prevail due to restriction of drainage by permafrost and/or high precipitation. Biological activity is reduced, resulting in development of poor quality soils. Where conditions favour plant growth,

soils with organic upper horizons may develop, varying from deep peat to shallow rankers and podzols. Where vegetation cover is more limited, poor quality unconsolidated gravel soils may develop on parent materials such as weathered regolith, outwash and glacial deposits. Many of these poor quality soils have a low fixation capacity for radiocaesium. In more organic soils, much of the ^{137}Cs remains in the upper soil horizon and is available for uptake by plants, whereas in coarser soils, ^{137}Cs is more readily leached down through the soil profile, although in some, layers of permafrost may act as barriers to its mobility in the soil profile.

In the most temperate Arctic regions, however, conditions encourage sufficient mineralization and humification to produce agricultural soils. These soils are not distributed evenly through the area and are generally associated with mild coastal regions. Production of some foodstuffs is, therefore, highly skewed toward a comparatively small and atypical part of the region as a whole.

Climate and soil type strongly influence land use. In areas of the Arctic with better quality soils, land use involves cultivation of crops and fodders, or grazing by cattle or sheep. As such areas are relatively small and well characterized, it is theoretically straightforward to derive rates of transfer for a given product and soil type. Areas with poorer quality soils support unimproved pasture or forest, and are examples of semi-natural ecosystems. The distribution of different semi-natural ecosystems, such as forests, tundra and mountain pastures, will again be important in modeling spatial variation in transfer and radionuclide flux. Different semi-natural ecosystems are utilized in varying ways: for summer grazing of sheep, goats and cattle, for herding reindeer, as sources of fungi and berries (e.g., blueberry, lingonberry and cloudberry) and for hunting game. Some of these products are distributed in a variety of different types of semi-natural ecosystem, whereas others such as moose, are largely confined to forests. Analysis of the variation in importance of transfer pathways can be refined by knowledge of the spatial distribution of different ecosystems and the rate of production and exploitation of products from them. However, quantifying transfer in semi-natural ecosystems is complicated by their size and diversity, particularly the intrinsic variability in deposition, soil and vegetation type between and within areas. As transfer of radiocaesium to different vegetation species varies, factors such as diet composition, which may itself be influenced by grazing pressure, will affect the rate of aggregated transfer to grazing animals.

Transfer through marine and freshwater ecosystems is dominated by the pathway water→fish/mollusk/crustacean →humans, although in some Arctic areas population groups also eat large quantities of marine mammals (seal and whale). Transfer in marine systems is generally low because of the magnitude of dilution and because the bulk of the activity of many radionuclides is sorbed to bottom sediments. Hence, contamination of marine organisms is generally low, although it may be expected that benthic species would contain more activity than pelagic species. Spatial variation in transfer to marine products is not well defined across the Arctic Ocean areas. Consequently this initial spatial analysis will not consider variation in transfer through marine pathways in detail.

Potentially, freshwater ecosystems are important contributors to radiocaesium intake (although less so than terrestrial ecosystems), and will exhibit spatial variation in transfer. Deposition to freshwater catchments could vary over quite small distances as a result of orographic influences. Over long periods, catchments act as a source for release of radionuclides into streams and lakes by runoff. The extent of radionuclide release into water bodies will be dependent both on catchment size and soil type. In particular, catchments dominated by organic soils can act as a long-term source of radiocaesium release into freshwater ecosystems (Smith *et al.* 1995b). Furthermore, although much of the radioactive contaminants will sorb to sediments, partitioning between water and sediments will be affected by local physical-chemical variables such as pH, and ionic concentration. There is, therefore, considerable potential for spatial variation in transfer in the food chain water→fish→humans among different freshwater systems.

8.7.4. Changes with time

Consideration of spatial vulnerability requires an understanding of how the relative importance of different transfer pathways varies with time. This may help determine whether the vulnerability of an area is short- or long-term.

Firstly, this will be due to differences in the physical half-lives of different radionuclides. For example, immediately after a release, ^{131}I may present the greatest radiological concern because of its high mobility and radiotoxicity. However, its short physical half-life renders it a short-term radiological problem. In the longer term, transfer of radiocaesium and ^{90}Sr will become more significant.

Secondly, long-term vulnerability will be influenced by variation in the effective ecological half-life ($T_{1/2\ \text{eff-eco}}$) of ^{137}Cs in food products, potentially altering the relative importance of different transfer pathways. Consequently, as time elapses after deposition, foodstuffs with longer $T_{1/2\ \text{eff-eco}}$ will proportionately assume greater importance for ^{137}Cs intake. The $T_{1/2\ \text{eff-eco}}$ of ^{137}Cs in most food products (other than reindeer) is related to the rate of depletion of the available pool, either due to fixation by the soil or to removal by runoff or leaching down the soil profile. Therefore, the contribution to the total ^{137}Cs intake from those products with the longest $T_{1/2\ \text{eff-eco}}$, namely those such as fungi and lamb meat produced on pasture with organic soils, can become proportionately greater with time. For reindeer, grazing lichen, the $T_{1/2\ \text{eff-eco}}$ can be further influenced by the depletion of ^{137}Cs in the available pool (lichen) by grazing, as well as losses through leaching.

Similarly, vulnerability will be influenced by variation in the throughput of activity in environmental compartments. For example, in freshwater ecosystems, throughput of ^{137}Cs along the pathway catchment→lake→river could vary considerably depending on such factors as catchment size, lake size and local physico-chemical characteristics. This will affect the effective ecological half life of ^{137}Cs in the lake, and therefore in the food chain.

8.7.5. Transfer coefficients and relationships

8.7.5.1. UNSCEAR transfer coefficients

The transfer coefficient from deposition to diet applied by UNSCEAR was defined in section 8.2. Using data series supplied by participating countries, time-integrated transfer coefficients to various important Arctic products have been derived. Such parameters can only be estimated when sufficient time-series data are available for a given product. Additionally, use of the coefficients on a wider scale requires the assumption that the raw data are representative of the wider area. Estimation of this transfer parameter is thus useful for broad comparisons among different areas, but of limited use when data are scarce. Therefore, the areas and products included in the following analysis represent those areas for which adequate data were available. Analysis of the data has been carried out as shown in the boxed example next page.

Example: Calculation of the integrated transfer coefficient for ^{137}Cs to Arctic Finnish lichen (see Table 8·49).

From Figure 8·28 the mean values for 5-year intervals from 1960 to 1994 are obtained. These values are summed and multiplied by 5 (to account for the 5-year means) to obtain the time integral 31 000 Bq/kg y for the period 1960-1994.

We get the time integral for 1951-1959 from multiplying the 5-year mean value for 1960-1964 (700 Bq ^{137}Cs/kg) in Figure 8·28 by 0.5 × 9 = 4.5, as we assume that the ^{137}Cs concentrations in lichen increased linearly from zero in 1951 to 700 Bq ^{137}Cs/kg in 1959. The time integral becomes 3100 Bq/kg y.

The time integral for 1995 and onwards is calculated from the 5-year mean value for 1990-1994 (700 Bq ^{137}Cs/kg) assuming an effective half-live of ^{137}Cs in Finnish Arctic lichen of 10 years from 1995 and onwards, i.e., 700 × 10/ln2 = 10 000 Bq/kg y.

The three time integrals obtained above are now added:

3100 + 31 000 + 10 000 = 44 100 Bq ^{137}Cs/kg y, and we have the total infinite time integral of ^{137}Cs in Arctic Finnish lichen, arising from a total (not decay-corrected) deposition of 2.5 kBq ^{137}Cs/m^2 since 1950. The so-called integrated deposition density of 2.5 kBq ^{137}Cs/m^2 was the sum of the annual depositions of ^{137}Cs in Arctic Finland as shown in Figure 8·20, which cover the period 1960-1994. For the years prior to 1960, the deposition of ^{137}Cs in Arctic Finland was assumed to be proportional to the deposition measured of ^{90}Sr in New York for these years (see also the text in 8.3.2.1).

The integrated transfer coefficient is finally calculated by dividing the infinite time integral (44 100) by the integrated deposition density (2.5) and we get 18 000 Bq/kg y per kBq/m^2 as shown in Table 8·49.

Table 8·49. Integrated transfer coefficients (Bq/kg y per kBq/m^2) for ^{137}Cs to Arctic lichen.

Area	Integrated deposition, kBq/m^2	Time-integrated concentration, Bq/kg y			Integrated transfer coefficient
		1950-59	1960-94	1995-	
Arctic Finland	2.5	3100	31000	10000	18000
Greenland	4.3	4800	25000	1700	7000
Arctic Russia	3.1	6000	18500	3000	9000

Table 8·50. Integrated transfer coefficients (Bq/kg y per kBq/m^2) for ^{90}Sr to Arctic lichen.

