RADIOLOGICAL ASSESSMENT REPORTS SERIES

RADIOLOGICAL CONDITIONS OF THE WESTERN KARA SEA:

Assessment of the radiological impact of the dumping of radioactive waste in the Arctic Seas

INTERNATIONAL ATOMIC ENERGY AGENCY

REPORT ON THE INTERNATIONAL ARCTIC SEAS ASSESSMENT PROJECT (IASAP)

RADIOLOGICAL CONDITIONS OF THE WESTERN KARA SEA

ASSESSMENT OF THE RADIOLOGICAL IMPACT OF THE DUMPING OF RADIOACTIVE WASTE IN THE ARCTIC SEAS

REPORT ON THE INTERNATIONAL ARCTIC SEAS ASSESSMENT PROJECT (IASAP)

The following States are Members of the International Atomic Energy Agency:

AFGHANISTAN
ALBANIA
ALGERIA
ARGENTINA
ARMENIA
AUSTRALIA
AUSTRIA
BANGLADESH
BELARUS
BELGIUM
BOLIVIA
BOSNIA AND HERZEGOVINA
BRAZIL
BULGARIA
BURKINA FASO
CAMBODIA
CAMEROON
CANADA
CHILE
CHINA
COLOMBIA
COSTA RICA
COTE D'IVOIRE
CROATIA
CUBA
CYPRUS
CZECH REPUBLIC
DEMOCRATIC REPUBLIC OF THE CONGO
DENMARK
DOMINICAN REPUBLIC
ECUADOR
EGYPT
EL SALVADOR
ESTONIA
ETHIOPIA
FINLAND
FRANCE
GABON
GEORGIA
GERMANY
GHANA
GREECE
GUATEMALA
HAITI
HOLY SEE
HUNGARY
ICELAND
INDIA
INDONESIA
IRAN, ISLAMIC REPUBLIC OF
IRAQ
IRELAND
ISRAEL
ITALY
JAMAICA
JAPAN
JORDAN
KAZAKHSTAN
KENYA
KOREA, REPUBLIC OF
KUWAIT
LATVIA
LEBANON
LIBERIA
LIBYAN ARAB JAMAHIRIYA
LIECHTENSTEIN
LITHUANIA
LUXEMBOURG
MADAGASCAR
MALAYSIA
MALI
MALTA
MARSHALL ISLANDS
MAURITIUS
MEXICO
MONACO
MONGOLIA
MOROCCO
MYANMAR
NAMIBIA
NETHERLANDS
NEW ZEALAND
NICARAGUA
NIGER
NIGERIA
NORWAY
PAKISTAN
PANAMA
PARAGUAY
PERU
PHILIPPINES
POLAND
PORTUGAL
QATAR
REPUBLIC OF MOLDOVA
ROMANIA
RUSSIAN FEDERATION
SAUDI ARABIA
SENEGAL
SIERRA LEONE
SINGAPORE
SLOVAKIA
SLOVENIA
SOUTH AFRICA
SPAIN
SRI LANKA
SUDAN
SWEDEN
SWITZERLAND
SYRIAN ARAB REPUBLIC
THAILAND
THE FORMER YUGOSLAV REPUBLIC OF MACEDONIA
TUNISIA
TURKEY
UGANDA
UKRAINE
UNITED ARAB EMIRATES
UNITED KINGDOM OF GREAT BRITAIN AND NORTHERN IRELAND
UNITED REPUBLIC OF TANZANIA
UNITED STATES OF AMERICA
URUGUAY
UZBEKISTAN
VENEZUELA
VIET NAM
YEMEN
YUGOSLAVIA
ZAMBIA
ZIMBABWE

The Agency's Statute was approved on 23 October 1956 by the Conference on the Statute of the IAEA held at United Nations Headquarters, New York; it entered into force on 29 July 1957. The Headquarters of the Agency are situated in Vienna. Its principal objective is "to accelerate and enlarge the contribution of atomic energy to peace, health and prosperity throughout the world''.

Printed by the IAEA in Austria
December 1998
STI/PUB/1068

RADIOLOGICAL ASSESSMENT REPORTS SERIES

RADIOLOGICAL CONDITIONS OF THE WESTERN KARA SEA

ASSESSMENT OF THE RADIOLOGICAL IMPACT OF THE DUMPING OF RADIOACTIVE WASTE IN THE ARCTIC SEAS

REPORT ON THE INTERNATIONAL ARCTIC SEAS ASSESSMENT PROJECT (IASAP)

INTERNATIONAL ATOMIC ENERGY AGENCY
VIENNA, 1998

VIC Library Cataloguing in Publication Data

Radiological conditions of the Western Kara Sea ; assessment of the radiological impact of the dumping of radioactive waste in the Arctic seas, report on the International Arctic Seas Assessment Project (IASAP). — Vienna : International Atomic Energy Energy, 1998.

p. ; 24 cm. — (Radiological assessment reports series, ISSN 1020–6566)

STI/PUB/1068

ISBN 92–0–104098–9

Includes bibliographical references.

1. Radioactive pollution—Kara Sea (Russia). 2. Kara Sea (Russia)—Environmental conditions. I. International Atomic Energy Agency. II. Series.

VICL 98–00210

FOREWORD

This report presents the findings of the International Arctic Seas Assessment Project (IASAP), which was instituted in 1993 and concluded in 1996 to address concerns over the potential health and environmental impacts of high level radioactive waste dumped in the shallow waters of the Arctic Seas. The IASAP study was endorsed by the Contracting Parties to the Convention on Prevention of Marine Pollution by Dumping of Waste and Other Matter, and a summary of the results and conclusions was provided to the Contracting Parties in February 1997.

The study covered the following aspects: (i) examination of the current radiological situation in Arctic waters due to the dumped wastes; (ii) the evaluation of potential future releases from the dumped wastes; (iii) prediction of environmental transport of potential releases and assessment of the associated radiological impact on humans and biota; and (iv) examination of the feasibility, costs and benefits of possible remedial measures.

Information on the situation at the dumping sites and surrounding sea areas was obtained by means of exploratory visits organized by the Joint Norwegian–Russian Expert Group. The study was partially supported by the Government of the United States of America.

This report represents the joint efforts of the scientists taking part in the IASAP Advisory Group and various working groups. Their names are listed at the end of the report. The main drafters of the sections were A. Salo, Finland (Section 2), P. Povinec, IAEA-RIML (Section 3), N. Lynn, UK (Section 4), M. Scott, UK (Section 5), J. Schwarz, Germany (Section 6) and J. Cooper, UK (Section 7). A special acknowledgement is due to the Chairperson of the Advisory Group, A. Salo, the Chairpersons of the working groups R. Dyer (USA), M. Scott and J. Schwarz. The IAEA staff member responsible for co-ordinating the study was K-L. Sjoeblom of the Division of Radiation and Waste Safety.

Other reports issued or to be issued under the IASAP study are:

Predicted Radionuclide Release from Marine Reactors Dumped in the Kara Sea: Report of the Source Term Working Group of the International Arctic Seas Assessment Project (IASAP), IAEA-TECDOC-938 (1997).

Anthropogenic Radionuclides in the Arctic Seas: Report to the International Arctic Seas Assessment Project (IASAP), IAEA-TECDOC (in preparation).

Modelling of the Radiological Impact of Radioactive Waste Dumping in the Arctic Seas: Report of the Modelling and Assessment Working Group of the International Arctic Seas Assessment Project (IASAP), IAEA-TECDOC (in preparation).

CONTENTS

EXECUTIVE SUMMARY

1. BACKGROUND

In 1992, it was reported that the former USSR had, for over three decades, dumped radioactive wastes in the shallow waters of the Arctic Seas. This news caused widespread concern, especially in countries with Arctic coastlines. The IAEA responded by outlining an international study to assess the health and environmental implications of the dumping. The plan was endorsed by the Fifteenth Consultative Meeting of the Contracting Parties to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention 1972). The Consultative Meeting requested that the study include consideration of possible remedial actions, e.g. the retrieval of the wastes for land storage.

The International Arctic Seas Assessment Project (IASAP) was launched by the IAEA in 1993 with the following objectives:

(1) To assess the risks to human health and to the environment associated with the radioactive wastes dumped in the Kara and Barents Seas; and
(2) To examine possible remedial actions related to the dumped wastes and to advise on whether they are necessary and justified.

This project was carried out by a multidisciplinary team of scientists from several countries within the normal Agency procedures, i.e. through a Co-ordinated Research Programme, technical contracts and consultancies. It was steered by an international Advisory Group. The project was partially supported by extrabudgetary funding from the United States of America and was co-ordinated with and supported by the Norwegian–Russian Expert Group for Investigation of Radioactive Contamination in the Northern Areas.

The team adopted the following approach:

- It examined the current radiological situation in Arctic waters, to assess whether there is any evidence for releases from the dumped waste.
- It predicted potential future releases from the dumped wastes concentrating on the solid high level waste objects containing the major part of the radionuclide inventory of the wastes.
- It modelled environmental transport of released nuclides and assessed the associated radiological impact on humans and biota.
- It examined the feasibility, costs and benefits of possible remedial measures applied to a selected high level waste object.

The total amount of radioactive waste dumped in Arctic Seas was estimated in the White Book of the President of Russia[1] to be approximately 90 PBq (90×10^{15} Bq) at the time of dumping. The dumped items included six nuclear submarine reactors containing spent fuel; a shielding assembly from an icebreaker reactor which contained spent fuel, ten nuclear reactors without fuel, and solid and liquid low level waste. Of the total estimated inventory, 89 PBq was contained in high level wastes comprising reactors with and without spent fuel. The solid wastes, including the above reactors, were dumped in the Kara Sea, mainly in the shallow fjords of Novaya Zemlya, where the depths of the dumping sites range from 12 to 135 m, and in the Novaya Zemlya Trough at depths of up to 380 m. Liquid low level wastes were released in the open Barents and Kara Seas. The dumping sites are indicated on the map (Fig. 2 in Section 1).

Additional information regarding the nature of the wastes has been obtained from Russian and international sources. There are, however, certain important gaps in the available information. For example, not all of the dumped high level wastes referred to in Russian Federation documents have been located or unambiguously identified. Furthermore, some information on the construction of the dumped reactors remains classified. Thus, the conclusions of this study are valid only in the context of the information publicly available at the time it was made.

2. RADIOLOGICAL PROTECTION CONSIDERATIONS

The basic concepts of radiological protection relevant to this project are those recommended by the International Commission on Radiological Protection (ICRP) and incorporated into the International Basic

[1] Facts and Problems Related to Radioactive Waste Disposal in the Seas Adjacent to the Territory of the Russian Federation, Materials for a Report by the Governmental Commission on Matters Related to Radioactive Waste Disposal at Sea, Established by Decree No. 613 of the Russian Federation President, 24 October 1992 (1993).

Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources (BSS) of the IAEA and other international agencies. These documents identify two classes of situations in which humans may be exposed to radiation – those for which protection measures applied at source can be planned prospectively, and the introduction of the source is a matter of choice, and other situations, where the sources of exposure are already present or unavoidable and so protective measures have to be considered retrospectively. These are characterized, respectively, as *practices* and *interventions*. The situation being considered in this assessment falls within the category of interventions. In this case, intervention could in principle be applied at source or, following radionuclide release, to the environmental exposure pathways through which humans might be exposed. Intervention at source could include, for example, the introduction of additional barriers to prevent radionuclide release. Intervention applied to environmental exposure pathways could involve restricting consumption of contaminated food and/or limiting access to contaminated areas. In either case, it is required that remedial actions be justified on the basis that the intervention does more good than harm, i.e. the advantages of intervening, including the reduction in radiological detriment, outweigh the corresponding disadvantages, including the costs and detriment to those involved in the remedial action. Furthermore, the form and scale of any intervention should be optimized to produce the maximum net benefit.

For the purposes of deciding on the need for remedial actions, the most important aspects of radiological impact are:

(1) The doses and risks to the most exposed individuals (the critical group) if action is not taken and the extent to which their situation can be improved by taking action; and
(2) The total health impact on exposed populations and how much of it can be avoided by taking remedial action. The total health impact is considered to be proportional to the collective dose, i.e. the sum of individual doses in an exposed population.

The high level radioactive wastes dumped in the Kara Sea and adjoining fjords are in discrete packages, which are expected to leak some time in the future. They therefore constitute a *potential* chronic exposure situation where the concern relates to future increments of dose to exposed individuals resulting from releases of radionuclides from the dumped wastes. Depending on the physical condition of these sources, intervention (remediation) at source is the most viable course of action rather than intervention at some later time in environmental exposure pathways. The precondition for intervention is that it is both justified and optimized. Currently, there are no internationally agreed criteria for invoking a requirement to remediate in chronic exposure situations except in the case of exposure of the public to radon, a naturally occurring radioactive gas, where international guidance suggests an action level at an incremental annual dose in the range of 3–10 mSv. Both the ICRP and the IAEA are developing guidance for applications to other types of intervention situations.

For perspective in the present assessment, it is worth noting that increments in individual radiation dose of a few µSv/a can be regarded as trivial, both in terms of the associated risk to health and on the basis of comparison with radiation exposures due to natural sources and their variation. The worldwide annual average radiation dose due to natural background radiation, excluding that due to radon gas, is about 1 mSv, and values up to 10 mSv occur depending upon local geology. The average annual radiation dose due to natural background radiation including radon exposure is 2.4 mSv.

Finally, it is noted that the discussion in this report is confined to the radiological aspects of decision making regarding the need for remedial action. The political, economic and social considerations that must form an important part of the decision making process are not considered here and are largely matters for the national government having jurisdiction and responsibility over the dumped radioactive wastes.

3. CURRENT RADIOLOGICAL SITUATION

The IASAP examined the current radiological situation in the Arctic, analysing information acquired during a series of joint Norwegian–Russian trips and other international expeditions to the Kara Sea. In addition, recent oceanographic and radiogeochemical surveys have provided new information on the physical, chemical, radiochemical and biological conditions and processes in the Arctic Seas. The open Kara Sea is relatively uncontaminated compared with some other marine areas, the main contributors to its artificial radionuclide content being direct atmospheric deposition and catchment runoff of global fallout from nuclear weapon tests, discharges from reprocessing plants in western Europe and fallout from the Chernobyl accident. The measurements of environmental materials suggest that annual individual doses from artificial radionuclides in the Kara and Barents Seas are only in the range of 1 to 20 µSv.