Area	Integrated deposition, kBq/m^2	Time-integrated concentration, Bq/kg y			Integrated transfer coefficient
		1950-59	1960-94	1995-	
Greenland	2.7	900	5200	300	2000
Arctic Russia	1.7	1100	6300	1000	5000

Table 8·51. Integrated transfer coefficients (Bq/kg y per kBq/m^2) for ^{137}Cs to Arctic reindeer meat.

Area	Integrated deposition, kBq/m^2	Time-integrated concentration, Bq/kg y			Integrated transfer coefficient
		1950-59	1960-94	1995-	
Arctic Finland	2.5	5700	34000	5500	18000
Greenland	4.3	1500	6500	600	2000
Arctic Norway	4.4	4200	31000	9500	10000
Arctic Russia	3.1	4300	21000	3300	9000

All the following calculations of time-integrated transfer coefficients are derived from deposition data published by UNSCEAR. This approach to quantifying transfer has some limitations. A more localized problem, especially in parts of the European Arctic, is the patchy nature of the Chernobyl deposit which adds uncertainty to estimates of integrated deposition. For instance, the value of 2.5 kBq/m^2 used for Finland has been derived from measurements of activity deposited at specific sampling sites that may not be representative of the entire area. Mobile-gamma surveys conducted in Arctic Finland after the Chernobyl accident suggest a higher value would be more appropriate (Arvela *et al.* 1990) which would give lower transfer coefficients. Whilst mobile gamma measurements are not supported by soil analyses from various parts of Arctic Finland, they do, however, give similar results to the sampling of wet and dry deposition as values are corrected for decay.

Tables 8·49 and 8·50 show comparisons between time-integrated transfer of ^{137}Cs and ^{90}Sr to lichen in Greenland, Arctic Russia, and Arctic Finland (^{137}Cs only). The transfer coefficients of ^{90}Sr are lower to lichen in Greenland than in Arctic Russia, and of ^{137}Cs, lower to lichen in Greenland than in Arctic Russia and Arctic Finland. Furthermore, for Greenland and Russia, the transfer coefficient of ^{137}Cs is almost 2-3 times higher than that of ^{90}Sr, which is consistent with the known ability of lichen to intercept and retain radiocaesium for longer than radiostrontium.

Table 8·51 shows transfer coefficients for ^{137}Cs to reindeer. Initial comparison of the transfer coefficients for ^{137}Cs to reindeer meat and lichen reveals that transfer to reindeer in Arctic Finland and Russia is proportional to that for lichen but transfer to reindeer is, relatively, lower in Greenland. This is probably because reindeer are slaughtered in Greenland during the summer, when activity concentrations in their meat will reflect their summer diet of green vegetation. A lower transfer coefficient to Greenland reindeer might, therefore, be expected, compared with reindeer from Finland and Norway, where animals are slaughtered in winter.

In Russia, some reindeer are slaughtered all the year round, which could potentially lead, on average, to lower transfer. However, the raw data for Russian reindeer were collected from a range of areas, with inadequate frequency of sampling to establish a time series for different indigenous regions. The data were, therefore, too scattered, spatially and temporally, to explore the seasonal factors influencing the transfer coefficient.

This highlights a more general problem with representativeness of lichen and reindeer samples. As deposition of ^{137}Cs from nuclear weapons tests was distributed fairly homogeneously, lichen samples collected locally at Inari in Finnish Lapland were representative of Finnish Lapland as a whole, i.e., they were comparable with the reindeer samples. However, after Chernobyl, Inari received rather more (three times higher) fallout than most of the surrounding area. Hence, after 1986, lichen from Inari would no longer be directly comparable to Finnish reindeer meat samples, which integrate deposition from a large area.

A further variable influencing the time-integrated transfer coefficient is the use of countermeasures. In many parts of Norway and Sweden that received deposition from Chernobyl, countermeasures were applied, including use of intraruminal boli and salt licks. This could explain why ^{137}Cs transfer to reindeer meat appears lower in Norway than in Finland, where such countermeasures were not introduced. Similarly, countermeasures have been observed to reduce the wholebody radiocaesium content of reindeer herders in Sweden relative to estimates based on dietary information and measurements of integrated deposition. The greatest discrepancies were observed where Chernobyl fallout was highest, with corresponding countermeasure applied (R. Bergman, Sweden, pers. comm). Comparison of time-integrated trans-

Table 8·52. Integrated transfer coefficients (Bq/m³ y per kBq/m²) for ^{90}Sr to Arctic freshwater.

Area	Integrated deposition, kBq/m²	Time-integrated concentration, Bq/kg y			Integrated transfer coefficient
		1950-59	1960-94	1995-	
Arctic Finland	1.72	130	580	120	500
Greenland	2.7	500	1400	90	700

Table 8·53. Integrated transfer coefficients (Bq/kg y per kBq/m²) for ^{137}Cs to human body for Arctic selected groups.

Area	Integrated deposition, kBq/m²	Time-integrated concentration, Bq/kg y			Integrated transfer coefficient
		1950-59	1960-94	1995-	
Arctic Finland	2.5	1400	6200	1100	3500
Arctic Norway	4.4	1100	7700	3700	3000
Arctic Russia	3.1	3000	17400	3000	7500

fer coefficients, therefore, requires knowledge of agricultural practices that may result in spatial variations within the data.

The integrated transfer coefficients for ^{90}Sr to freshwater (Table 8·52) were surprisingly similar, even though the samples consisted of river water from Finland and Russia, but of drinking water from Greenland. The integrated transfer coefficient of ^{137}Cs to Finnish river water was estimated to be 900 Bq ^{137}Cs/m³ y per kBq ^{137}Cs/m² which is somewhat higher than that observed for ^{90}Sr in Finnish rivers.

As with terrestrial systems, freshwater ecosystems are subject to uncertainties with regard to estimation of integrated deposition and net influx and efflux of radionuclides. Effects of catchment size and soil type, lake size and distribution of radionuclides between sediments and solution can all contribute significantly to spatial variability in transfer.

Transfer to the human body will be influenced by the rate of transfer in all the ecosystems contributing to the human diet. The integrated transfer coefficients of ^{137}Cs for the wholebody of selected groups (reindeer herders) are shown in Table 8·53. The integrated transfer coefficient for the Russian group is twice that for the Finnish and Norwegian groups.

Because human wholebody content is normally derived from a range of dietary sources, comparison of the transfer coefficient to reindeer with that to humans would not normally be valid. However, for the selected populations, so much of their ^{137}Cs intake is derived from consumption of reindeer that cautious comparisons may be drawn. Compared with the integrated transfer coefficient for reindeer meat (Table 8·51), that for the human body is significantly lower in Norway and Finland, but for Russia the difference is smaller. These differences could be due to local dietary preferences such as differences in rate of consumption of foodstuffs other than reindeer, or for some foodstuffs (e.g., fungi, berries), spatial variation in consumption rates of different species, or even in the method of preparation. Finnish Saami that consume fungi apparently select the genus *Lactarius* specifically, and prepare them by parboiling, which reduces the ^{137}Cs content resulting in lower intake via fungi than might otherwise have been expected. Furthermore, the extent of utilisation of natural foods by indigenous peoples is, in many cases, not well known.

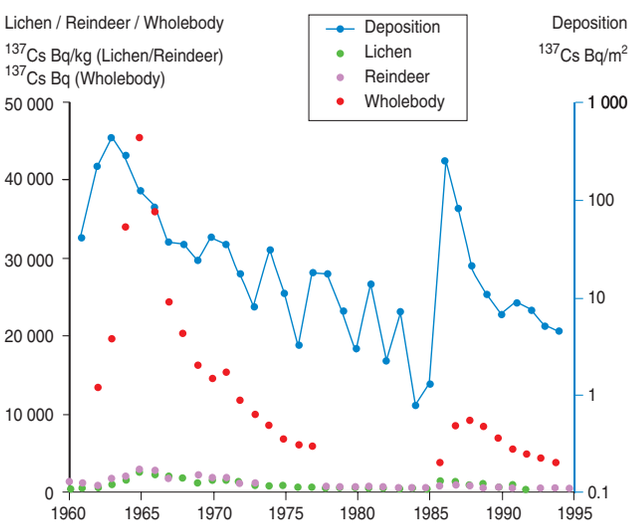

Figure 8·72. Changes with time in ^{137}Cs contamination in the food chain lichen → reindeer → humans. The figure, showing the relationships between ^{137}Cs in deposition, lichen, reindeer meat and the human body for northern Finland, is an example of real rather than calculated transfer.

Figure 8·72 summarizes the relationship between ^{137}Cs activity concentrations in estimated Finnish deposition and measurements in lichen, reindeer meat and the human body.

8.7.5.2. Spatial and temporal variations in transfer to Arctic food products using aggregated transfer coefficients

The analysis carried out in section 8.7.1 and 8.7.5.1 used data generalized to represent the whole Arctic area of the countries involved (although in section 8.7.1, eastern and western Russia were considered separately for the selected groups). However, ^{137}Cs activity concentrations in foodstuffs will vary spatially within, as well as among, countries, particularly depending on such factors as local deposition rates, soil and vegetation type, presence of forests and dietary preferences of domestic and game animals. Rates of production or harvesting will also vary spatially, depending on ecological factors such as the availability of forage for animals. In this section, such variation is considered in order to predict how both the ^{137}Cs activity concentrations and total output in reindeer and milk may vary spatially both within and among countries in the event of deposition from a future radionuclide release.