In two of the fjords where both high and low level wastes were dumped, elevated levels of radionuclides were detected in sediments within a few metres of the low level waste containers, suggesting that some containers have leaked. However, these leakages have not led to a measurable increase of radionuclides in the outer parts of the fjords. At present, therefore, the dumped wastes have a negligible radiological impact.

4. FUTURE RADIOLOGICAL SITUATION

The assessment of the potential risks posed by possible future releases from the dumped wastes focused on the high level waste objects containing the majority of the radioactive waste inventory. Release rates from these wastes were estimated and the corresponding radiation doses to humans and biota were assessed using mathematical models for radionuclide transfer through the environment.

Source inventories and release rates

In order to provide appropriate release rate scenarios that can be used as input terms to the modelling of transport and exposure pathways leading to exposure estimates for humans and biota, the team examined in considerable detail the characteristics of the dumped reactors and their operating histories. This information, based on reactor operating histories and calculated neutron spectra, provided estimates of fission product, activation product and actinide inventories of the dumped reactors and fuel assemblies. It was concluded that the total radionuclide inventory of the high level radioactive waste objects at the time of dumping was 37 PBq. The difference between this value and the preliminary estimate of 89 PBq given in the Russian 'White Book' can be explained by the more accurate information on the actual operating history of the reactors provided to IASAP by the Russian authorities. The corresponding inventory of high level dumped wastes in 1994 was estimated to be 4.7 PBq of which 86% are fission products, 12% activation products, and 2% actinides. The main radionuclides in these categories were ^{90}Sr and ^{137}Cs; ^{63}Ni; and ^{241}Pu, respectively.

The rates of release of radionuclides to the environment will depend upon the integrity of materials forming the reactor structure, the barriers added before dumping and the nuclear fuel itself. For each of the dumped high level waste objects, the team investigated in detail the construction and composition of barriers, identified weak points and used the best estimates of the corrosion rates and barrier lifetimes in the calculation of release rates. External events, such as collision of ships or, more generally, global cooling following by glacial scouring of the fjords could damage the containment. This would lead to faster releases of radionuclides to the environment. In order to adequately represent the possible range of release rates to the environment, the team considered three release scenarios:

A. a *best estimate* scenario — release occurs via the gradual corrosion of the barriers, waste containers and the fuel itself;
B. a *plausible worst case* scenario — normal gradual corrosion followed by a 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; and
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.

It should be noted that no attempt was made to assign probabilities to the events described in plausible worst case and climate change scenarios and the consequences have been assessed on the assumption that such events will occur in the years indicated.

For the best estimate scenario, the combined release rate from all sources peaks at about 3000 GBq/a (1 GBq = 10^9 Bq) within the next 100 years with a second peak of about 2100 GBq/a in about 300 years. For most of the remaining time, total release rates lie between 2 and 20 GBq/a. The plausible worst case scenario results in a release 'spike' of 110 000 GBq followed by releases of between 100 and 1000 GBq/a for the next few hundred years due to the accelerated release of radionuclides from the fuel container and reactor compartment of the nuclear icebreaker. In the climate change scenario, which assumes that glacial scouring causes an instantaneous release of the remaining inventory of all the wastes in the year 3000, about 6600 GBq are released.

Modelling and assessment

The calculated release rates were used with mathematical models of the environmental behaviour of radionuclides to estimate radiation doses to humans and biota. Different modelling approaches were adopted, and experts from several countries and from the IAEA participated in the exercise. Substantial effort was devoted to a synthesis of existing information on marine ecology, oceanography and sedimentology of the target area as a basis for model development. Specific processes were identified as peculiar to the area and, thus, of potential importance for incorporation into models. Because of the need to provide predictions on very diverse space- and time-scales, a number of different models for the dispersal of radionuclides within and from the Arctic Ocean were developed. The team of experts adopted two main modelling approaches: compartmental or box models, and hydrodynamic circulation models. In addition, one hybrid model (using compartmental structure but on a finely resolved spatial scale) was developed and applied. By modelling advective and diffusive dispersal, compartmental models provide long time-scale, spatially averaged, far field predictions, while the hydrodynamic models provide locally resolved, short time-scale results.

Separate attention was devoted to one of the most poorly quantified transport pathways — sea ice transport. A simple exemplar calculation, or scoping exercise, demonstrated that, for the radioactive waste sources considered here, sea ice transport would make only a small contribution to individual dose compared with the transport of radionuclides in water.

For estimation of doses to individuals, the team of experts considered individuals in three population groups. Calculations of individual doses were undertaken for time periods covering the peak individual dose rates for each of the three scenarios identified. The groups were defined as follows:

(1) Groups 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, seabirds and their eggs, and who spend 250 hours/year on the seashore. These habits are also typical of subsistence fishing communities in other countries with Arctic coastlines.

(2) A hypothetical group of military personnel patrolling the foreshores of the fjords containing dumped radioactive materials, for assumed periods of 100 hours/year. The exposure pathways considered include external radiation and the inhalation of seaspray and resuspended sediment.

(3) A group of seafood consumers considered representative of the northern Russian population situated on the Kola peninsula eating fish, molluscs and crustaceans harvested from the Barents Sea. No consideration was given to the consumption of seaweed or marine mammals, or to external radiation.

The calculated peak doses to members of these three groups from all sources are shown in Table I.

The total annual individual doses in each critical group of seafood consumers (Groups 1 and 3) for all three scenarios are small and very much less than variations in natural background doses. Doses to the hypothetical critical group of military personnel patrolling the fjords (Group 2) are higher than, but nevertheless comparable to, natural background doses.

TABLE I. MAXIMUM TOTAL ANNUAL INDIVIDUAL DOSES FOR SELECTED POPULATION GROUPS

Scenario	Annual doses[a] to seafood consumers (Groups 1 & 3) (μSv)	Annual doses to military personnel (Group 2) (μSv)
Best estimate scenario	< 0.1	700
Plausible worst case scenario	< 1	4000
Climate change scenario	0.3	3000

[a] For perspective, the annual doses to the critical groups 1 and 3 from naturally occurring polonium-210 in seafood are 500 and 100 μSv, respectively. In addition, the worldwide average total annual dose from natural background radiation is 2400 μSv.

Collective doses were estimated only for the best estimate release rate scenario. The collective dose to the world population arising from the dispersion of radionuclides in the world's oceans (nuclides other than ^{14}C and ^{129}I) were calculated for two time periods: (i) up to the year 2050 to provide information on the collective dose to the current generation; and (ii) over the next 1000 years, a time period which covers the estimated peak releases. Because of the increasing uncertainties in predicting future events, processes and developments, it was not considered meaningful to extend the assessment beyond 1000 years. Nevertheless, these calculations provide some illustration of the temporal distribution of dose. The estimated collective doses are 0.01 and 1 man·Sv, respectively.

The team used appropriate global circulation models to calculate collective doses from ^{14}C and ^{129}I, which are long lived 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 some 8 man·Sv. The corresponding value for ^{129}I is much lower, i.e.

0.0001 man·Sv. Thus, the total collective dose over the next 1000 years to the world's population from all radionuclides in the dumped radioactive waste is of the order of 10 man·Sv. By comparison, the annual collective dose to the world's population from naturally occurring polonium-210 in the ocean is estimated in other studies to be about three orders of magnitude higher. It is also informative to compare the collective dose associated with wastes dumped in the Kara Sea with the collective dose estimated for low level radioactive waste dumped in the north-east Atlantic. The collective dose from the latter practice to the world population is 1 man·Sv over 50 years and 3000 man·Sv over 1000 years.

The team of experts calculated the radiation dose rates to a range of populations of wild organisms, from zooplankton to whales, and found them to be very low. The peak dose rates predicted in this assessment are about 0.1 μGy/h — a dose rate that is considered unlikely to entail any detrimental effects on morbidity, mortality, fertility, fecundity or mutation rate that may influence the maintenance of healthy populations. It is also relevant to note that only a small proportion of the biota population in local ecosystems could be affected by the releases. The team concluded that the dumping in the Kara Sea has no radiological implications for populations of aquatic organisms.

5. REMEDIATION

Feasibility and costs

A preliminary engineering feasibility and cost study was conducted for five remediation options for the container of spent fuel from a nuclear icebreaker. This source was chosen because it contains the largest radionuclide inventory among the dumped waste objects and is the best documented as to construction and introduced container barriers.

The five specific options selected for evaluation were:

(1) Injection of material to reduce corrosion and to provide an additional release barrier;
(2) Capping in situ with concrete or other suitable material to encapsulate the object;
(3) Recovery and removal to a land environment;
(4) Disposal into an underwater cavern on the coast of Novaya Zemlya; and
(5) Recovery and relocation to a deep ocean site.

Further consideration of these five options by salvage experts screened out options 1, 4 and 5. Option 1 was screened out on the grounds that the spent fuel package had previously been at least partly filled with Furfurol(F), which might make the injection of additional material difficult. Option 4 was omitted from further consideration because the creation of an underwater cavern would be too expensive a proposition for a single recovered source and would have to be justified in a larger context. Option 5 was discarded because, first, it is doubtful whether special approval could be obtained under the London Convention 1972 for an operation that entailed redumping of a high level waste object in the ocean, and, second, underwater transport on the high seas would involve undue risks of losing the package during carriage to a new disposal site. Further evaluation of remedial actions was therefore confined to the two remaining options, i.e. in situ capping and recovery for land treatment or disposal. Both options were deemed technically feasible and the costs of marine operations were estimated to be in the range of US $5–13 million. It should be appreciated that for the recovery option, there would be major additional costs to those considered here for subsequent land transport, treatment, storage and/or disposal. Radiation exposures to the personnel involved in remedial actions were considered, as was the likelihood of a criticality accident. It was concluded that, with the appropriate precautions and engineering surveys proposed as a basis for proceeding with remediation, the radiation risks to the personnel involved in remedial activities would not be significant.

Analysis of the justification for remediation

A number of factors require consideration in reaching a decision about the need for remedial actions. From a radiological protection perspective, the most important is that of health risks to individuals and populations if no remediation is undertaken.

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 the Kara and Barents Seas. The risk of fatal cancer induction from a dose of 1 μSv is estimated to be about 5×10^{-8} — a trivial risk. Therefore, members of local populations will not be exposed to significant risks from the dumped wastes. The predicted future doses to the members of the hypothetical group of military personnel patrolling the foreshores of the fjords of Novaya Zemlya are higher than those predicted for members of the public and are comparable with doses from natural background radiation. Taking into account that the doses to this hypothetical group could be controlled if required, none of the calculated individual doses indicates a need for remedial action.

Although the risks to each individual may be trivial, when summed over a population, some health effects might be predicted to arise as a result of the additional exposure. These health effects are considered to be proportional to the collective dose arising from the dumped radioactive wastes. The collective dose to the world's population over the next 1000 years from the radioactive wastes dumped in the Barents and Kara Seas is of the order of 10 man·Sv. This calculated collective dose is small but should, nevertheless, be considered further in reaching a decision about the need for remediation. A simplified scoping 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 were implemented. If this scoping approach indicates that remedial action might be justified, a more detailed analysis in which the components of the collective dose are more closely examined would be warranted. Using the scoping approach, it can be shown that remedial measures applied to the largest single source (the spent fuel package from the nuclear icebreaker) costing in excess of US $200 000 would not appear to offer sufficient benefit to be warranted. Since any of the proposed remedial actions would cost several million US dollars to carry out it is clear that, on the basis of collective dose considerations, remediation is not justified.

Overall, from a radiological protection viewpoint, including consideration of the doses to biota, remedial action in relation to the dumped radioactive waste material is not warranted. However, to avoid the possible inadvertent disturbance or recovery of the dumped objects and because the potential doses to the hypothetical group of military personnel patrolling the Novaya Zemlya fjords in which high level wastes have been dumped are not trivial, this conclusion depends upon the maintenance of some form of institutional control over access and activities in the vicinity of the fjords of Novaya Zemlya used as radioactive waste dump sites.

6. CONCLUSIONS

(1) Monitoring has shown that releases from identified dumped objects are small and localized to the immediate vicinity of the dumping sites. Overall, the levels of artificial radionuclides in the Kara and Barents Seas are low and the associated radiation doses are negligible when compared with those from natural sources.

Environmental measurements suggest that current annual individual doses from all artificial radionuclides in the Barents and Kara Seas are at most 1–20 μSv. The main contributors are global fallout from nuclear weapons testing, discharges from nuclear fuel reprocessing plants in western Europe and fallout from the Chernobyl nuclear accident. These doses can be compared with the worldwide average annual individual dose of 2400 μSv due to radiation from natural sources.

(2) Projected future doses to members of the public in typical local population groups arising from radioactive wastes dumped in the Kara Sea are very small. Projected future doses to a hypothetical group of military personnel patrolling the foreshores of the fjords in which wastes have been dumped are higher and comparable in magnitude to doses from natural sources.

These conclusions are drawn from a consideration of the high level solid waste which contains the vast majority of the dumped radionuclides. The radionuclide inventories of dumped waste objects were estimated on the basis of the design and operating histories of the nuclear reactors from which they were derived. The predicted future rates of radionuclide release to the environment from these sources were combined with mathematical models of radionuclide behaviour to calculate radiation doses to humans.

The predicted future maximum annual doses to typical local population groups are less than 1 μSv, while those to the hypothetical group of military personnel are higher, up to 4 mSv, but still of the same order as the average natural background dose.

(3) Doses to marine organisms are insignificant in the context of effects on populations.

These doses are delivered to only a small proportion of the population and, furthermore, are orders of magnitude below those at which detrimental effects on populations of marine organisms might be expected to occur.

(4) It is concluded that, on radiological grounds, remediation is not warranted. Controls on the occupation of beaches and the use of coastal marine resources and amenities in the fjords of Novaya Zemlya must, however, be maintained.

The condition is specified to take account of concerns regarding the possible inadvertent disturbance or recovery of high level waste objects and the radiological protection of the hypothetical group of individuals occupying the beaches adjacent to the fjords of Novaya Zemlya in which dumping has taken place. Efforts should be made to locate and identify all the high level waste objects whose locations are at present not known.

7. RECOMMENDATIONS

(1) Efforts should be made to locate and identify all high level waste objects.

(2) Institutional control should be maintained over access and activities in the terrestrial and marine environments in and around the fjords of Novaya Zemlya in which dumping has occurred.

(3) If, at some time in the future, it is proposed to terminate institutional control over areas in and around these fjords, a prior assessment should be made of doses to any new groups of individuals who may be potentially at risk.

(4) A limited environmental monitoring programme at the dump sites should be considered in order to detect any changes in the condition of the dumped high level wastes.

1. INTRODUCTION

In 1992, it was reported that the former USSR had, for over three decades, dumped radioactive wastes in the shallow waters of the Arctic Seas. This news caused widespread concern, especially in countries with Arctic coastlines. The IAEA responded by proposing an international study to assess the health and environmental implications of the dumping. The proposal was endorsed by the Fifteenth Consultative Meeting of the Contracting Parties to the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention 1972). The Consultative Meeting requested, in addition, that the study include consideration of possible remedial actions, e.g. the retrieval of the wastes for land storage.

The International Arctic Seas Assessment Project (IASAP) was launched by the IAEA in 1993 with the following objectives:

(1) To assess the risks to human health and to the environment associated with the radioactive wastes dumped in the Kara and Barents Seas; and
(2) To examine possible remedial actions related to the dumped wastes and to advise on whether they are necessary and justified.

The project was partially supported by extrabudgetary funding from the United States of America and was co-ordinated with the work of the Norwegian–Russian Expert Group for Investigation of Radioactive Contamination in the Northern Areas.

1.1. RADIOACTIVE WASTE DISPOSAL AT SEA AND THE INTERNATIONAL SYSTEM FOR ITS CONTROL

1.1.1. London Convention 1972

The first recorded sea disposal of radioactive wastes was carried out by the USA in 1946 at a site in the northeast Pacific Ocean, about 80 km off the coast of California [1]. In subsequent years, as sea disposal became increasingly widely used as a radioactive waste disposal option, the pressure for its control also increased.

The first United Nations Conference on the Law of the Sea, in 1958, recommended that the IAEA should assist States in controlling the discharge of radioactive materials into the sea, in promulgating standards and in drawing up internationally acceptable regulations to prevent pollution of the sea by radioactive materials. Accordingly, the IAEA set up successive scientific panels to provide specific guidance and recommendations relevant to the disposal of radioactive wastes at sea. The first IAEA experts meeting on the subject was held in 1957 and the first related publication was issued in 1961.

The Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter (London Convention 1972) was established in 1972 and entered into force in 1975. The Convention is recognized as the globally applicable legal instrument for the control of waste dumping at sea. It defines ‘dumping’ for its purposes as “any deliberate disposal at sea of wastes or other matter from vessels, aircraft, platforms or other man-made structures at sea”. Accordingly ‘dumping’ and ‘disposal at sea’ are used synonymously in this study.

From the beginning, the Convention prohibited, inter alia, the dumping at sea of high level radioactive wastes and required that low level radioactive wastes be dumped only provided that a special permit had been issued by the relevant national authority. The London Convention charged the IAEA with developing a definition of high level radioactive wastes which are unsuitable for disposal at sea and with preparing recommendations for national authorities on the issue of special permits for sea disposal of other radioactive wastes. In 1974, the IAEA published Provisional Definition and Recommendations as requested by the London Convention and has subsequently kept them under periodic review [2, 3]. The most recent Definition and Recommendations were published in IAEA Safety Series No. 78 [4].

In 1983, following the concerns of some Contracting Parties to the Convention over the possible health and environmental risks which might result from radioactive waste disposal operations, the Consultative Meeting of Contracting Parties to the Convention adopted a Resolution suspending radioactive waste dumping at sea pending a wide ranging review of the issue. The IAEA provided technical expertise for this discussion. In 1993, the Consultative Meeting of Contracting Parties to the Convention reached the decision to prohibit the sea dumping of all types of radioactive waste and amended the London Convention accordingly. This decision was reached mainly on social, moral and political grounds, without any new evidence of health risks. The Russian Federation did not accept the amendment to the Convention on the radioactive waste disposal at sea;

however, it declared that “Russia will continue its endeavours to ensure that the sea is not polluted by the dumping of wastes and other matter, the prevention of which is the object of the provisions contained in the above mentioned amendment ....”.

1.1.2. Dumping operations

A summary of the global dumping operations, which up to 1991 were officially reported to the Consultative Meeting of the Contracting Parties to the London Convention, was published by the IAEA in 1991 [1]. Of the total amount of radioactive material (46 PBq), more than 98% was disposed of in the northern part of the Atlantic Ocean, 92% thereof in the eastern basin, mainly in the OECD/NEA dumpsite (Fig. 1). The radioactive wastes noted in this summary were predominantly low level and originated in the application of radionuclides in research and medicine, the nuclear industry and military activities. They were packaged, usually in metal drums lined with a concrete or bitumen matrix. Unpackaged waste and liquid waste were also disposed of between 1950 and 1960, before the London Convention 1972 and associated IAEA Definition and Recommendations entered into force.

The inventory contained in Ref. [1] does not include the data supplied in the spring of 1993 by the Russian Federation, at the request of the Consultative Meeting of Contracting Parties to the London Convention, on the dumping activities of the former USSR and the Russian Federation. The ‘White Book of the President of Russia’ (provided to the London Convention as LC16/INF.2) indicates that high and low level radioactive waste was dumped in the Arctic Seas and the North-West Pacific during the period 1959 to 1992 and gave rough estimates of their activity content [5]. The items dumped included six nuclear submarine reactors and a shielding assembly from an icebreaker reactor containing spent fuel, totalling, as estimated, 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 (Tables I to III). The packaged and unpackaged solid waste and the nuclear reactors were dumped in the Kara Sea — in the shallow fjords of Novaya Zemlya, where the depths of the dumping sites range from 12 to 135 m and in the Novaya Zemlya Trough, at a depth of 380 m. The dump sites of the world’s oceans are depicted in Fig. 1 and the dump sites in the Kara and Barents Seas are shown in Fig. 2.

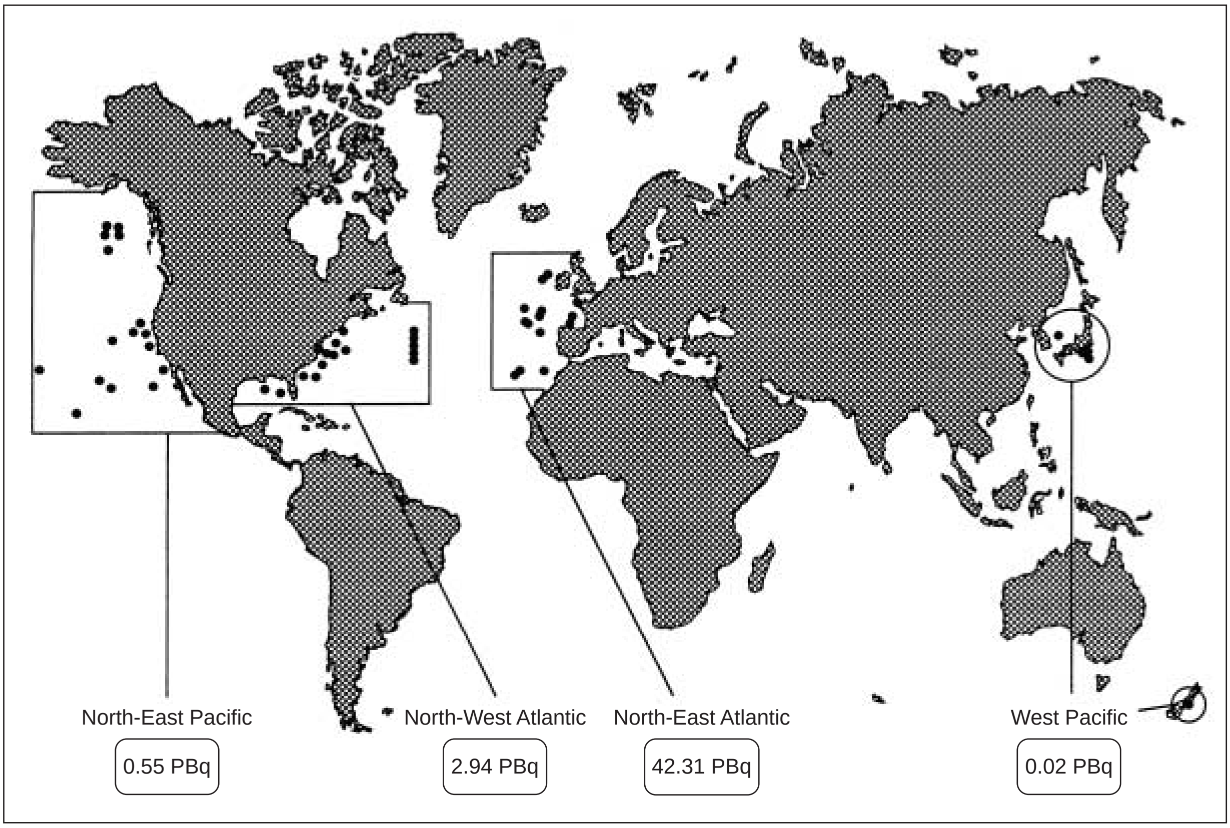


FIG. 1. Dumping sites for radioactive waste in the oceans, as officially reported up to 1991.

TABLE I. OBJECTS WITH SPENT NUCLEAR FUEL DUMPED IN THE ARCTIC SEAS, ACCORDING TO THE RUSSIAN 'WHITE BOOK' [5]

Object	Co-ordinates, year	Depth (m)	Total activity (max.) (kCi (PBq)[a])	Radionuclide content	Description of protective barriers
Compartment of NS[b] Number 285 with two reactors, one containing SNF[c]	71°56′2″ N, 55°18′5″ E, Abrosimov Fjord, 1965	20	800 (29.6)	Fission products	Reactor compartment and interior structures filled with Furfurol(F) mixture
Compartment of NS Number 901 with two reactors containing SNF	71°56′2″ N, 55°18′9″ E, Abrosimov Fjord, 1965	20	400 (14.8)	Fission products	Reactor compartment and interior structures filled with Furfurol(F) mixture
Shielding assembly of reactor from nuclear icebreaker Lenin with residual SNF (60% of original UO_2 fuel charge)	74°22′1″ N, 58°42′2″ E, Tsivolka Fjord, 1967	49	100 (3.7)	^{137}Cs (~50 kCi) (~1.85 PBq), ^{90}Sr (~50 kCi) (~1.85 PBq), ^{238}Pu, ^{241}Am, ^{244}Cm (~2 kCi) (~0.074 PBq)	SNF residue bound by Furfurol(F) mixture, shielding assembly placed in reinforced concrete container and metal shell
Reactor of NS Number 421 containing SNF	72°40′ N, 58°10′ E, Novaya Zemlya Trough, 1972	300	800 (29.6)	Fission products	Metal container with lead shell dumped along with barge
NS Number 601 with two reactors containing SNF	72°31′15″ N, 55°30′15″ E, Stepovoy Fjord, 1981	50	200 (7.4)	Fission products	Reactor compartment and interior structures filled with Furfurol(F) mixture
Total: five objects with seven reactors containing SNF	1965–1981		2300 (85.1)		

[a] Expert estimates were made at the time of dumping, on the basis of the power generated by NS reactors (12.5 GW·d).
[b] NS = nuclear submarine.
[c] SNF = spent nuclear fuel.

The White Book estimated the total activity of fission and activation products and actinides in the high level wastes dumped in the Kara Seas as 89 PBq at the time of disposal. A re-evaluation by IASAP resulted in a revised estimate of 37 PBq for the total activity at the time of disposal. The inventory had fallen to 4.7 PBq in 1994, solely as a result of radioactive decay (Section 4). Fission products constitute 86%, activation products 12% and actinides 2% of the current inventory. The single object representing the largest proportion (47%) of the 1994 inventory is the container of spent fuel from the icebreaker Lenin.

According to the White Book, more than 6000 containers of solid low level radioactive wastes have been disposed of in the Kara and Barents Seas. They were dumped either individually or on barges. The packaged wastes comprised mainly film coverings, tools, personal protective devices, filters and other contaminated objects produced during maintenance work. In addition, the White Book reports the dumping of more than 150 large objects such as steam generators and reactor lids. The activity of these solid intermediate and low level wastes was estimated in the White Book to be about 0.6 PBq at the time of dumping (Table III).

TABLE II. OBJECTS WITHOUT SPENT FUEL DUMPED IN THE ARCTIC SEAS, ACCORDING TO THE RUSSIAN ‘WHITE BOOK’ [5]

Object	Co-ordinates, year	Depth (m)	Total activity (kCi (PBq)[a])	Radionuclide content	Description of protective barriers
Reactor compartment of NS[b] Number 285 with two reactors, one without SNF[c]	71°56′2″ N, 55°18′5″ E, Abrosimov Fjord, 1965	20	Requires special analysis	Unclear	Reactor compartment structures
Reactor compartment of NS Number 254 (with two reactors)	71°55′13″ N, 55°32′32″ E, Abrosimov Fjord, 1965	20	Requires special analysis	Unclear	Reactor compartment structures
Reactor compartment of NS Number 260 (with two reactors)	72°56′2″ N, 55°18′5″ E, Abrosimov Fjord, 1966	20	Requires special analysis	Unclear	Reactor compartment structures
Steam generating installation of icebreaker Lenin, comprising three reactors with primary loop pipelines and watertight stock equipment	74°26′4″ N, 58°37′3″ E, Tsivolka Fjord, 1967	50	~50 kCi (~1.9 PBq)[a]	Mainly ^{60}Co	Biological shielding unit (B-300 steel + concrete)
Two reactors from NS Number 538	73°59′ N, 66°18′ E, Techeniye Fjord, 1972	35–40	Requires special analysis	Unclear	Metal container with lead shell
Total: five objects with ten reactors without SNF	1965–1988		Requires special analysis (possibly up to 100 kCi (3.7 PBq) at time of dumping)		

[a] Expert estimates were made at the time of sinking, on the basis of the power generated by NS reactors (12.5 GW·d).
[b] NS = nuclear submarine.
[c] SNF = spent nuclear fuel.

Low level liquid radioactive wastes were also dumped in the Barents and Kara Seas during the period fo 1960 to 1993 [5]. Wastes with a total activity of 0.45 PBq were dumped in the Barents Sea in five designated areas and further 0.43 PBq outside the designated areas, including 0.32 PBq in the Kara Sea in 1973, 0.07 PBq in Andrayev Fjord in 1982 and 0.07 PBq in Ara Fjord in 1989 (Fig. 2). However, the total amount of liquid radioactive waste dumped in these Russian Arctic seas (0.88 PBq) is considerably smaller than the total activity of solid waste dumped in the area. It is also a small proportion (~5%) of the activity entering the Barents and Kara Seas from global fallout (6 PBq of ^{137}Cs and ^{90}Sr) and from Sellafield fuel reprocessing plant discharges (10–15 PBq ^{137}Cs) during the same period [6].