Where data are available, spatial information on such variables as food production, ^{137}Cs deposition and ^{137}Cs activity concentrations in food products have been incorporated into a Geographical Information System (GIS). These layers of information have been combined to derive aggregated transfer coefficients (T_{ag} values) for the spatial unit being modeled.

Spatial analysis has, therefore, been conducted with the aim of assessing the vulnerability of different regions or products to ^{137}Cs contamination. Analysis of vulnerability can be undertaken using two approaches:

- ***Specific vulnerability*** is calculated using the specific activity of a radionuclide in a product at a known time from a specified level of deposition. For large scale analysis, this requires prior knowledge of aggregated transfer and effective half-lives for different soil types and products, which are then applied generically. As aggregated transfer is time-dependent, specific vulnerability is described as a year-specific T_{ag}. This allows simple comparison among areas of different soil type or land use and the development of optimal and cost-effective countermeasure strategies. Alternatively, vulnerability could be demonstrated through the level of deposition needed for a product to exceed a given activity concentration. This is a similar approach to that used previously in studies of soil acidification, to assess critical loads for different soils.

- ***Flux vulnerability*** takes spatial variation in productivity into account by estimating the total output of radionuclides from an area in each food product. Total radionuclide flux is calculated by applying a knowledge of differences in food production and aggregated transfer. The flux vulnerability for an area is, therefore, a product of the time-dependent aggregated transfer (specific vulnerability) and annual production of the appropriate foodstuffs, divided by the area of the spatial unit being considered.

Example: To calculate the aggregated transfer coefficient (T_{ag}) for reindeer meat.

Collate information on activity concentrations in reindeer meat from a defined area or ecosystem. For reindeer information on season is also important.

Collate information on ground deposition (from measurements of activity in soil per unit area) in the area where the reindeer graze. (Alternatively, transfer to reindeer can be derived from measurement of lichen samples, where lichen cover is sufficient to intercept deposited activity. For other terrestrial foodstuffs only soil measurements are appropriate).

The aggregated transfer coefficient (T_{ag}) is calculated using the following expression:

$$T_{ag} = \frac{\text{Activity concentration in reindeer meat (Bq/kg)}}{\text{Activity of deposit (soil or lichen) per unit area (Bq/m}^2\text{)}}$$

with units of m^2/kg.

Aggregated transfer coefficients, defined, for ^{137}Cs, as the ^{137}Cs activity concentration in the food product (Bq/kg fresh weight) divided by the current ground deposition (Bq/m^2), were described in section 8.2. An example calculation of the T_{ag} for reindeer meat in Finland, Russia and Norway is shown in the boxed example above. In this initial spatial assessment, T_{ag}s have been used for quantifying and modeling transfer to terrestrial products only. It is acknowledged that variability in activity concentration and transfer between individual lakes is considerable, and therefore T_{ag} values in freshwater ecosystems are subject to considerable uncertainty.

T_{ag} values can be combined with appropriate regional estimates of effective ecological half life to give spatial information on short- and long-term vulnerability. The scale of the analysis presented here has been chosen according to the availability of information at the AMAP radioactivity data center.

A weakness of this approach is that predictions using T_{ag}s are not applicable during the initial phase of direct foliar contamination and rapid changes in radionuclide availability in the soil following the release (Howard *et al.* 1996). This is not a problem with predictions for the food chain lichen → reindeer → humans, but may be for the soil → plant → animal → humans pathway, when the duration and importance of foliar interception would vary according to the season of the release and the extent of outdoor production of agricultural products. Consequently, T_{ag} values would be most likely to underestimate the flux in the first few weeks after deposition.

Prediction of spatial differences in radionuclide flux requires adequate data, which are, as yet, not always available. In this initial assessment, two approaches are adopted to demonstrate the wider applicability of this analysis: 1) collation of detailed spatial information on radiocaesium flux in reindeer and milk production throughout the Arctic, and 2) detailed analysis of spatial differences in radiocaesium flux in one country, Norway, at the sub-national scale, showing how vulnerability varies spatially and temporally within a relatively small Arctic area.

8.7.5.2.1. Spatial variation in total production

The rate of production of foodstuffs, or extent of exploitation via hunting or fishing, varies considerably both between and within different Arctic areas of each country. Therefore, radiocaesium flux will vary spatially, both through individual products and as a whole. An initial attempt has been made at collating production and harvesting statistics for the major foodstuffs identified in section 8.4 for each country, and is shown in Table 8·54. Much of the production information has been collated from national statistics gathered by GRID-Arendal, but where information was not readily available, supplementary information was provided by national experts.

From Table 8·54, it is apparent that information on production of some foodstuffs is more readily available than others. Commercial production of the agricultural and fishery industries is reasonably well documented in most areas, whereas, by comparison, exploitation of natural foodstuffs such as game, berries, fungi and freshwater fish is poorly documented. Additionally, production of such foodstuffs as berries and fungi can be extremely variable, depending on the climate during the growth season.

Reindeer Production

Analysis of reindeer production required differentiation between herding of domestic reindeer, which is the practise in most of Arctic Fennoscandia, and exploitation of wild or feral reindeer herds, which is more typical of parts of Arctic Russia (e.g., Taimyr Peninsula), Canada and Alaska. This is because the rate of cull is different, generally being higher in herded stocks, and therefore varies spatially, according to the dominant mode of production.

For semi-domestic reindeer production, information was obtained from national statistics and national experts. Information on the exploitation of wild reindeer, which is an important activity in Alaska, Canada and Russia and to a lesser extent in Greenland and Iceland, are less comprehensive and subject to greater uncertainty. For this assessment, information was obtained on the worldwide distribution of

Table 8·54. Annual production and harvesting of foodstuffs in Arctic countries.

Product	Alaska	Russia	Finland	Sweden	Norway	Greenland	Iceland	Canada	Total
Milk, L	5.00×10^5	1.34×10^8	9.88×10^7	1.88×10^8	1.82×10^8	–	1.03×10^8	1.37×10^3	7.06×10^8
Goat Milk, L	–	–	–	4.82×10^6	1.03×10^7	–	–	2.68×10^2	1.51×10^7
Pork, kg	2.20×10^4	1.98×10^7	6.40×10^5	1.78×10^6	1.51×10^6	–	3.21×10^6	4.54×10^2	2.70×10^7
Beef, kg	1.86×10^4	1.98×10^7	3.38×10^6	2.10×10^6	8.52×10^6	–	3.06×10^6	1.25×10^4	3.69×10^7
Lamb, kg	9.11×10^2	–	8.00×10^4	2.59×10^5	1.80×10^6	2.54×10^4	8.80×10^6	–	1.10×10^7
Potato, kg	6.40×10^6	8.53×10^7	4.40×10^6	3.61×10^7	4.72×10^6	–	1.11×10^7	9.81×10^4	1.48×10^8
Fruit/vegetables, kg	–	–	–	3.15×10^5	2.26×10^6	–	–	–	2.57×10^6
Reindeer, kg	9.08×10^5	1.89×10^7	3.04×10^6	2.20×10^6	1.59×10^6	2.06×10^5	8.99×10^3	1.33×10^6	2.82×10^7
Moose, kg	–	–	6.45×10^6	2.10×10^6	4.07×10^5	–	–	–	8.96×10^6
Fungi, kg	–	–	1.05×10^7	1.60×10^6	–	–	–	–	1.21×10^7
Berries, kg	2.38×10^4	–	2.68×10^6	2.40×10^6	4.72×10^6	–	–	1.27×10^4	9.84×10^6
Freshwater fish, kg	–	–	8.00×10^5	1.20×10^6	5.80×10^6	4.70×10^4	3.15×10^6	–	1.10×10^7
Marine fish, kg	–	1.23×10^9	1.10×10^5	n.a.	3.79×10^8	6.50×10^7	1.56×10^9	–	3.23×10^9

Table 8·55. Estimated total reindeer production in the AMAP area.

	Wild reindeer			Semi-domestic reindeer		
Country/ region	Estimated number	Estimated hunter kill	Estimated production, kg	Estimated number	Estimated slaughter	Estimated production, kg
Alaska	597000	25000	576000	36000	14000	332000
Canada	1287000	58000	1328000	–	–	–
Finland	–	–	–	346000	132000	3040000
Greenland	18000	7000	168000	4600	1600	38000
Iceland	3000	400	9000	–	–	–
Norway	–	–	–	164000	69000	1590000
Svalbard	8800	–	–	–	–	–
Russia	887000	222000	5110000	1494000	598000	13800000
Sweden	–	–	–	229200	95000	2200000
Total	2800800	312400	7191000	2273800	909600	21000000

wild reindeer populations (Williams and Heard 1986) and used to help allocate herds into appropriate spatial units in the GIS. Updated information was incorporated where available, for Alaska (Swanson and Barker 1991) and Canada (Ferguson and Gauthier 1992). The rate of exploitation of wild reindeer stocks has been estimated largely from data published by Williams and Heard (1986). In all cases, it has been assumed that the carcass yield is broadly similar to that reported for Finnish semi-domestic reindeer of 23 kg. It is recognized that the extent to which the carcass is utilized may vary spatially, but this has not been investigated.