The activity of waste dumped in the North-East Pacific estimated in the White Book [5] was 0.7 PBq. No spent nuclear fuel was dumped at sea in this latter area.

1.1.3. Dumping in the Arctic in relation to the London Convention 1972

Until its amendment in 1993, the London Convention 1972 prohibited the disposal at sea of high level radioactive waste as defined by the IAEA, but allowed, under special permit, the dumping of other types of radioactive waste, in accordance with

TABLE III. LOW AND INTERMEDIATE LEVEL SOLID RADIOACTIVE WASTE DUMPED IN THE KARA AND BARENTS SEAS, ACCORDING TO THE RUSSIAN ‘WHITE BOOK’ [5]

Area	Activity (Ci)	Activity (TBq)	Number of dumpings	Years	Remarks
Novaya Zemlya Trough	3320	123	22	1967–1991	3174 C[a,b], 9 LO[c], 8 V[d]
Sedov Fjord	3410	126	8	1982–1984	1108 C[a], 104 LO
Oga Fjord	2027	75	8	1968–1983	472 C[a], 4 LO, 1 V
Tsivolka Fjord	2684	99	8	1964–1978	1600 C[a], 6 LO, 1 V
Stepovoy Fjord	1280	47	5	1968–1975	5 LO
Abrosimov Fjord	661	25	7	1966–1981	8 C[a], 7 LO, 4 V
Blagopoluchiye Fjord	235	8	1	1972	1 LO
Techeniye Fjord	1845	68	3	1982–1988	146 C[a], 18 LO, 1 V
Off Kolguyev Island	40	1.5	1	1978	1 V
Zornaya Bay (Novaya Zemlya)	300	11	1	1991	1 LO
Barents Sea	>100	>3.7	1	Unknown	Barge with solid waste in welded hold
Total	16 000	590	65		6508 C[a], 155 LO, 17 V

[a] Number of recorded containers; the actual number can be higher.
[b] C = container.
[c] LO = large object.
[d] V = vessel.

recommendations prepared by the Agency. Much of the material dumped in the Kara Sea falls into the category of high level waste. It should be noted, however, that most of the spent fuel was dumped in the years before the 1972 adoption of the London Convention and its entry into force in 1975. The USSR became a Contracting Party to the Convention in 1976. In the years following the 1983 agreement to suspend dumping at sea, dumping of low level and some high level radioactive waste was carried out in the Arctic Seas.

The Arctic dumping sites are not in conformity with the specifications for dumping sites provided by the IAEA in its recommendations [2–4], particularly as many of them are in shallow waters. Nevertheless, it should be noted that, before the entry into force of the London Convention 1972, several dumping operations had been carried out in shallow coastal waters in other parts of the world [1].

1.2. RATIONALE FOR ESTABLISHING IASAP

When information on dumping practices in the Arctic Seas was revealed, most of the technical and environmental data needed for the proper evaluation of hazards to human health and the environment resulting from the dumped wastes were not generally available. The first joint Norwegian–Russian exploratory cruise in summer 1992 was not able to take samples in the immediate vicinity of the dumped wastes. However, samples taken in the Kara Sea showed that present levels of radioactive contamination in that area were lower than, or similar to, those in other sea areas. This resulted in the preliminary conclusion that the impact of the dumped wastes on the overall level of radioactive contamination in the Kara Sea was insignificant [7].

However, it was understood that gradual deterioration of the waste packages and containments could lead to impacts in the future. These could result in contamination of the marine food chain, possibly with additional radiation exposure of humans through the consumption of fish and other marine foodstuffs as a consequence. Since the wastes are lying in shallow waters, the possibility of radiation exposure through other routes, such as the movement and transport of the waste packages by natural events (ice or storm action), or by accidental or deliberate human intrusion, cannot be ruled out. In order to provide more information on these issues it was deemed necessary to evaluate the condition of the waste objects, existing and potential radionuclide releases, the transport and fate of released radionuclides and associated radiological exposures. The International Arctic Seas Assessment Project (IASAP) was established as

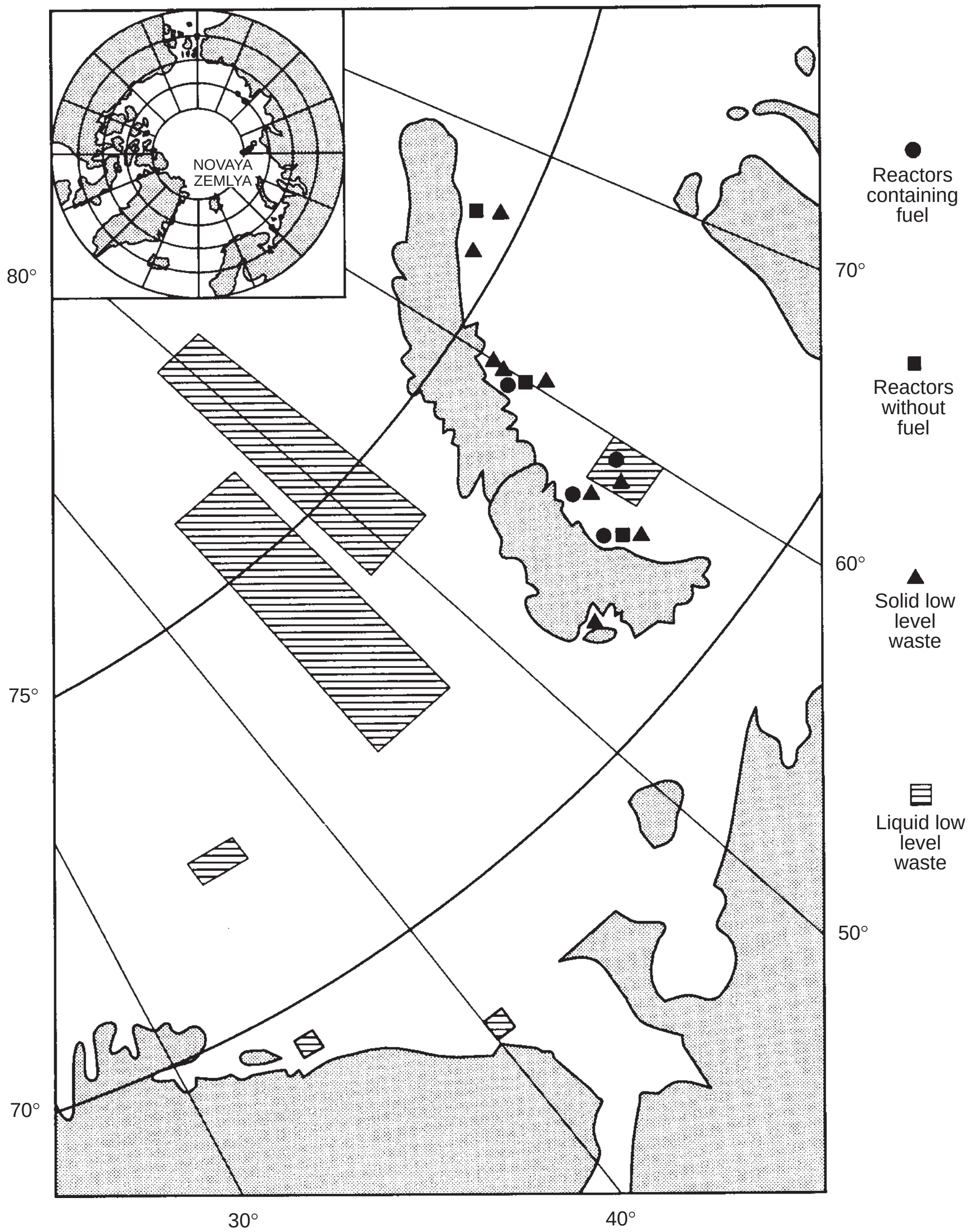


FIG. 2. Dumping sites for radioactive waste in the Arctic Seas.

mentioned earlier to answer these and other related questions. The multidisciplinary team of scientists adopted the following approach:

- It examined the current radiological situation in Arctic waters to assess evidence for releases from the dumped waste.
- It predicted potential future releases from the dumped wastes concentrating on the solid high level waste objects containing the major part of the radionuclide inventory of the wastes.
- It modelled environmental transport of released nuclides and assessed the associated radiological impact on humans and the biota.
- It examined the feasibility, costs and benefits of possible remedial measures applied to a selected high level waste object.

The work was carried out in 1993 to 1996 by using standard IAEA mechanisms, i.e. consultants andadvisory group meetings, establishment of a co-ordinated research programme and technical contracts.

2. RADIOLOGICAL PROTECTION AND DECISION MAKING

2.1. BASIC CONCEPTS IN RADIOLOGICAL PROTECTION RELEVANT TO IASAP

This section outlines the central concepts of the System of Radiological Protection recommended by the ICRP [8] and incorporated into the International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources (BSS) published the IAEA [9]. Particular emphasis is given to concepts relevant to the International Arctic Seas Assessment Project (IASAP).

Both ICRP's System of Radiological Protection and the BSS make a distinction between *practices* and *interventions*. Human activities that either add radiation exposure to that which people normally incur owing to background radiation, or that increase the likelihood of their incurring radiation exposure, are designated as *practices*. Any action intended to reduce or avert exposure or the likelihood of exposure to sources which are not part of a controlled practice, or which are out of control as a consequence of an accident, are *interventions*.

2.1.1. Practices

For a practice, provisions for radiation protection and safety can be made before its commencement, and the associated radiation exposures and their likelihood can be restricted from the outset. Practices for which the BSS include activities involving the production of radiation sources, the use of radiation and radioactive substances in a variety of applications and the generation of nuclear power extending throughout the nuclear fuel cycle (from mining and processing of radioactive ores through the operation of nuclear reactors and fuel cycle facilities to the management of radioactive wastes) require prior notification and authorization by registration or licensing.

2.1.1.1. Exclusion, exemption and clearance

There are certain exceptions to the administrative requirements outlined in the BSS. All materials are radioactive to some extent because they contain natural radionuclides. They may also be contaminated with residues from past practices. The nature of some of these materials and other sources of radiation exposure is such that control is not practicable or even not possible. Exposures associated with potassium-40 naturally present in the human body, cosmic radiation at ground level and unmodified concentrations of radionuclides in most natural and raw materials are examples of such exposures that are not amenable to control. In other words, any exposure whose magnitude or likelihood is essentially unamenable to control through regulation is deemed *excluded* from control.

Practices, and sources within a practice, may be *exempted* from *entering* the requirements of the standards provided that:

(a) the radiation risks to individuals caused by the exempted practice or source are sufficiently low (of the order of 10 μSv or less per year) as to be of no regulatory concern;
(b) the collective radiological impact of the exempted practice or source is sufficiently low (no more than about 1 man·Sv committed by one year's performance or if an assessment shows that exemption is the optimum option) as not to warrant regulatory control under the prevailing circumstances; and
(c) the exempted practices and sources are inherently safe, with no appreciable likelihood of events occurring that could lead to a failure to meet the criteria in (a) and (b) above.

Sources, including substances, materials and objects, within notified and authorized practices may be *released* from the requirements of the regulatory instruments subject to compliance with *clearance* levels which take account of the above exemption criteria.

2.1.2. Intervention

If the circumstances giving rise to exposure, or the likelihood of exposure, already exist, the only type of action tenable is intervention in which the reduction of dose/risk is achieved by remedial or protective actions. Situations that may require intervention include: chronic exposures to naturally occurring sources of radiation, such as radon in dwellings; exposure to radioactive residues from past activities and events; and emergency exposure situations, such as those resulting from accidents or from deficiencies in existing installations.

Intervention is most efficient if it can be applied at source. However, in many situations, this is not possible and intervention has to be applied within the environment or to the habits of individuals, thereby restricting their freedom of action.

Decisions to apply countermeasures after an accident or remedial actions in chronic exposure situations should not be taken lightly, particularly where they restrict people's freedom of action or choice, impose costs on society or may cause direct harm and disruption to people — either the public or the persons implementing the protective measures.

2.2. BASIC REQUIREMENTS OF RADIOLOGICAL PROTECTION AS ESTABLISHED IN THE BASIC SAFETY STANDARDS

The basic principles of radiological protection applicable to practices and intervention situations, as outlined above, are provided as follows:

2.2.1. Practices

A practice that entails, or that could entail, exposure to radiation shall only be adopted if it yields sufficient benefit to the exposed individuals or to society to outweigh the radiation detriment it causes or could cause, *i.e. the practice shall be justified*.

Individual doses due to the combination of all relevant practices should not exceed specified *dose limits*.

Radiation sources and installations should be provided with the best available protection and safety measures under the prevailing circumstances, so that the magnitudes and likelihood of exposures and the number of individuals exposed are as low as reasonably achievable and economic and social factors are taken into account and the doses they deliver and the risks they entail should be constrained, *i.e. protection and safety shall be optimized*.

2.2.2. Intervention

Radiation exposures due to sources of radiation that are not part of a practice should be reduced by *intervention* when this is justified, and the intervention measures should be optimized. The introduction of any particular protective action entails some risk to the individuals affected and some harm to society in terms of financial costs or social and economic disruption. Therefore, before undertaking a protective action, it should be shown that it can produce a positive net benefit. In other words, implementation of a given protective action will be *justified* if its benefits, which include radiation detriment averted, are greater than its associated detriments, which include doses to workers undertaking the protective actions, non-radiological risks, financial costs and other less readily quantifiable consequences associated with the protective action, such as social disruption. Public anxiety, which can be either relieved or increased by a protective action, is a further factor to be considered.

For protective actions that are justified, it is necessary to establish the level at which the best protection will be provided. In other words, the radiation detriment averted by each protective action should be balanced against its costs and detriment in such a way that the net benefit achieved by the protective action is maximized, *i.e. protection is optimized*.

In intervention, *dose limits for members of the public do not apply*. They are intended for use in the control of practices. The use of *dose limits*, or any other predetermined dose levels, as a basis for deciding on intervention, might involve measures that would be out of all proportion to the benefit obtained and would then conflict with the principle of justification [8]. The only exception to the use of a predetermined limit is in cases where the dose approaches a level likely to cause serious deterministic effects.

2.3. PREVIOUS SEA DUMPING IN THE ARCTIC — A PRACTICE OR AN INTERVENTION SITUATION?

Section 3 describes the levels of radionuclides in the Arctic marine environment derived from a variety of sources, to which the system of radiological protection has varying applicability. Predominantly, the radionuclides are of natural origin, without any human enhancement. Natural sources are unamenable to control, and these radionuclides are excluded from regulation.

Additionally, there are residues from nuclear weapons tests, residues from previously authorized releases (some of which would not fulfil contemporary requirements), leakages from dumped objects, and residues from past accidents. Together these constitute a chronic exposure situation. The concept relevant to the dispersed radionuclides is intervention in the environment or restricting the freedom of people to act, provided that it is justified.

This publication focuses on the sources dumped at sea, the physical conditions of which vary. They can be seen as constituting a *potential chronic exposure situation*. Depending on their physical condition, intervention (remediation) at source is the most viable course of action, rather than intervention in the environmental exposure pathways or restricting the freedom to act of persons. As already stated, the precondition for *intervention at source* is that it be both justified and optimized.

There are currently no internationally agreed criteria (action levels) covering intervention (remediation) in chronic exposure situations, except for that due to radon in dwellings. Such criteria are, however, under development by the ICRP. The IAEA is at present developing intervention criteria for cleanup in land areas. Thus, in the case of sources dumped at sea giving rise to potential exposures, the guiding principles are justification and optimization of the actions as far as the public is concerned.

Justification of remedial actions encompasses considerations beyond the scope of radiological protection. Indeed, the estimated radiological detriment may represent only a small proportion of the considerations leading to a decision regarding justification. This will be discussed in Section 2.4. Optimization of the protection needs to be considered in combination with justification in order to ensure that the method yielding the maximum net benefit is evaluated.

For workers carrying out the remedial actions, the principles applicable to practices, including dose limits, apply. This is because remedial actions can be planned in advance and the associated doses/risks properly controlled.

2.4. MAKING DECISIONS REGARDING REMEDIAL ACTIONS

Without going into detail regarding the requirements of the London Convention 1972, it is worth stressing that a Contracting Party is responsible for all dumping activities carried out in its territorial seas, continental shelf and exclusive economic zone. The responsibility for deciding on remedial actions relating to abandoned or dumped radioactive sources, in cases where the source resides in the area of jurisdiction of the State from which the material originated, lies with that State. This is the case for sources previously dumped by the former USSR and the Russian Federation in the Kara and Barents Seas.

This means that final decisions regarding remedial actions will be made by the relevant Russian authorities. Pursuant to the request from the Contracting Parties to the London Convention 1972, the IAEA limits its advice to the necessity and justification of remedial actions from the radiation safety viewpoint, i.e. the risk to human health and to the environment associated with radiation exposures. Nevertheless, for contextual purposes, an outline is provided below of the entire decision process for intervening with radioactive wastes such as those contemplated by this project.

As pointed out in Section 2.3, the justification of remedial actions may involve broader considerations than the health risk associated with radiation. Psychological factors and broader socioeconomic and political factors may also have to be considered in the justification process. This implies that the decision maker is faced with a complex multifactorial problem.

2.4.1. Decision aiding techniques

There are several decision aiding techniques that may prove helpful in such circumstances. These enable the decision maker to gain greater insight into the nature of the problem, to clarify the decision making process and to make the basis for the decision transparent. The available techniques differ in their ability to deal with reasonably quantifiable factors (such as avertable doses and financial costs), and non-quantifiable factors (e.g. many psychological, social and political factors). The technique selected must be capable of handling, within a common framework, all relevant quantifiable and non-quantifiable factors. This means that simple cost–benefit analyses are inadequate and other decision aiding techniques are required. The previously abstract mathematical discipline of decision theory has been developed into a potentially valuable technique known as decision analysis. This technology enables decision makers to handle large and complex problems and the attendant information. Decision analysis is not intended to solve problems directly; its purpose is to provide insight and bring about a better understanding of the nature of the problem, thereby facilitating informed decisions [10–12].

In a decision process, identification of the multidimensional values is essential. These values, expressed as *objectives, criteria or performance measures*, are vital to a well structured approach in dealing with the problem. Identifying values and assessing their relative importance in the decision provide better alternatives from which to choose.

Since environmental values are multidimensional, different kinds of objective will necessarily be involved in any decision dealing with environmental and health issues. Some of these objectives might be directly measured on a numerical scale, and some can be subdivided into subobjectives that are measurable in quantitative terms. This kind of numerical variable is called an *attribute* and it is used to measure performance of actions in relation to an objective. The assessment of some attributes is simple because the representative variables can be identified and measured quantitatively. However, for some attributes such as reassurance, it is more difficult to find proxy attributes or variables that can be quantified. For unquantifiable attributes, for example, *direct rating* can be used. In this technique, the most

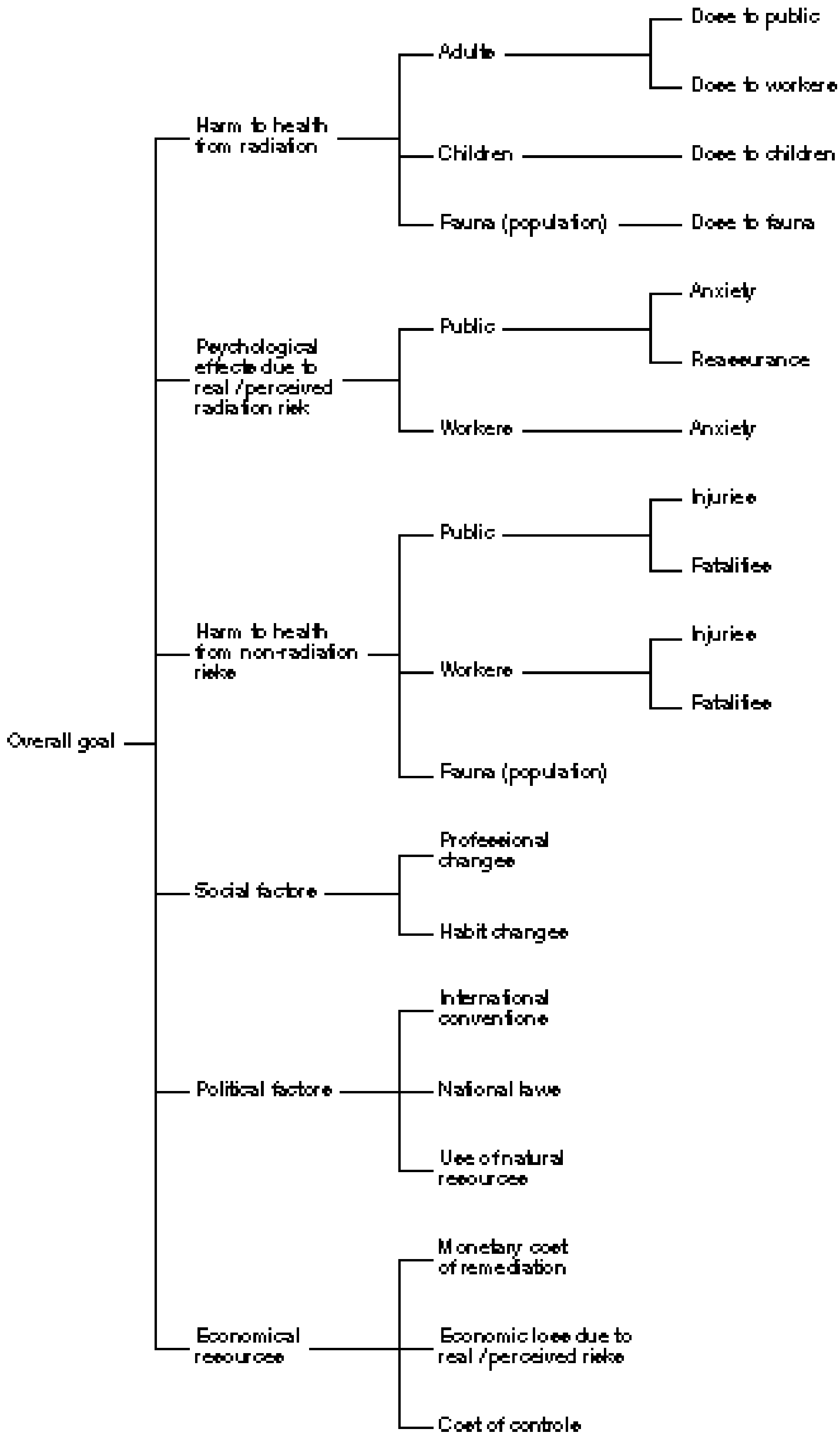


FIG. 3. Value tree to be considered for each remedial action strategy, including non-action.

preferred option is given a value of 100 and the least preferred option the value of zero. Other options are assigned values between zero and 100, according to relative preference.

Before one can combine values for different attributes in order to obtain a view of the overall benefits offered by each strategy one has to assess the weights of the attributes. The weights represent the judgement of the decision maker on the relative importance of the levels of attributes. When assessing trade-off values between the attributes, it should be noted that the importance of an attribute is dependent not only on its conceptual value, such as health, but also on its *length of scale*, such as the number of cancer cases. Such scaling constants can be obtained, for example, by an assessment method called *swing weighting*. In this method the decision maker compares a sequence of pairs of hypothetical actions, which differ only in their values along two attribute scales, until an indifferent pair of options is found.

Finally, the values can be aggregated to determine how well each strategy performs overall. If an additive model is used, the weighted attribute values associated with each strategy are added together to obtain the benefit. The strategies can then be ranked according to the benefit scores. This is a simplified description of decision analysis. More detailed information on this topic can be found in Refs [10–12].

An attribute hierarchy (value tree) can be useful in defining attributes and objectives. Figure 3 portrays one such value tree for comparing different remedial action strategies for dumped waste in Arctic Sea areas. Assessments need to be carried out both for the cases of successful remediation and for accidents occurring during implementation (various scenarios). Equally, the non-action strategy for each waste type needs to be considered. The objectives and associated attributes depicted in Fig. 3 are as follows:

- The reduced harm to health from radiation, which involves consideration of cancers, hereditary effects and possible deterministic effects for humans and effects to fauna populations, can be measured by reduction of relevant potential radiation doses to the public and to fauna, as will be discussed in Section 5. For workers, only the collective dose is relevant here because the dose limits are applied to the protection of workers; however, accidental doses to individuals need to be considered in order to deal with deterministic effects.
- Psychological effects may cause harm to health, as a consequence of real and perceived risks associated with potential exposures and/or contamination. Harm to health derived from psychological effects may become comparable with health effects directly associated with radiation, as was observed after the Chernobyl accident. Both remedial actions and non-action can give rise to anxiety among the population. It should also be noted that remedial action may transfer anxiety from one population group to another, if the sources are moved. Remedial actions may also result in stress reactions in workers, who implement the actions (direct rating or other methods can be used to quantify the attribute).
- Reassurance to the population may be offered by implementation of remedial actions (through direct rating or other methods).
- There is a probability of non-radiation harm to health of the workers implementing remedial actions. Successful implementation of remedial actions and accident scenarios may differ considerably in the consequences for workers (injuries, fatalities). Also, the non-radiation harm to the public and the fauna needs to be assessed.
- Social disruption may be caused by the absence of remedial actions or, conversely, by their implementation. For example, whether indigenous groups or craftsmen need to change their working or living habits must be taken into account (by direct rating or other methods).
- Political factors will probably have to be taken into account. International conventions and national laws may have a bearing on decisions — for example, a portion of the dumping operations of the former USSR and the Russian Federation occurred after the USSR had become a Contracting Party to the London Convention in 1976, and some of these dumping activities contravened the provisions of the Convention. In addition, environmental disruptions such as prejudice to marine resources and amenities may need to be included in this evaluation (direct rating or other methods).
- Regarding the resources and financial costs involved, the following costs need to be considered: costs of remedial actions, possible economic losses associated with real and perceived risks, restrictions on the use of natural resources and amenities and cost of regulatory controls (quantified in monetary units).

The various remedial strategies can be defined by combining the different types of dumped sources and feasible remediation options (e.g. capping or recovery to land disposal). The values for the attributes (see Fig. 3) for each strategy can then be assessed, weighted and combined to obtain the overall scores for ranking the remedial strategies. In this way, remedial strategies which offer a positive net benefit (justified strategies) and a strategy that offers the maximum net benefit (optimized strategy) can be defined. The strength of the choice of a strategy can be studied by sensitivity analysis.

3. THE ARCTIC ENVIRONMENT

3.1. DESCRIPTION OF THE REGION

3.1.1. Oceanography

3.1.1.1. Arctic Ocean

The Arctic Ocean (Fig. 4) is nearly landlocked and is divided into two major basins separated by the Lomonosov Ridge: the Eurasian Basin and the Canadian Basin. Both the Eurasian Basin, with maximum depths of around 4000 m, and the Canadian Basin, with maximum depths of about 5000 m, are further subdivided by lesser submarine ridges. Another important topographical feature is the continental shelf, underlying about 30% of the area of the Arctic Ocean and comprising several shallow marginal seas. The shelf bordering Eurasia extends out to 500–1700 km, while along the Alaskan and Greenland coastlines its width does not exceed 200 km. The total area of the Arctic Ocean is 9.5×10^6 km^2 and its volume is 1.7×10^7 km^3. The most important connection of the Arctic Ocean with the rest of the world's oceans is through the 2600 m deep Fram Strait between Greenland and Svalbard. Shallower openings in the land contiguous to the Arctic Ocean connect it to the Pacific Ocean through the Bering Strait and to the Atlantic Ocean through the Canadian Archipelago.