Based on these data and assumptions, the estimated contemporary annual production of reindeer meat throughout the Arctic is shown in Table 8·55.

Milk production

Information on milk production was collated from various sources; both national statistics and data provided by national experts. Many Arctic areas have little or no milk production and, in other areas, animals graze outdoors for only a few months of the year. Furthermore, unlike the flux through reindeer meat, which is derived from an activity concentration at slaughter, the flux through milk is continuous and subject to seasonal variation depending on what the animal is eating. Calculation of fluxes through milk production are, therefore, complicated by differences in husbandry, both spatially and temporally.

The collation of estimated annual milk production in Arctic countries, given in Table 8·54, shows that the majority of Arctic milk production occurs in the Nordic countries and Russia. Consequently, information on ^{137}Cs contamination of milk is relatively sparse for the other Arctic areas. Monitoring of milk from dairies in the Nordic countries has provided some useful time series.

8.7.5.2.2. Spatial variation in fluxes

The raw data used to collate national production (Table 8·54) incorporated into the GIS were allocated to appropriate spatial units such as regional administrative areas, to give greater detail.

Reindeer

Reindeer production is highest in Arctic Russia and Fennoscandia. Taking account of the much smaller areas involved, production is densest in Fennoscandia and western Russia.

The flux of radiocaesium into the human diet via reindeer was calculated by combining production information with appropriate aggregated transfer coefficients for individual regions. These were calculated by combining measurements of reindeer contamination at known times and locations (collated in the AMAP data center for radioactivity) with estimates of ground deposition in the same area at the corresponding time. Ground deposition from global fallout was predicted using the GIS-based method described in section 8.3. Temporal variation in T_{ag}s was calculated for each country and is shown in Figure 8·73. Generally, aggregated transfer has been decreasing since a peak in the early 1960s corresponding to the maximum input of global fallout. This is a function of the declining magnitude of the pool of radiocaesium available to the reindeer because ^{137}Cs activity concentrations in lichens will decrease due to grazing and leaching of the radiocaesium into the underlying soil. The data suggest that the highest rates of transfer have been recorded in Fennoscandia. However, it is important to note that the quantity and quality of the data used to generate these T_{ag} values varies: measurements of ^{137}Cs in reindeer meat were most comprehensive from Fennoscandia; in other areas coverage was less comprehensive; data were readily available from Russia for most years, but covered a vast area rather thinly; for Alaska, data were only available from the 1960s, and for some other areas data were sparse.

The assessment of vulnerability has been modeled for a hypothetical accident resulting in new uniform ^{137}Cs deposition of 100 kBq/m^2 across the entire Arctic. This level of contamination was typical of that experienced over large areas of the former Soviet Union following the Chernobyl accident. This has been assumed to enable comparison among different areas. Obviously after an actual accident, deposition would vary spatially.

The T_{ag} values selected to estimate flux in individual areas were the pre-Chernobyl maxima, as the evidence from Figure 8·73 suggested that transfer maxima were recorded during the peak period of deposition of global fallout. The spatial trends in net flux from reindeer meat for the first year after deposition are shown in Figure 8·74, where it is apparent that fluxes would be greatest in Fennoscandia and Russia, with the highest flux in Arctic Finland. Spatial variations in flux in the Arctic regions of each country are shown more clearly by dividing the net flux by its area to obtain the flux vulnerability as shown in Table 8·56. It must, however, be emphasized that this calculation illustrates the flux vulnerability from production during the initial year following the deposition event only. With adequate information on effective ecological half-lives it is possible to calculate and compare spatial variations in fluxes of ^{137}Cs over defined time intervals. This has been carried out for Norway in section 8.7.8.

Milk

The information on milk production has been combined with time series data compiled in the AMAP radioactivity data center from the monitoring of ^{137}Cs contamination in milk from dairies in Sweden, Finland, Norway and Russia. Time dependent T_{ag} values were calculated from predictions of ^{137}Cs ground deposition in the same way as for reindeer. Trends for Norway, Sweden, Finland and Russia are shown in Figure 8·75. T_{ag} values to milk have also decreased as a

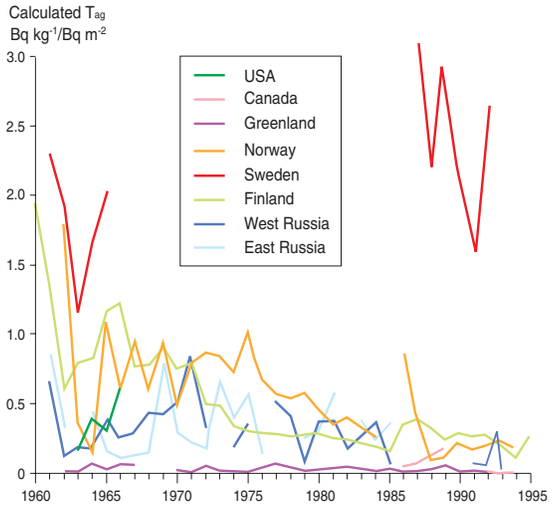

Figure 8·73. Temporal variation in calculated reindeer T_{ag} values for different Arctic regions.

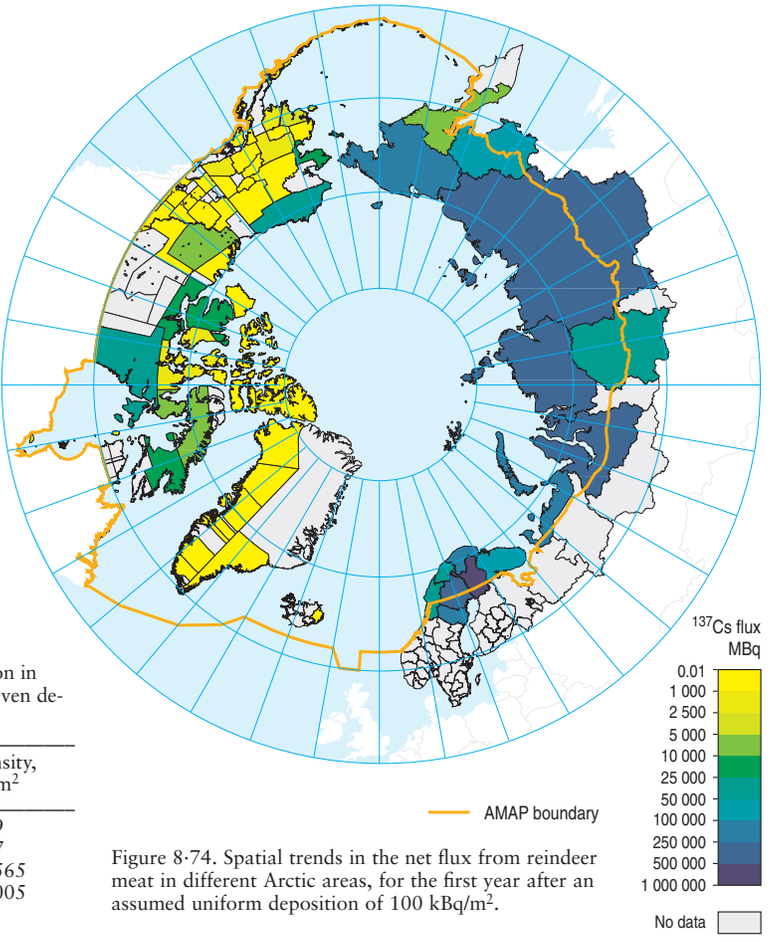

Figure 8·74. Spatial trends in the net flux from reindeer meat in different Arctic areas, for the first year after an assumed uniform deposition of 100 kBq/m^2.

Table 8·56. ^{137}Cs flux and flux density through reindeer production in Arctic regions of each country in the first year after an assumed even deposition of 100 kBq/m^2.

Arctic region	Area, km^2	Total flux, MBq	Flux density, kBq/km^2
Alaska	1410000	5.06×10^{10}	35.9
Arctic Canada	3990000	7.85×10^{10}	19.7
Greenland	2140000	1.21×10^{9}	0.565
Iceland	102000	5.40×10^{5}	0.005
Arctic Norway	167000	1.89×10^{11}	1130
Arctic Sweden	164000	4.25×10^{11}	2590
Arctic Finland	98800	6.99×10^{11}	7070
Arctic Russia (west)	411000	2.02×10^{11}	491
Arctic Russia (east)	6790000	1.39×10^{12}	205
Total	15272800	2.91×10^{12}	–

function of the decreasing pool of radiocaesium that is available for uptake by plants and ingestion by cows. In contrast to the factors influencing transfer to reindeer, the dominant mechanism in lowering the rate of transfer to milk is fixation of radiocaesium in the soil.