Three main water masses can be defined in the Arctic Ocean: surface water, Atlantic water and bottom water [13, 14]. The surface water layer originates in the marginal shelf seas and derives from local mixing of waters below the marine ice cover following the freeze/melt cycle. It is the most variable of the water masses, its properties being influenced by the seasonality of ice formation and shelf/river water input. It can be divided into the surface Polar Mixed Layer, 30 to 50 m deep, containing fresh, cold water (–1.4 to –1.7°C), and

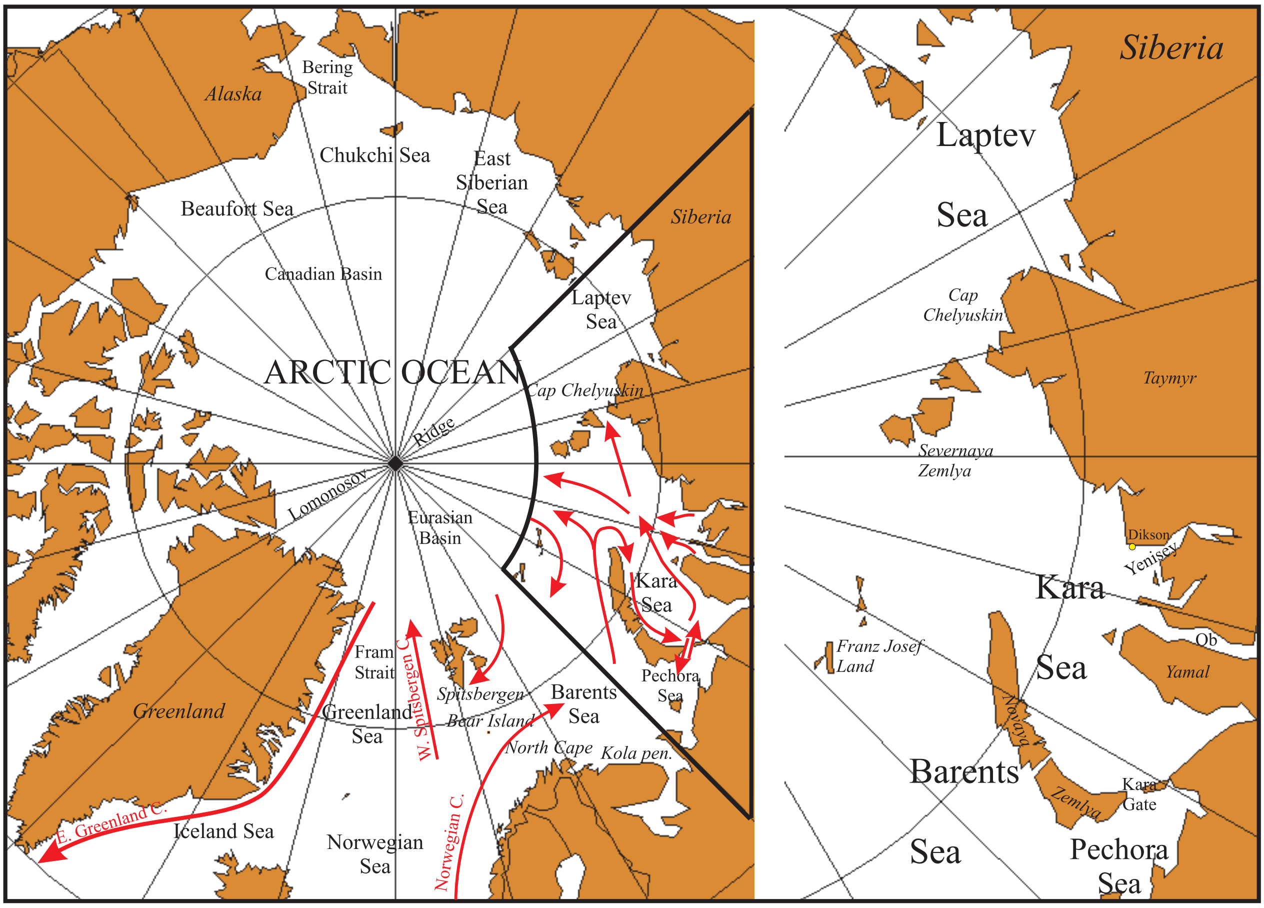


FIG. 4. Arctic Ocean with main currents.

TABLE IV. MAIN WATER FLOWS OF THE ARCTIC OCEAN [15]

From	To	Through	Flow (sverdrup)[a]
Atlantic Ocean	Arctic Ocean	Fram Strait	4
Atlantic Ocean	Barents Sea	Bear Island — North Cape	3.1
Arctic Ocean	Atlantic Ocean	Fram Strait	5
Arctic Ocean	Atlantic Ocean	Canadian Archipelago	2.1
Arctic Ocean	Barents Sea	Spitsbergen — Franz Josef Land	0.5
Pacific Ocean	Arctic Ocean	Bering Strait	0.8–0.9
White Sea	Barents Sea	Gorlo Strait	0.015
Barents Sea	Atlantic Ocean	Spitsbergen — Bear Island	1.2
Barents Sea	Arctic Ocean	Franz Josef Land — Sv. Anna Trough	1.2
Barents Sea	Kara Sea	Franz Josef Land — Novaya Zemlya	0.15–0.54
Barents Sea	Kara Sea	Kara Gate	0.04–0.6
Kara Sea	Arctic Ocean	West of Severnaya Zemlya	0.6–0.7
Kara Sea	Laptev Sea	Severnaya Zemlya — Mainland	0.16–0.3

[a] 1 sverdrup = 10^6 m^3/s.

the underlying halocline, down to about 200 m, with increasing salinity and temperatures up to 0°C. The warm and salty Atlantic water layer extends below the pycnocline down to depths of 800–900 m and is characterized by a mid-depth temperature maximum ranging between 0.5 and 4°C. This water mass is generated from the North Atlantic waters inflowing through the Fram Strait and over the Barents and Kara Sea shelves. The deep waters, extending below the Atlantic layer to the ocean floor, have a relatively high and uniform salinity (34.93 to 34.99‰) and temperature (–0.7 to –0.8°C in the Eurasian Basin and –0.3 to –0.4°C in the Canadian Basin). Besides the three main water masses described, there is a thin intermediary layer of Pacific water underlying the surface waters in the Canadian Basin.

The circulation in the Central Arctic Ocean is increasingly well understood (Fig. 4). The flow pattern in the Polar Mixed Layer is closely related to that of the overlying sea ice. The prominent long term features are the Beaufort Gyre in the Canadian Basin and the Transpolar Drift Stream, directed from the Pole towards the Fram Strait. Circulation in the halocline is driven by the anticyclonic flows on the shelves. The flow in the Atlantic layer is generally cyclonic. It appears to occur in boundary currents, is influenced by processes on the shelves (mainly those of the Barents and Kara Seas) and may be steered by topographical features. Most of the Atlantic water enters the Arctic Ocean through the Fram Strait in the West Spitsbergen Current and exits it via the East Greenland Current. A branch of the Norwegian Atlantic Current also brings Atlantic water into the Arctic Basin over the Barents and Northern Kara Sea shelves. At depth, the Central Arctic Basins are decoupled from extensive exchange. There, cyclonic flows appear to exist around the boundaries of the Canadian and Eurasian Basins. A summary of the main inflows and outflows of the Arctic Ocean is given in Table IV [15], which is the basis for the compartment models used in Section 5.

The Central Arctic is permanently covered by sea ice, the exchange with the atmosphere being thus limited to polynias (spaces of open water in the ice pack) and leads. This, together with the sharp pycnocline at about 200 m, almost completely isolates the deep ocean from non-particulate vertical exchange with the upper waters [14]. In the Polar Mixed Layer, brine release leads to limited vertical mixing. The halocline is a permanent, advective feature and never appears to be penetrated. Its waters are formed during winter freezing on the marginal shelves and advected into the Central Arctic Basins, sinking underneath the Polar Mixed Layer owing to their higher density. The salty Atlantic waters get cool enough in reaching Fram Strait to sink beneath the Arctic upper waters. For the deep waters of the Arctic Basin, vertical mixing is restricted to near boundary convection, which may ventilate water masses down to the bottom of the ocean. Waters entering or exiting the Arctic Ocean through Fram Strait are affected by open ocean convection which reaches down to the bottom in the Greenland Gyre and to mid-depths in the Iceland Sea.

Residence times of 2.5–3.5 a have been derived for fresh water on the Barents and Kara Sea shelves [14]. The mean residence time of fresh water in the upper layers of the Arctic Basin is 10 a, possibly 2–6 a shorter for surface waters than for the halocline. Conservative tracers transported from the shelves to the upper layers

FIG 5. Mean annual runoff to the Arctic Ocean (km^3/a) (drawn up from Ref. [16]).

of the Arctic Basin could therefore exit through Fram Strait in about 5 a. Renewal times of tens of years, 50–100 a and 250–300 a have been estimated for the intermediate, deep and bottom waters of the Arctic Basin, respectively.

3.1.1.2. Barents Sea

The Barents Sea (Fig. 4), with an area of 1 424 000 km^2 and a volume of 316 000 km^3, lies mainly on the northern European continental shelf. It has a complicated bottom topography, which contributes to the complexity of distributions of water masses and sedimentation regimes. For the most part, its depth varies between 100–350 m and only drops to below 500 m over 1% of the sea's area, with a maximum of about 600 m attained at the boundary with the Norwegian Sea.

The Barents Sea has a very unstable climate [15]. The atmospheric Arctic front forms at the boundary between the predominantly temperate air masses in the south and the cold Arctic air masses in the north. Considerable changes occur across the Barents Sea in the mean air temperature (in winter 0°C in the south-west, –10°C in the south-east and –24°C at Spitsbergen; in summer 9°C in the south and 0°C in the north), precipitation (annual average values of 300 mm in the north-east and up to 1000 mm in the south-west) and dominant winds.

Besides its main connections to the Atlantic and central Arctic, the Barents Sea also communicates with the Kara Sea and the White Sea. It receives runoff from rivers on the mainland (Fig. 5) either directly, notably the Pechora River (about 134 km^3/a), or through the White Sea, mainly the Severnaya Dvina, Mezen and Onega (altogether 136 km^3/a) [16]. A number of smaller rivers flow into the Barents Sea from the Scandinavian and Kola coasts, amounting to about 10% of the total river runoff to the Barents Sea. Rivers from Spitsbergen, Franz Josef Land and Novaya Zemlya add about 36, 3.7 and 33 km^3/a, respectively. Snow or ice melting on the mainland and the islands also contributes to the fresh water input. Runoff significantly affects only the south-eastern part of the sea.

Four principal water masses can be identified in the Barents Sea [17]. The Atlantic water, warm (4–12°C) and saline (about 35‰), incoming from the west but also from the north, through the Arctic basin, is located in the western part of the sea. The surface, cold (<0 to 1°C), less saline (33–34‰) Arctic water flows in from the north. The relatively fresh (28–34.5‰) coastal water, with significant seasonal temperature and salinity variation, extends along the mainland and western Novaya Zemlya coasts. The Barents Sea waters, forming as a result of mixing between the previously described water masses and transformation under the effect of local conditions, reside mainly in the eastern and northern parts of the sea. A characteristic feature is the polar front forming at the boundary between cold Arctic and warm Atlantic waters, with temperature and salinity gradients reaching maxima of 1.3°C/km and 0.2‰/km, respectively. It strongly influences water circulation and affects chemical and biological conditions in its vicinity, leading to increased contents of biogenic elements and a general abundance of life.

The general water circulation in the Barents Sea is cyclonic. The strongest and most stable flow runs eastwards along the northern coast of Norway. It is formed from waters of the Norwegian Atlantic Current and of the Norwegian Coastal Current [18] entering the Barents Sea south of Bear Island and is called the North Cape Current. Further east it bears the name of Murmansk Current, and at about 30° E separates into a coastal branch and a northward flowing branch, part of which recirculates westwards. The Barents Sea also features a number of cold currents flowing westwards and southwards. Important from the point of view of possible dispersion of contaminants from the Kara Sea is the Litke Current, coming into the Barents Sea through the northern part of the Kara Gate. Its intensity is not well known, but appears to be a factor of at least 7–10 times weaker than the current flowing eastward through the Kara Gate [19]. The principal water inflows and outflows of the Barents Sea are summarized in Table IV.

Tides decrease in amplitude towards the north-east, from extreme maxima attained at Strait Gorlo of the White Sea (6 m) at the North Cape and in Murmansk fjords (4 m) to lesser maxima at Spitsbergen (1 m), Franz Josef Land (0.8 m) and Novaya Zemlya (0.8–1.4 m).

The sea is stratified in the warm season but convective mixing occurs in winter. In summer the halo-, thermo- and pycnoclines are situated at depths of 25–50 m. Owing to the bottom topography, there are quasi-permanent zones of downwelling and upwelling, such as the Bear Island and the north-eastern region [15].

As a consequence of the intensive vertical mixing, the Barents Sea waters are well ventilated. The surface waters are oversaturated with oxygen in summer and even in winter the oxygen saturation is not less than 70–80% [13]. Owing to low temperatures, the deep layers of water are rich in carbon dioxide.

Because of the inflow of warm Atlantic waters, the Barents Sea is never completely covered with ice. A typical feature of the Barents Sea ice regime is the large seasonal and interannual variability. The largest ice extent is usually observed in April and varies from 55–60% of the sea surface area in warm winters to over 70% in cold winters [15]. The smallest ice extent is recorded in August and September when, in especially warm years, the sea is completely ice free, while in anomalously cold years the ice cover persists over 40–50% of the sea area. The annual average, minimum and maximum ice cover extents are 38, 22 and 52%, respectively. Stable fast ice forms near the coasts. The ice thickness usually does not exceed 0.3 m at the margins of the sea. Drifting ice reaches maximum thicknesses of 0.8 m in the south-eastern Barents Sea and 1.5 m in the north. In winter, northward ice drift prevails, while in

TABLE V. ICE TRANSPORT IN THE ARCTIC [17]

From	To	km^3/a
Barents Sea	Arctic Ocean	32–33
Kara Sea	Barents Sea (Kara Gate)	4.6
Kara Sea	Barents Sea (northern top of Novaya Zemlya)	140–198
Barents Sea	Kara Sea	20
Kara Sea	Laptev Sea	50
Kara Sea	Arctic Ocean	170
Arctic Ocean	Barents Sea	43.5–58
Barents Sea	White Sea	50
White Sea	Barents Sea	13.6
Arctic Ocean	Greenland Sea by the East Greenland current via Fram Strait	2600

summer southward transport dominates. Extensive polynias are formed near the coasts of the islands, owing to local winds. Ice formed within the Barents Sea itself is usually predominant, with some import from the Arctic Ocean and the Kara and White Seas. Ice import is considerably larger than export (Table V). However, the ice exchange of the Barents Sea is insignificant compared with the ice volume in the sea.