The relevant T_{ag} values have been used to estimate the flux of ^{137}Cs through milk in Fennoscandia and western Russia assuming a hypothetical uniform 100 kBq/m^2 deposition event. As with reindeer, the highest pre-Chernobyl T_{ag} has been applied to modeling the flux. However, it is important to remember that for milk ^{137}Cs transfer will vary with fodder type and source. Nevertheless, even in winter, much of the fodder hay is likely to be produced locally; concentrates, however, are likely to be imported.

A histogram showing the relative flux of ^{137}Cs in reindeer and milk throughout the Arctic in the first year following deposition, is shown in Figure 8·76. Where data was inadequate for calculation of accurate local T_{ag} values, a value of 0.0113, based on a mean of the data from Fennoscandia, has been applied. It is apparent that in the event of a future release the flux vulnerability of the Arctic to reindeer production is greater than that for milk production. However, there

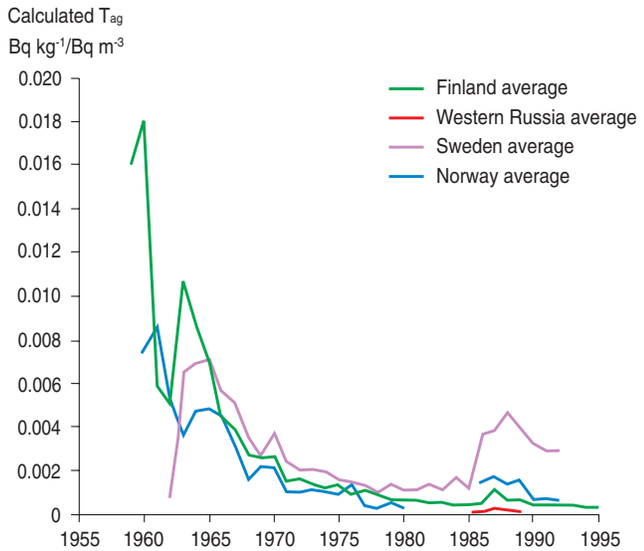

Figure 8·75. Temporal variation in cow milk T_{ag} values for Sweden, Norway and Finland.

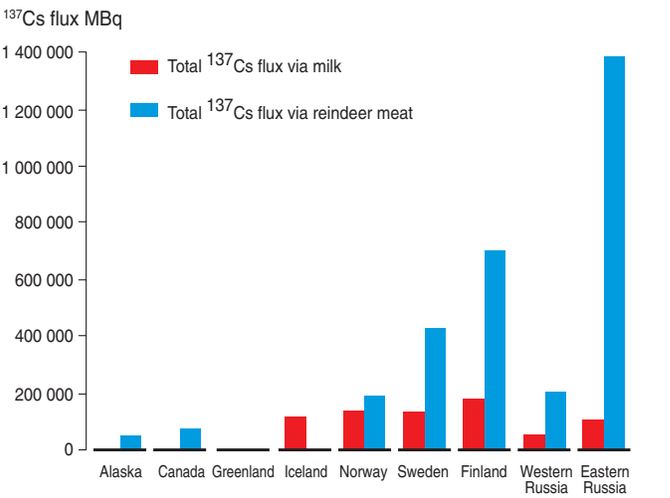

Figure 8·76. Comparison of the predicted flux of ^{137}Cs in milk and reindeer meat in different Arctic areas, for the first year after an assumed uniform deposition of 100 kBq/m^2.

are differences among countries. Whilst for most counties the reindeer flux is greater, the milk flux dominates in Iceland, whilst in Norway the milk and reindeer fluxes are similar.

Radioiodine contamination of milk

In the immediate period following a release, the radionuclide ^{131}I is of major radiological concern due to its volatility, mobility and radiotoxicity to human thyroid. The short half-life of this radionuclide (8 days) means that the primary ingestion pathway for exposure is via fresh produce, particularly the consumption of milk.

Generally, milk production in the Arctic is low compared with temperate areas. Imported fodders are often used in winter, and there is comparatively little production of sheep and goat milk (to which transfer of radioiodine is higher than that to cow milk). However, on the Kola Peninsula, close to the Kola Nuclear Power Plant which is a potential source of ^{131}I, milk is produced on both private and collective farms. This area would, therefore, have some vulnerability following an ^{131}I release. If a release occurred during the outdoor grazing season, individual doses to consumers of privately produced milk might potentially be similar to those in temperate areas. However, collective doses would be likely to be lower, as net production is low compared to temperate areas. Furthermore, as the animals spend relatively little time grazing pasture, there is reduced probability of an accident happening during that period.

Generally, therefore, for human dose, Arctic ecosystems might be expected to have lower vulnerability for ^{131}I compared to temperate regions, and the inhalation pathway may be of greater relative importance than ingestion. Locally, however, incorporating the location of milk-producing areas in relation to potential sources into spatial models could improve predictions of areas vulnerable to a release of ^{131}I in the Arctic.

8.7.7. Sensitivity to uncertainties: radiocaesium in fungi and berries

Previous dietary studies have demonstrated the potential importance of consumption of semi-natural and natural food products to radiocaesium intake. Fungi and berries are important dietary components in many areas. Unfortunately, information on intake, and contamination, of such foodstuffs is not readily available. For fungi, there are further uncertainties.

Radiocaesium transfer to different species of fungi from the same location can vary considerably. Currently, however, there are very few relevant data on transfer of radiocaesium to fungi in Arctic ecosystems.

The rate of production of fungi varies greatly from year to year, as a function of the prevailing climate. This will probably be reflected in consumption.

The exploitation of different natural foodstuffs varies spatially, according to preference and availability, and is not well documented.

In some Arctic areas, estimates of net radiocaesium flux may be especially sensitive to variation in, and lack of knowledge regarding, production, transfer to, and exploitation of semi-natural and natural foodstuffs. Unfortunately, however, information is currently too sparse to assess the extent to which spatial variation in production and exploitation of these products can influence fluxes.

8.7.8. Flux vulnerability of Arctic Norway

An initial attempt at a more comprehensive spatial analysis has been carried out for the mainland Norwegian AMAP area, considering the three counties Finnmark, Troms and Nordland (not all of the latter county is in the AMAP area) (Figure 8·77). This analysis is included as an example of the approach that could be used in other parts of the AMAP area. Most major foodstuffs with relevance to ^{137}Cs transfer have been included, but current deficiencies are noted and discussed.

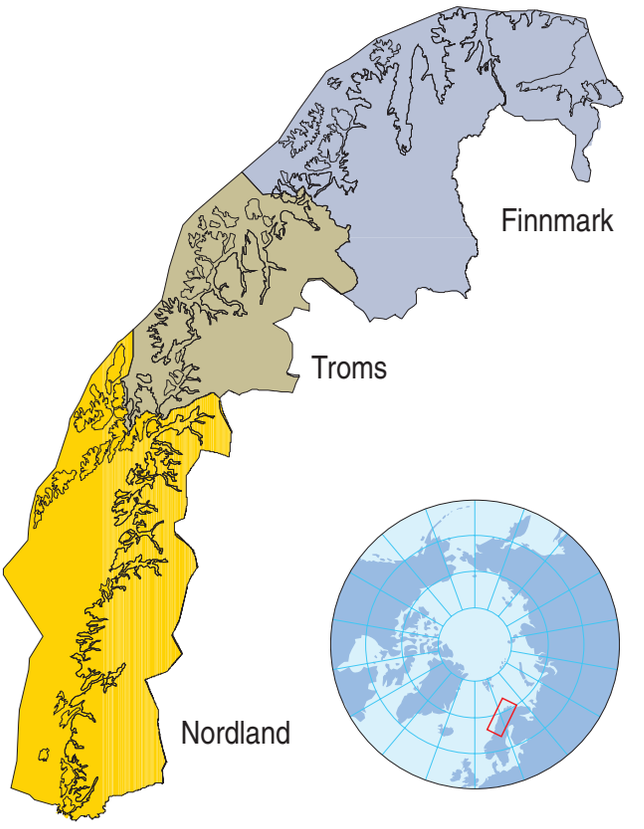

Figure 8·77. Norwegian counties considered in the flux vulnerability case study.

8.7.8.1. Production data

Compared to much of western Europe, Arctic Norway has only a small proportion of land that is actively managed as arable or pasture. Foodstuff production data (given in Table 8·54) were entered into the GIS at the smallest spatial unit for which they were available, which was at county level with the exception of cow and goat milk, moose and reindeer. The GIS was used to aggregate the total production for each county. Production of cow and goat milk was available for each municipality. The numbers of moose culled in different age and sex classes for each county were combined with average carcass weights to calculate the total production of moose meat. Similarly, for reindeer meat, the number of animals slaughtered was combined with average carcass weights for reindeer grazing areas. Because reindeer grazing areas are not based upon administrative boundaries, the production of reindeer meat was partitioned within each county using the GIS, assuming an even density of meat production per unit area for each reindeer grazing area.