The Barents Sea is characterized by a typical polar sedimentogenesis, with relatively low sedimentation rates and a marked predominance of terrigenous sediments over bio- and chemogenic ones [20]. A total of about 7.5×10^7 tonnes of terrigenous material is supplied to the Barents Sea every year, mainly by coastal and sea floor abrasion, but also by rivers and deposition from the atmosphere. The White Sea, a semiclosed bay of the Barents Sea, receives an input of 9.3×10^7 t/a. Suspended sediment loads range from a few mg/L in the open Barents Sea and southern coastal area to several tens, even hundreds, of mg/L, in some of the bays on the western coast of Novaya Zemlya. Besides key factors such as the complex topography of the sea floor and hydrodynamics, which control sedimentation regimes and mechanical characteristics of the bottom sediments, the presence of large permanently ice free areas contributes to active resuspension of bottom sediments. This process occurs mainly during the stormy autumn–winter period, when wave action can affect sea floor sediments at water depths up to 80 m and even deeper. Owing to the existing system of permanent bottom currents, the resuspended sediment is entrained in long range transport. Sedimentation rates of 0.1 to 0.5 mm/a have been estimated for the Barents Sea at large [17], but values an order of magnitude higher can be expected for the western Novaya Zemlya bays. The entire spectrum of lithological types of bottom deposits is represented in the Barents Sea. South of 72° N, sands are dominant, whereas, to the north, mud prevails. Generally, at depths of less than 100 m, the sea floor is covered mainly by sand, often containing boulders, gravel and shells. Deeper basins are covered by mud. Clayey mud occurs especially in the northern part of the sea. Because of the active hydrodynamic regime, bimodal grain size distributions, as well as polycomponent sediments, are dominant and monogranular sediments are very rare. Light minerals (quartz, feldspar) and, in the fine grained sediments, clay minerals (mainly illite), predominate [13]. Heavy minerals, in general, rarely constitute more than between 1 and 5% of the surface layer. The proportion of SiO_2 ranges from 85% in sands to 58% in muds and clayey muds, while that of metallic oxides varies from 8 to 25%, showing a clear correlation between chemical and mechanical compositions. The manganese content increases from 0.01% in the sands of the southern Barents Sea to about 0.6% in the northern muds. The organic matter content in the Barents Sea sediments ranges between 0.15–3.12% for carbon and 0.02–0.42% for nitrogen, both increasing in proportion with the clay content. The phosphorus content increases from below 0.05% in the south to 0.32% in the north. Active chemical processes are related to precipitation of compounds leached by rivers from the mainland and occur principally in the area where fresh water and sea water mix. Densities between 1.35 and 1.99 g/cm^3 and water contents of 30 to 67% have generally been observed for bottom sediments [20].

3.1.1.3. Kara Sea

The Kara Sea (Fig. 4) lies on the western part of the Arctic continental shelf of Asia and is connected with the Barents Sea to the west, the Arctic Basin to the north and the Laptev Sea to the east. It has an area of 883 000 km^2 and a volume of 98 000 km^3. The sea is rather shallow (Fig. 6), with 82% of its area lying on the continental shelf at depths under 200 m and depths exceeding 500 m

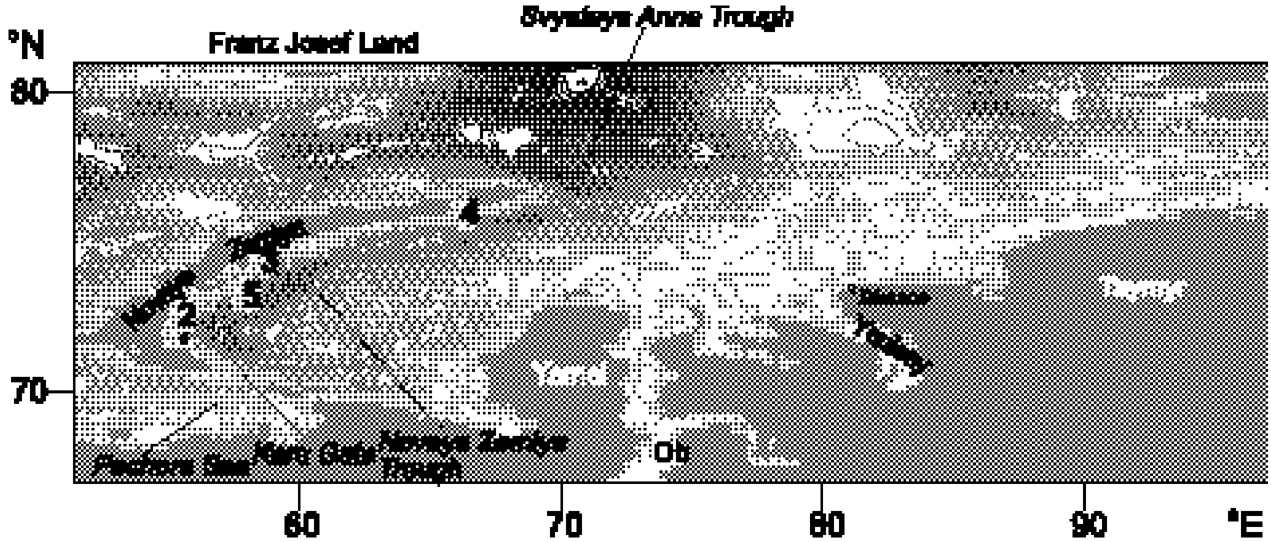


FIG. 6. Kara Sea bathymetry with the main dump sites. 1 – Abrosimov Fjord, 2 – Stepovoy Fjord, 3 – Tsivolka Fjord, 4 – Techeniye Fjord, 5 – Novaya Zemlya Trough.

over only 2% of its area [15]. The most prominent topographical features are three submarine rifts: the Sv. Anna and Voronin Troughs in the north and the Novaya Zemlya Trough in the south-west, along the eastern coast of Novaya Zemlya, with maximum depths of 620, 450 and 430 m, respectively. The Novaya Zemlya and Sv. Anna Troughs are separated by a 100 m deep sill. The northern troughs form pronounced re-entrants from the Arctic Basin into the shelf and play important roles — particularly the Sv. Anna Trough — in shaping the water circulation between the Barents Sea, the Kara Sea and the Central Arctic.

Strong winter cooling and weak summer warming, unstable weather in the cold season and relatively calm atmospheric conditions in summer are characteristic features of the Kara Sea climate [15]. Mean monthly air temperatures range between –20 and –28°C in winter and 1 to 6°C in summer. The extreme air temperatures recorded are –48 and –20°C. In summer and autumn winds are generally between 4 and 7 m/s, the frequency of winds weaker than 11 m/s being 88%. Extreme winds of 40–60 m/s — the boras — occur near the mountainous coasts of Novaya Zemlya and Severnaya Zemlya, typically lasting several hours but possibly lasting two to three days in winter. Storms are most frequent in the western Kara Sea. In winter six to seven storm days per month have been recorded. Precipitation amounts to 20–30 cm/a.

There are essentially five water masses in the Kara Sea [15]. The distribution and extent of these water masses vary not only seasonally on the basis of ice formation and melting, but also interannually, as they are controlled by atmospheric processes generated by the dominance of either the Icelandic depression or the Arctic pressure maximum. The surface waters of the Arctic Basin come in through the north from the Central Arctic and are located in the upper layers of the Kara Sea. They are characterized by rather small seasonal changes in salinity and temperature around typical values of 32‰, slightly increasing with depth, and –1.8°C, i.e. close to freezing point, respectively. The surface Kara Sea water can suffer modifications due to ice formation and mixing with other water masses. It is therefore characterized by significant seasonal fluctuations of salinity and temperature: between 25‰ and –1.4°C, respectively, in winter, and 22‰ and 7°C, in summer. The Barents Sea waters are of Atlantic origin and come into the Kara Sea between the northern tip of Novaya Zemlya (Mys Zhelaniya) and Franz Josef Land and also through the Southern Novozemel'skiye straits, mainly between the southern tip of Novaya Zemlya and Vaygach Island (Kara Gate). This water mass is characterized by high salinity all year round: from 35.3‰ in summer to 35.6‰ in winter. Its temperature is vertically homogeneous and varies from –1.9°C in winter to 6°C in summer. The salty and warm Atlantic water mass is formed from water flowing into the Kara Sea at depth from the Arctic Basin through the Sv. Anna and Voronin Troughs. It presents no significant seasonal variation in its salinity (35‰) or temperature (2.2°C).

River waters have a strong influence on the hydrological regime of the Kara Sea. Their temperature ranges between 0°C in winter and 11.7°C in summer. The extent of the water mass formed by river water discharge, usually defined by an upper limit of salinity of 25‰, may reach up to 30% of the area of the Kara Sea. A strong frontal zone develops where river runoff meets more saline shelf waters. Five large rivers flow into the Kara Sea from the mainland (Fig. 5). There are Yenisey, Ob, Pyasina, Taz and Pur, the first two delivering more than 90% of a total estimated mean inflow of 1100 to 1500 km^3/a [16]. Several small rivers run into the Kara Sea from Novaya Zemlya, totalling less than 3% of the inflow from mainland rivers. As in the Barents Sea, there is a small contribution of fresh water inflow related to melting of snow and ice along the coastlines.

The water circulation pattern in the Kara Sea is strongly influenced by river inflow. A cyclonic gyre forms in the south-western Kara Sea, while, in its north-eastern part, northward and eastward currents dominate. The main flows in and out of the Kara Sea are given in Table IV. As can be observed, the dominant paths of water flowing out of the Kara Sea are northward, to the Arctic Basin, and eastward, to the Laptev Sea.

The average values of tides range between 0.3 and 0.8 m [15]. Maximum values of 1.8 m are observed in the northern part of the Ob estuary. Along the shores of Novaya Zemlya, tidal amplitudes reach 0.5–0.9 m.

During summer, owing to the melting of sea ice, increased river discharge and insolation, well defined thermo-, halo- and pycnoclines form at 6–8 m in shallow parts of the sea and at 20–30 m in deeper regions [21]. Cooling and ice formation accompanied by brine release result in strong vertical mixing during winter. Zones of upwelling and downwelling persist in the region of the continental slope and the troughs [17].

Oxygen concentrations are fairly high throughout the Kara Sea waters. Even in the relatively enclosed Novaya Zemlya Trough, oxygen values are typically no more than 15% lower than on the adjacent shelf [21]. Ventilation due to convection occurs following ice formation so that only near the bottom at depths in excess of 300 m are oxygen values less than 70% saturation, indicating reduced renewal of waters.

Owing to its severe climate, the Kara Sea freezes over in autumn and winter and is never totally ice free [17]. New ice formation starts in September in the north and about one month later in the south. On the average, from October to May, about 94% of the area of the Kara Sea is covered by ice. The mean thickness of fast ice is between 1.2 m in the south-west and 1.8 m in the north-east. Drifting ice in the central part of the Kara Sea can reach thicknesses of 1.5–2 m. The Novaya Zemlya ice massif generally melts in late summer but is a perennial feature in some years. In the northern regions, ice persists all year round. Icebergs are observed in the south-western Kara Sea, generally along the coast of Novaya Zemlya, and appear to be calved mostly from glaciers on the northern island of Novaya Zemlya. It is noted [21] that most of the icebergs generated by glaciers on Novaya Zemlya are trapped in the shallow bays. There is a net outflow of ice from the Kara Sea and a dominant flux northward. The average transit time required for ice to drift from the Kara Sea through the Arctic Basin to Fram Strait is 2–2.5/a [21, 22], considerably faster than the 5–8 a transit time estimated for water. The total ice export to the Barents and Laptev Seas and to the Arctic Basin exceeds 400 km^3/a (Table V).

The extensive ice cover for at least ten months a year, which considerably damps wave action, and the river discharge are critical factors controlling the sedimentation regime in the Kara Sea. A total of 1.4×10^8 t/a terrigeneous material is supplied to the Kara Sea, 80% of which comes from coastal abrasion [15]. The main rivers discharge some 3×10^7 t/a of suspended matter, 80% of it being deposited in the estuaries. The Novaya Zemlya bays are an important source of clayey material for the Kara Sea. However, only 1.5% of the particles discharged to the Kara Sea are dispersed beyond it [14]. The suspended matter content in south-western Kara Sea waters is below 10 mg/L, generally between 3 and 7 mg/L. Sedimentation is slow on the Kara Sea shelf and is not augmented by organic oozes, since the metabolic productivity rates are also low. Sedimentation rates are estimated to be between 0.1 and 2 mm/a, the higher values being attained in areas of the Novaya Zemlya fjords. In the Kara Sea, bigranular silts (55%), sands (20%) and muds (20%) are dominant. In coastal areas and on rises, sandy and gravelly sands prevail whereas the deep sea bottom is characterized by clayey and silty sediments. Large numbers of ferromanganese nodules are found in some areas of the Kara Sea [13], along the coast and in the bays of Novaya Zemlya. This is typical for an Arctic continental shelf area with important river inflow and was also observed during the Joint Russian–Norwegian 1993–1994 expeditions. Feldspar quartz and micaceous quartz varieties of terrigeneous sediments are dominant [13]. In the western Kara Sea, densities of 1.40 to 1.65 $g \cdot cm^{-3}$ and water contents of 32 to 60% were determined for bottom sediments [17]. The organic carbon content of sediments is in the range of 0.27 to 1.99% [13].

3.1.1.4. Kara Sea dumpsites

The only specific information available for the sites where radioactive waste has been dumped is derived

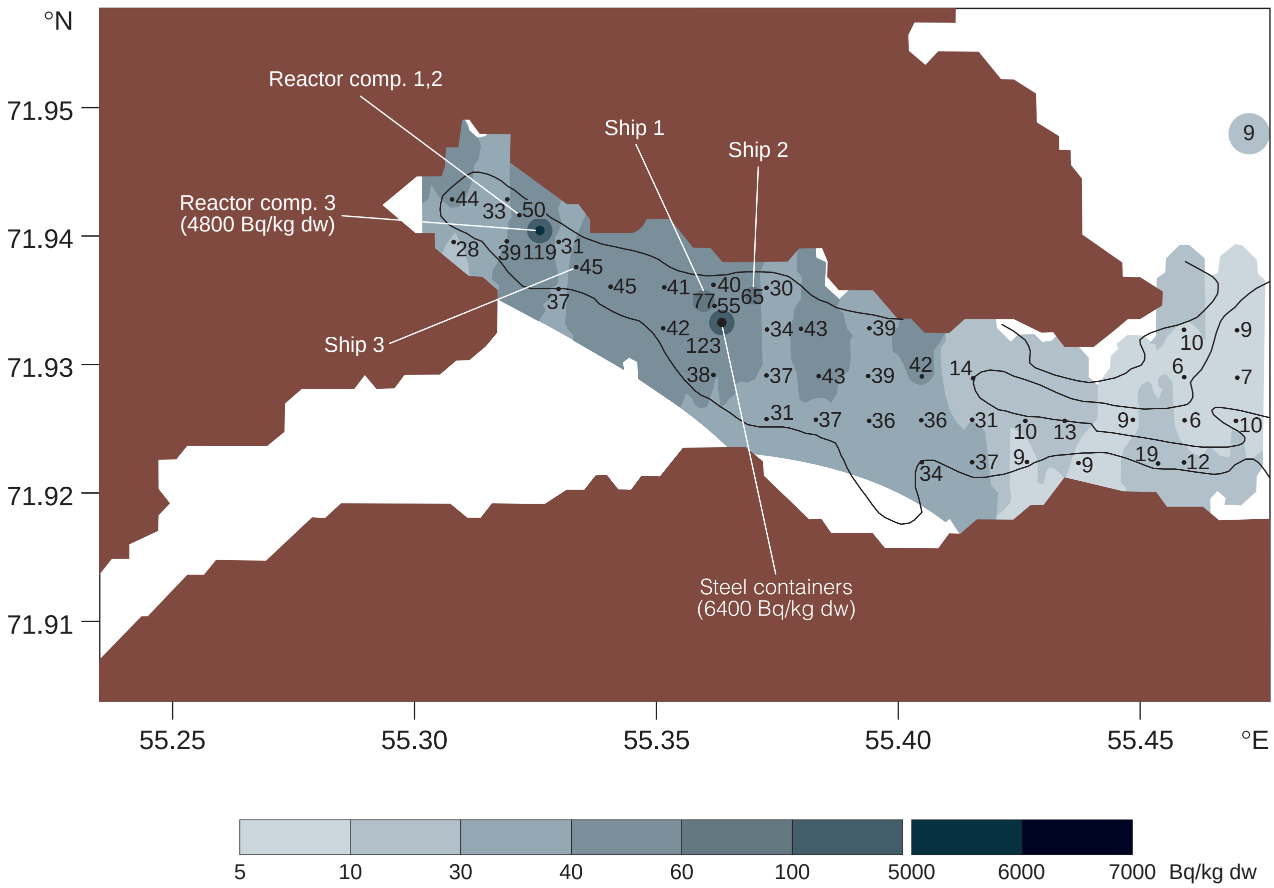


FIG. 7. Bathymetry of Abrosimov Fjord with localized dumped objects. The figures refer to ^{137}Cs levels in surface sediments at various sampling points. The contours of ^{137}Cs concentrations are also shown. White areas: no data available.

from reports of the 1992–1994 joint Russian–Norwegian expeditions to the Kara Sea [23–25].