The production values in Table 8·54 for freshwater fish only consider salmon and sea trout, and will therefore underestimate the total production. Furthermore, this table has notable omissions because data sources for harvesting rates of mushrooms were not identified.

8.7.8.2. Aggregated transfer coefficients

Aggregated transfer coefficients for the counties of Arctic Norway have been derived by comparing values for relevant

Table 8·57. Aggregated transfer coefficients for Norwegian Arctic counties.

	Aggregated transfer coefficients, m^2/kg fw				
County	Reindeer	Cow milk	Potato	Lamb	Fruit/vegetables
Finnmark	2.566	0.020	0.003	0.164	0.002
Troms	2.400	0.009	0.001	0.631	0.001
Nordland	0.971	0.014	0.005[a]	0.142	0.001

a. Based on national Norwegian values.

products compiled in the AMAP data center with predicted values using the GIS. The T_{ag} values shown in Table 8·57 represent the maximum calculated values prior to the Chernobyl accident in 1986; these have been taken to be representative of transfer soon after a contamination event and would constitute a worst-case scenario. In the AMAP assessment, the maximum yearly average T_{ag} values for reindeer meat, calculated at a country level, for Fennoscandia ranged between 1.1871 and 2.2981 m^2/kg (Howard *et al.* 1996). In the absence of relevant data for Arctic Norway, recommended values have been used for goat milk (0.004 m^2/kg), moose (0.02 m^2/kg), beef (0.006 m^2/kg) and wildfowl/small game (0.02 m^2/kg) (Howard *et al* 1996). For freshwater fish, a value of 0.03 m^2/kg was adopted based on data from Saxén *et al.* (1996).

The T_{ag} value used for reindeer meat in Nordland is almost three times lower than that for Finnmark and Troms, but is based upon far fewer observations. Indeed, because sample numbers are low and taken over many years, it is not possible to statistically test whether the transfer values used for each county are significantly different. T_{ag} values are highly variable between seasons and years. For example, T_{ag} values in reindeer meat are higher in winter when reindeer feed exclusively upon lichen that may contain high radiocaesium concentrations. Similarly, using a single T_{ag} value for lamb neglects the seasonal pattern of sheep grazing in Norway (sheep are stabled in the winter and fed upon stored feed) and assumes an exclusive consumption of locally-produced feed. However, the slaughter of reindeer and sheep normally occurs once, during the autumn/winter, in a given year. These temporal variations make it difficult to compare T_{ag} values between sites in different years. To predict changes with time, T_{ag} values need to be combined with effective ecological half-lives.

8.7.8.3. **Total ^{137}Cs output**

The total ^{137}Cs output was estimated for an assumed contamination event of 100 kBq/m^2 uniformly distributed over the three counties. Predicted ^{137}Cs contamination levels in each product were calculated for each spatial unit and then the total Bq contained in the total output of each product was calculated by multiplying the predicted activity concentration by the weight of product.

The data are summarized in Figure 8·78 where it is clear that reindeer constitutes the most important route of ^{137}Cs output in Finnmark and Troms, but not in Nordland. In Nordland, the comparative importance of different food products in the first year declines in the order:

> cow milk > reindeer > lamb > beef > goat milk > potato > moose

while in Troms the order of importance of different food products is:

> reindeer > lamb > cow milk > goat milk > beef > potatoes > moose

and in Finnmark it is:

> reindeer > cow milk > lamb > freshwater fish > beef > moose > potatoes.

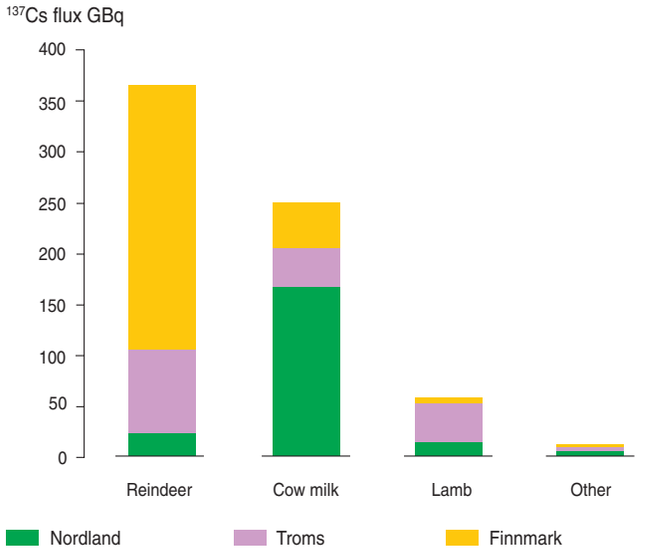

Figure 8·78. Relative ^{137}Cs output in foodstuffs from the different Arctic Norwegian counties in the first year after an assumed uniform deposition of 100 kBq/m^2.

The total ^{137}Cs flux from each county declines in the order:

> Finnmark > Nordland > Troms

with total estimated fluxes for the first year of 3.0×10^{11}, 2.1×10^{11} and 1.7×10^{11} Bq, respectively, from the ten products in Table 8·54, (i.e. the listed products excluding fungi, for which no data are available, berries and freshwater fish).

Such an approach can be used for the first year after deposition, although caution must be used as the initial phase of direct ingestion of radiocaesium deposited onto grass surfaces or vegetables, if a release occurred during the growing season, would not be incorporated.

To account for the reduction in contamination levels with time, the total ^{137}Cs output over a 50-year period has been calculated, using appropriate effective ecological half-lives for the major three products which dominate the total ^{137}Cs flux in each county, namely reindeer, cow milk and lamb. The effective ecological half-lives used were based on the most relevant Norwegian data available and were:

> Reindeer: 4 years
> Cow milk: 4 years
> Lamb: 15 years

The relative importance of the three different products over a 50-year period for each county is shown in Figure 8·79 (next page). For all three counties, the relative contribution to radiocaesium output from lamb has increased due to the longer effective ecological half-life. In Nordland, cow milk still provides the most significant contribution to radiocaesium output whilst in Finnmark reindeer meat still dominates. Lamb has become the most important food product in terms of radiocaesium output for the county of Troms.

It must be emphasized that, due to limited data availability, the analysis does not currently include mushrooms, freshwater fish and berries. After the Chernobyl accident, these products were shown to be potentially important sources of radiocaesium intake due to their high radiocaesium activity concentrations (especially for mushrooms) and to high rates of consumption by certain population groups (Skuterud *et al.* 1997).

Regional output does not necessarily relate directly to human consumption, because some produce will be exported to other areas. In the same way, no estimation of food products imported into the three counties is consid-

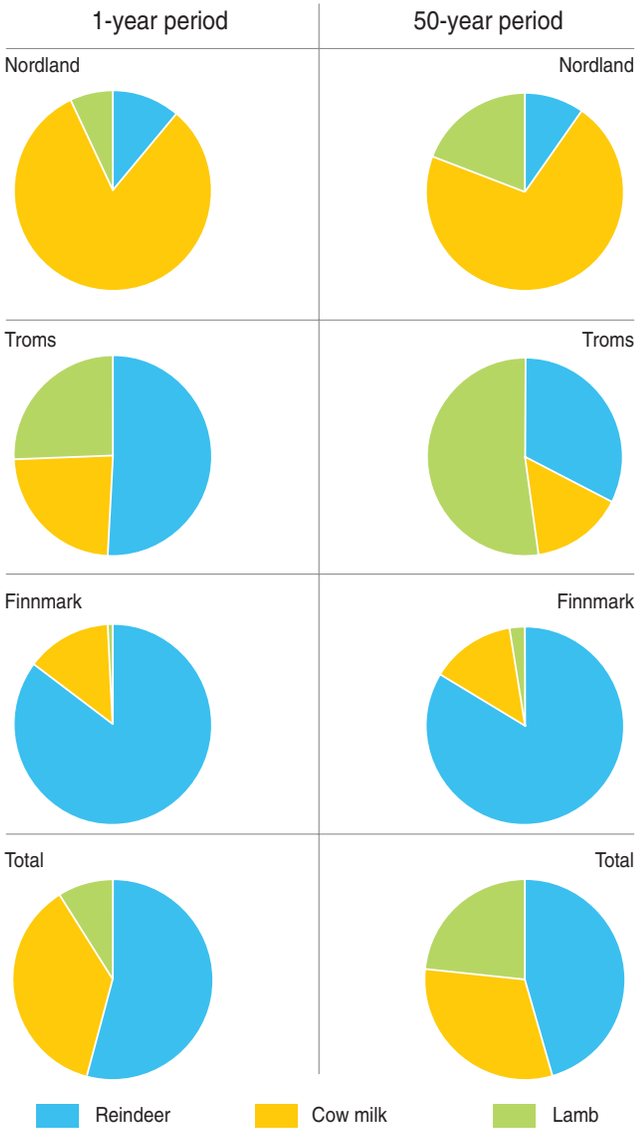

Figure 8·79. The comparative importance of different foodstuffs as sources of ^{137}Cs for the three Arctic Norwegian counties over the 1-year and 50-year period following an assumed uniform deposition of 100 kBq/m^2.

ered. However, if production associated with indigenous people living by traditional methods can be identified, the total flux to these people could be estimated and compared with individual dose assessments. Total ^{137}Cs intake will also be affected by imported food but the comparative importance of these foodstuffs as a source of radiocaesium compared with locally-produced food is likely to be low. To assess the importance of internal cycling within discrete Arctic areas it is important to consider the numbers and proportion of indigenous people in each area.

8.7.8.4. Spatial distribution of the Norwegian Arctic population

The population of the three Arctic Norwegian counties has been divided into three categories: urban, rural-non-indigenous and rural-indigenous. The total number of inhabitants of the three counties decreases with increasing latitude so that the most northerly county, Finnmark, has the fewest inhabitants and the greatest number of indigenous people (Figure 8·80).

Both the relative proportion of the total population and the total number of indigenous (Saami) show increases with latitude.

8.7.8.5. Conclusions

The comparative importance of different foodstuffs in contributing to collective dose varies spatially due to differing rates of production and transfer. In this assessment, Finnmark has been identified as being most vulnerable to radiocaesium deposition due to the importance of reindeer meat, the large proportion of indigenous people and proximity to potential Russian sources of radioactive contamination. Currently, there are clear limitations to this initial approach which relies upon the homogeneity of production and transfer within certain spatial units and uses conservative T_{ag} values, thereby constituting a worst-case scenario. Furthermore, the method does not consider the time of year in which a release occurs. Accordingly, it would probably underestimate radiocaesium contamination of milk and fresh vegetables in the first month if the release occurred during the growing season.

The integration of information on deposition, transfer and production, within a GIS, provides an efficient and rapid method of quantifying collective dose in the event of a radioactive release. However, it is important to incorporate appropriate information on the spatial variation in transfer. For comparative purposes, we have assumed an unrealistic even deposition occurring as the result of a nuclear release. If information about the spatial variation of radiocaesium deposition could be integrated with this methodology, within the GIS, either using dispersion modeling or actual measurements, it would be possible to refine current predictions of the consequences of a nuclear release. For example, the consequences of a release at the Kola nuclear power plant could be more accurately predicted for Finnmark.

Further improvements to the spatial analysis of vulnerability to radiocaesium could be achieved by considering the following factors:

- Identification of areas used for different types of agricultural production.
- Consideration of import and export of both food products and animal feedstuffs.
- Identification of soil type and ecosystem dependent transfer values.
- Incorporation of the effect of possible countermeasures.

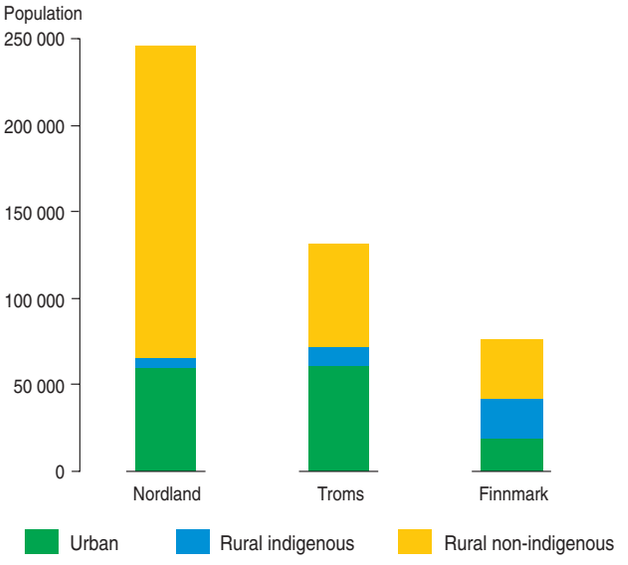

Figure 8·80. Distribution of the population of the three Arctic Norwegian counties.

Further data refinements would include improved consideration of potentially significant sources such as mushrooms, freshwater fish and berries for which data on transfer, harvesting and consumption are currently inadequate.

8.7.9. Summary

Arctic ecosystems and food production systems have been considered with respect to their vulnerability to deposition from a radioactive release. Pathways and factors influencing fluxes of radionuclides in the Arctic have been discussed, including the location and dietary habits of some Arctic indigenous peoples. Appropriate parameters and methods for modeling radionuclide transfer have been considered, and Geographical Information Systems used to collate, store and analyse data. Studies have been undertaken to compare potential fluxes through important Arctic food products in different geographical areas.

It is clear that the most vulnerable food product to contamination following a radioactive release is reindeer/caribou meat, although, under some circumstances, other products such as milk and lamb may also be important. The comparative importance of different foodstuffs varies within Arctic regions of each country, and therefore generalizations based at a country level may be inappropriate. The potential contribution of foodstuffs, which are known to readily accumulate radiocaesium, needs further consideration particularly for mushrooms, freshwater fish and berries.

The total number, geographical distribution and dietary composition of indigenous peoples within different Arctic regions are important factors affecting potential individual and collective doses arising from a nuclear accident in the Arctic. To date, little analysis has been carried out concerning the effect of the spatial distribution of the indigenous people, variations in diet among ethnic groups, or variations in transfer rates to major food items.

8.8. Conclusions and recommendations

8.8.1. Conclusions

The overall conclusion of this assessment is that the greatest threats 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 following provides detailed conclusions arising from the assessment:

- Large-scale contamination of the Arctic with artificial radionuclides is derived from three primary sources: global fallout from past atmospheric nuclear weapons testing; releases from European nuclear fuel reprocessing plants; and fallout from the Chernobyl reactor accident.
- Some localized areas of the Arctic are also contaminated with radionuclides from other sources such as nuclear device explosions, spent fuel storage sites, and radioactive wastes dumped at sea. In the case of radioactive wastes dumped at sea and releases from underground and underwater nuclear explosions, the radionuclides remain mainly localized. Radionuclides released from Russian fuel reprocessing plants and in liquid radioactive wastes dumped in the Arctic marine environment have been distributed more widely. Nevertheless, the additional contamination of the Arctic by radionuclides from these diverse sources is of negligible radiological significance.
- The levels of artificial radionuclides in the Arctic attained maximum values during the period 1950-1970, primarily as a consequence of atmospheric nuclear weapons testing. Following the cessation of widespread atmospheric weapons testing in the early-1960s, other sources, such as releases from European nuclear fuel reprocessing plants, increased in relative importance. A second, but lower, peak in fission product radionuclides occurred in the Arctic marine environment in the early 1980s as a consequence of the peak in the rates of radionuclide discharge from Sellafield in the mid-1970s. Finally, fallout from the Chernobyl accident in 1986 made an additional contribution to radionuclide contamination of the Arctic. Since then, the levels of radionuclides have been in general, but not ubiquitous, declining.
- The major contribution to radiation doses of Arctic residents delivered by artificial radionuclides originates from previous nuclear weapons explosions in the atmosphere giving rise to global fallout. However, in some geographically limited, but populated, areas of the Arctic (Fennoscandia and western Russia), a substantial dose contribution has been made by additional fallout from the Chernobyl reactor accident. This contribution to the dose to Norwegian and Swedish Arctic residents was, and continues to be, reduced through the application of justified countermeasures.
- Arctic residents, whose diets comprise a large proportion of traditional terrestrial and freshwater foodstuffs, receive the highest radiation exposures to both natural and artificial radionuclides in the Arctic. Doses to members of both the average population and selected indigenous population groups in the Arctic depend on the rates of consumption of locally-derived terrestrial and freshwater foodstuffs, including reindeer/caribou, freshwater fish, goat cheese, berries, mushrooms and lamb. In contrast, Arctic residents having diets largely comprising marine foodstuffs receive comparatively low radiation exposures because of the lower levels of contamination of marine organisms.
- The vulnerability of Arctic terrestrial ecosystems results in a five-fold higher exposure to radioactive contamination compared to that in temperate areas. Because of the unique ecology of the Arctic, the comparative importance of both radionuclides and exposure pathways differs from those in temperate areas. For example, exposures to artificial radionuclides are dominated by ^{137}Cs contained in a wide variety of traditional Arctic (native) foods of terrestrial and freshwater origin but most importantly reindeer/caribou meat. For reindeer-herders and others consuming comparatively large quantities of caribou/reindeer meat, the dominant pathway of natural radiation exposure is the intake of ^{210}Po through caribou/reindeer meat consumption. Furthermore, unlike the situation in temperate areas, where immediate exposures to radioiodine are of primary concern following accidents, in the Arctic the low rate of milk production reduces the significance of this pathway.
- The highest time-integrated radiation exposures to members of average populations of the eight Arctic countries

from global fallout occurred in Canada, and the lowest in Greenland. The variations in individual dose distribution are not primarily due to geographical heterogeneities in radionuclide fallout. Rather, they result from variations in diet among Arctic residents. Indigenous peoples comprise a relatively high proportion of the inhabitants of Arctic Canada, some of whom rely comparatively heavily on caribou as a source of food. In contrast, the population of Greenland is confined to coastal areas and has a diet containing a comparatively large proportion of marine foodstuffs having low radionuclide contamination.

- Selected indigenous Arctic population groups can have individual radiation exposures up to 50 times larger than those of the members of the average populations. Individual doses within these selected groups are distributed among the Arctic countries in a similar manner to those to the average populations. It cannot be ruled out that there are small numbers of individuals within other Arctic countries having similar dietary habits to those of the selected Canadian community which has the highest calculated individual doses. Accordingly, comparable, or indeed higher, doses than those calculated for the Canadian subgroup may exist within the Arctic.
- Releases of radionuclides from the Thule B-52 accident, the sunken *Komsomolets* nuclear submarine and radioactive wastes dumped in the Arctic marine environment have not resulted in any significant increases in human exposures or risks to human health of Arctic residents. There is minimal likelihood of significant radiological consequences associated with any future releases of radionuclides from dumped radioactive wastes or from the sunken submarine *Komsomolets*. There is inadequate radiological justification for remediation measures to be taken in the case of either radioactive wastes dumped in the Kara Sea or the *Komsomolets* submarine.
- The greatest radiological threats to human health and the environment in the Arctic are associated with the potential for nuclear accidents and failures in the containment of the large inventories of radioactive materials in storage such as high-level liquid and solid wastes. Issues of major concern in relation to the potential for effects on the Arctic environment and its inhabitants are:

  Accidents at nuclear power plants sited within, or close to, the Arctic.

  Accidents in military operations, including the handling and storage of nuclear weapons, decommissioning and refueling of nuclear powered vessels and radioactive waste storage and disposal.

  Accidents during civilian vessel operations including refueling.

  Migration of radionuclides from major uncontained sources in the drainage basins of the Ob and Yenisey rivers.

  Releases from contained sources situated in the terrestrial environment.

- It should be noted that the risk of accidents in the handling and disposal of radioactive waste, especially spent nuclear fuel, from military vessels has probably been increased by the accelerated rate of submarine decommissioning partly imposed by recent disarmament agreements. These activities have imposed additional technical, infrastructural and financial demands on processes of waste management that were already inadequate to meet the requirements of normal operations.
- There are deficiencies both in the assessments of some previous accidents and of the probabilities and consequences of accidents in contemporary activities. These need to be rectified to enable more authoritative and comprehensive evaluations to be made of the risks posed to human health and the environment by such accidents. Major effort has been devoted to determining, with high degrees of confidence and precision, the consequences of radioactive waste dumping at sea. These assessments have clearly shown that there is little associated risk to human health or the environment. However, the risks associated with other major activities, which have considerable potential for widespread and serious consequences (such as the operation of nuclear-powered vessels and of nuclear reactors in the Arctic and the handling and carriage of nuclear weapons), have been inadequately addressed. Ideally, a risk assessment for all potential sources should be undertaken, not only those of contemporary political and economic concern. The priority and detail with which assessments of practices are conducted should be commensurate with the probability and severity of consequences to humans and the environment.
- The objective of this assessment was to obtain a balanced and objective scientific assessment of the relative risks posed by radioactivity in the Arctic. The assessment is fairly comprehensive from the perspective of sources, although some sources/activities are not assessed in as much detail as others. The correction of deficiencies in source/activity coverage are reflected in the recommendations specified below. The most serious limitation in the assessment has been related to the heterogeneity in the detail with which the individual-related assessments, the consequences of previous and potential releases of radionuclides to the environment and the estimation of radiological vulnerability could be addressed. The AMAP radioactivity assessment group had to make some generalisations regarding prevailing conditions and human activities because of the diversity of population characteristics in the Arctic. It also faced difficulty with the variable reliability and detail of information available on dietary habits within population groups among the Arctic countries. Finally, information available within, or provided by, contributing countries on other relevant topics, especially the nature, consequences and probabilities of potential releases, was of variable quality and completeness. All of these deficiencies are addressed within the recommendations for further study.

8.8.2. General recommendations

- Contemporary international guidance on radiation protection, nuclear safety, radioactive waste management and emergency preparedness should be rigorously adhered to by all Arctic states to minimize the probabilities and consequences of accidents.
- More authoritative and comprehensive evaluations should be made of the risks posed to human health and the environment by accidents in nuclear power installations. Assessments of the risks of releases of radionuclides and the radiological consequences for humans and the environment should be performed for all existing nuclear installations in, and near, the Arctic, including Probabilistic Safety Analyses for nuclear power reactors, preferably at PSA Level 3.
- International recommendations regarding the improvement of nuclear and radiation safety in the nuclear industry, which cover reactor refueling, decommissioning, and associated spent fuel storage and disposal operations, should be extended to, and implemented in, nuclear fleet operations.

- Additional information should be obtained regarding: the habits and diets of Arctic residents; the transfer rates of radionuclides to terrestrial and freshwater foodstuffs; and spatial and temporal variations in production and consumption patterns of locally-produced foodstuffs. Such information would enable more precise estimates of radiological exposures and risks to Arctic inhabitants to be obtained and provide a basis for deciding on intervention measures in the event of nuclear accidents.

8.8.3. Specific recommendations

The general recommendations, above, lead to the following specific recommendations:

8.8.3.1. Recommendations regarding storage of spent nuclear fuel and radioactive waste

- Comprehensive and detailed evaluations of the handling and storage of spent nuclear fuel and other high-level radioactive waste should be continued to identify and implement any additional measures required to ensure that the risks of accidental and unauthorized releases of radionuclides to the environment are minimized.
- Spent nuclear fuel should be removed from decommissioned nuclear submarines currently located in Arctic marine areas and defueled reactor compartments should be stored safely on land.

8.8.3.2. Recommendations regarding monitoring

- A harmonized system for monitoring radionuclide contamination of the Arctic environment for radiological assessment purposes should be developed. This should be carried out in conjunction with an evaluation of the purposes and efficacy of existing trend monitoring studies. Particular attention in monitoring design should be paid to:

  Monitoring of radionuclides in key components of human exposure pathways for Arctic residents (i.e., reindeer, mushrooms, freshwater fish).

  Monitoring of atmospheric fallout in the Arctic.
- Systems should be developed for:

  Early warning monitoring of fission product releases from nuclear power plants, military and civilian nuclear vessel operations.

  Surveillance monitoring of areas of severe environmental contamination (such as those in the Ob and Yenisey drainage basins) and large contained radioactive sources on land (such as radionuclide thermal generators in navigational aids).

8.8.3.3. Recommendations for further study to correct information deficiencies

The principal deficiencies in information are of two main types: limitations in the availability of information; and gaps in scientific understanding that restrict the ability to estimate reliably the effects of nuclear activities. Limitations in the availability of information relate primarily to military activities and dietary habits. Gaps in scientific understanding relate to the terrestrial and hydrological transport of radionuclides in the environment, the role and importance of sea ice transport, and the transfer of radionuclides among environmental compartments within exposure pathways. Correction of these deficiencies requires:

- Enhanced exchange of information under the aegis of the Arctic Council/Arctic Environmental Protection Strategy. Under AMAP, an international data center on radioactive contamination and sources within the Arctic has been established. The data center's tasks should be extended to ensure that the identity and source of all relevant documents, including policy and interpretative documents, are catalogued to make them accessible throughout the Arctic States on a timely basis.
- Dietary analyses and measurements of the uptake of radionuclides into radiologically significant constituents of diet, carried out by all countries in a systematic manner, to enable improved dose and risk assessments for both average residents and members of the most-exposed population groups in the Arctic.
- Improved information on the harvesting rates of semi-natural and natural food products that are not normally included in most national statistics, particularly for mushrooms. In addition, the collation of information on harvesting of all natural and semi-natural products at a higher spatial resolution than that currently available to the AMAP assessment for most Arctic countries is warranted. Analysis of the precision of intake-related estimates of dose, and comparisons between dose estimates obtained from dietary calculations and wholebody counting, respectively, to assess the accuracy of dose estimations.
- Increased attention to:

  Characterizing source terms for uncontained environmental sources and any other sources not covered by probability safety assessments of nuclear installations.

  The study and modeling of terrestrial transport pathways of radionuclides and exposure pathways to Arctic residents to improve the resolution and comprehensiveness of radiological assessments.

  The transport of radionuclides from land sources through river catchments, particularly those of major Russian rivers.

  Estimating radiation exposures to biota and associated effects on biological populations.

  The processes of incorporation, transport and deposition of contaminants in sea ice, to enable an evaluation of the relative importance of sea ice transport of radionuclides compared with water and sediment transport in the Arctic marine environment.

  Source term reconstruction, through the analysis of cores from glaciers and accumulating freshwater and marine sediment columns.

  Determining the contribution of nuclear testing on Novaya Zemlya to local and regional fallout.

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Annex

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