Abrosimov Fjord (Fig. 7) has smooth topography, with depth reaching maximum values of about 20 m inside the fjord [25]. There are two inflowing rivers, the main inflow occurring through the Abrosimov River. The fjord is clearly under the influence of Barents Sea waters flowing into the Kara Sea through the relatively nearby Kara Gate. High salinity (33.4 to 34.1‰, except for a very thin desalinated surface layer) and temperature (0.4°C at the surface to over –1.2°C at the bottom) were recorded in August and September, and several water masses can be clearly identified on the basis of the CTD profiles. Bottom sediment generally consists of fine mud with clayey mud covering the central part of the fjord. The bottom of the shallow areas at the mouths of the rivers is covered with sand. Coarse sands covered with gravel lie at the mouth of the fjord. Life is relatively abundant in the Abrosimov Fjord.

Stepovoy Fjord has widths not exceeding 2 km on the 10 km stretch investigated by 1993 and 1994 expeditions (Fig. 8). A shallow sill at about 20 m depth separates the inner, deeper part (maximum depth 60 m) from the outer part of the fjord (depths up to 40 m). A dissolved oxygen profile measured in the inner part of the fjord indicates good ventilation of bottom waters [25]. In September, the inner basin was relatively strongly stratified compared to the outer part of the fjord and the nearby coastal area. The water temperature decreased from 3.7°C at the surface to –1.7°C near the bottom, while salinity ranged from about 20‰ in the top 10 m to 34.8‰ at the bottom [24, 26]. Temperature and salinity were quite constant below 35 m. In the outer part of the fjord, temperature varied very little with depth in the range of 2.6–3.4°C. Salinities slowly increased from 17 to 28‰ in the central part of the outer basin, while at the mouth of the fjord they were constant at 18‰ down to 15–20 m, then gradually increased to 24‰ near the

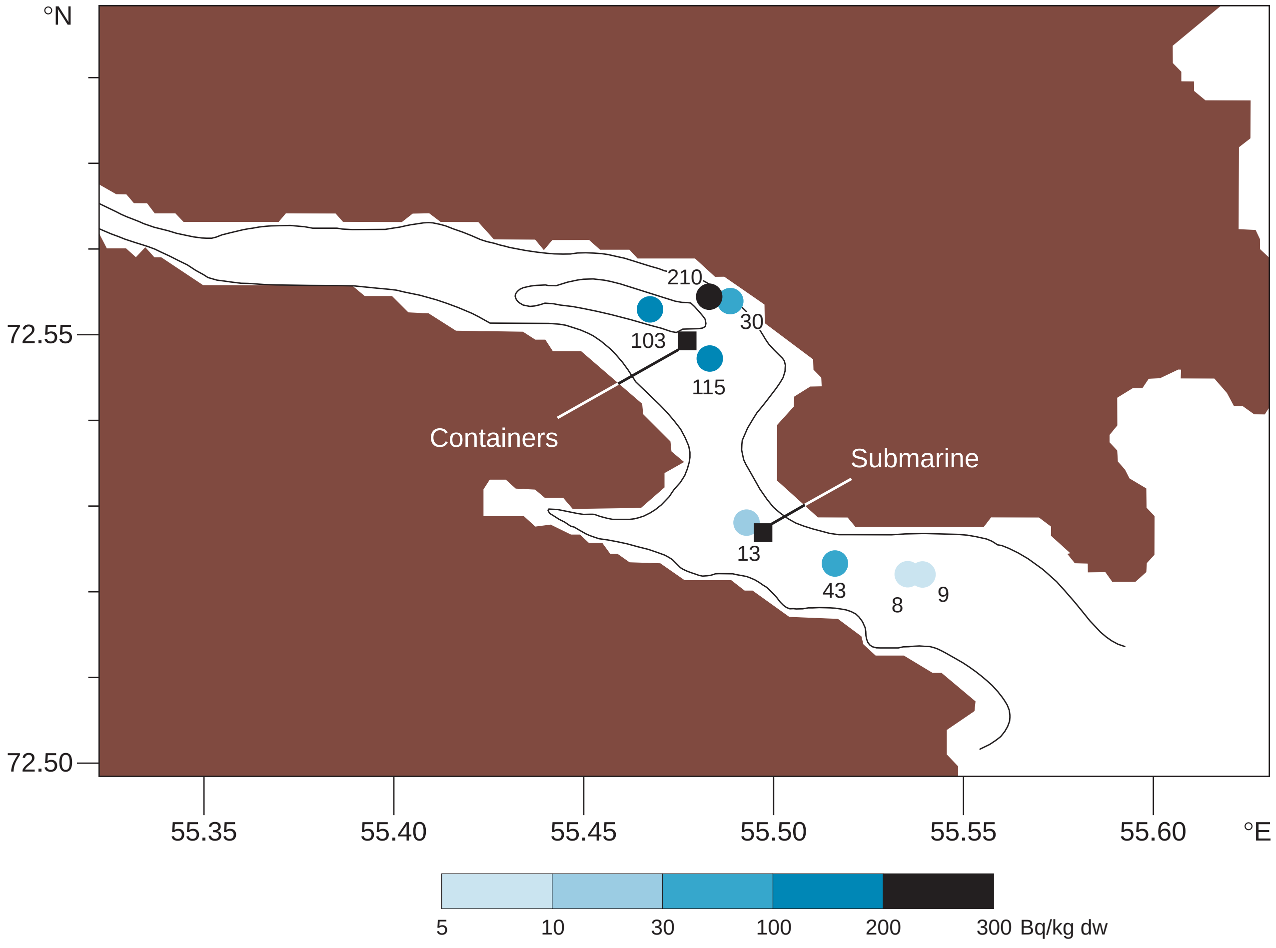


FIG. 8. Bathymetry of Stepovoy Fjord with localized dumped objects. The figures refer to ^{137}*Cs levels in surface sediments at various sampling points.*

bottom. The fresh water component is thus more important in the outer part of the fjord. The bottom sediments in this area also indicate strong flushing of the bottom. In the inner basin, gravel and concretions line the bottom close to the shores and sill. The deepest region is covered with fine silt.

Tsivolka Fjord is a relatively open inlet (Fig. 9) with typical fjord-like topography. In September, the water temperature was 3.5°C at the surface and –1.5°C at the bottom, with a thermocline at around 20 m depth [24, 26]. Salinity ranged between 14‰ at the surface and 38.4‰ at the bottom. In the bottom layer water turbidity was high. Except for shallow nearshore sites and high bottom current areas at the mouth of the fjord, the bottom sedimentary material consists of fine mud. The fjord is rich in benthic fauna, particularly sea stars (*Ophiuroidea*), isopods and amphipods (*Gammarida*). The most abundant species of seaweed are *Fucus evanescens* and *Laminaria digitata* [26]. Fish are rare.

The fjords are completely frozen in for at least ten months a year. Average flushing times of a few months can be expected; somewhat less in winter owing to the ice cover [27]. Complete flushing of the fjords, especially the smaller and shallower ones, can occur within a couple of days in stormy conditions. No signs of ice scouring were seen during visual inspection of the fjord bottoms.

Novaya Zemlya Trough (Fig. 10) reaches depths up to about 430 m. Temperature and salinity profiles [24] have allowed the identification of four main water masses, in which the influence of river water, water of Atlantic origin and processes related to ice formation can be recognized [26]. The surface layer has a relatively low salinity (about 32‰) and high temperature (2–2.5°C). Below the sharp halocline at 20 m, the salinity increases regularly with depth from about 34.5 to almost 35.4‰. The temperature, however, is constant at –1.75°C down to about 100 m, then rises to a maximum of about 1.2°C at 150 m and finally diminishes slowly to –1.86°C (near freezing temperature) near the bottom. The high density of the bottom water obviously leads to increased residence time of water in the lower layers of the trough. At the sites investigated, the bottom is covered with fine sediments and is inhabited by a benthic fauna composed mainly of brittle stars and amphipods.

3.1.2. Ecology

3.1.2.1. Biological production in the Arctic

Biological production in high latitudes is usually low and very uneven throughout the year. The very poor illumination over much of the year with darkness during the extreme winter months, loss of light due to reflection

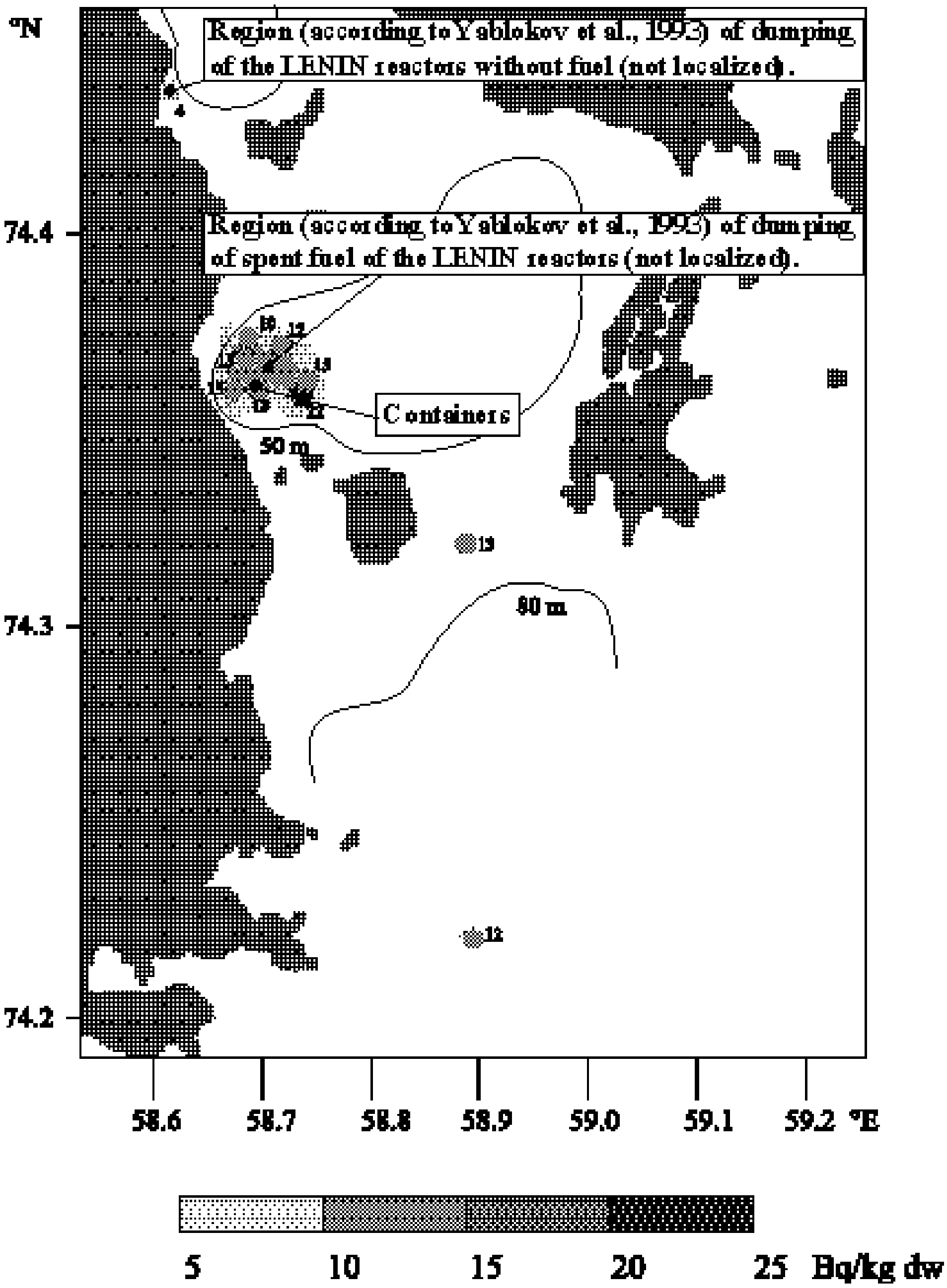


FIG. 9. Tsivolka Fjord with dumped objects. The figures refer to ^{137}Cs levels in surface sediments at various sampling points.

and the greatly reduced penetration due to the thick ice cover severely limit primary (phytoplankton) production. A marked seasonal change in primary production may be evident, however, since over the very brief summer of long days, a bloom of phytoplankton, mainly diatoms, can occur. Seeding of diatoms is normally satisfactory at high latitudes because some species, especially those which are in their resting stage, can survive in the ice. As soon as the snow cover disappears, the ice thins and melt ponds appear, sufficient light can penetrate for growth even at very low temperatures, as some indigenous species photosynthesize at temperatures below 0°C, even though growth may not be efficient.

In the Arctic Ocean, light penetration through the snow covered ice is so poor that production increases only in late June to a fairly low maximum in August. The

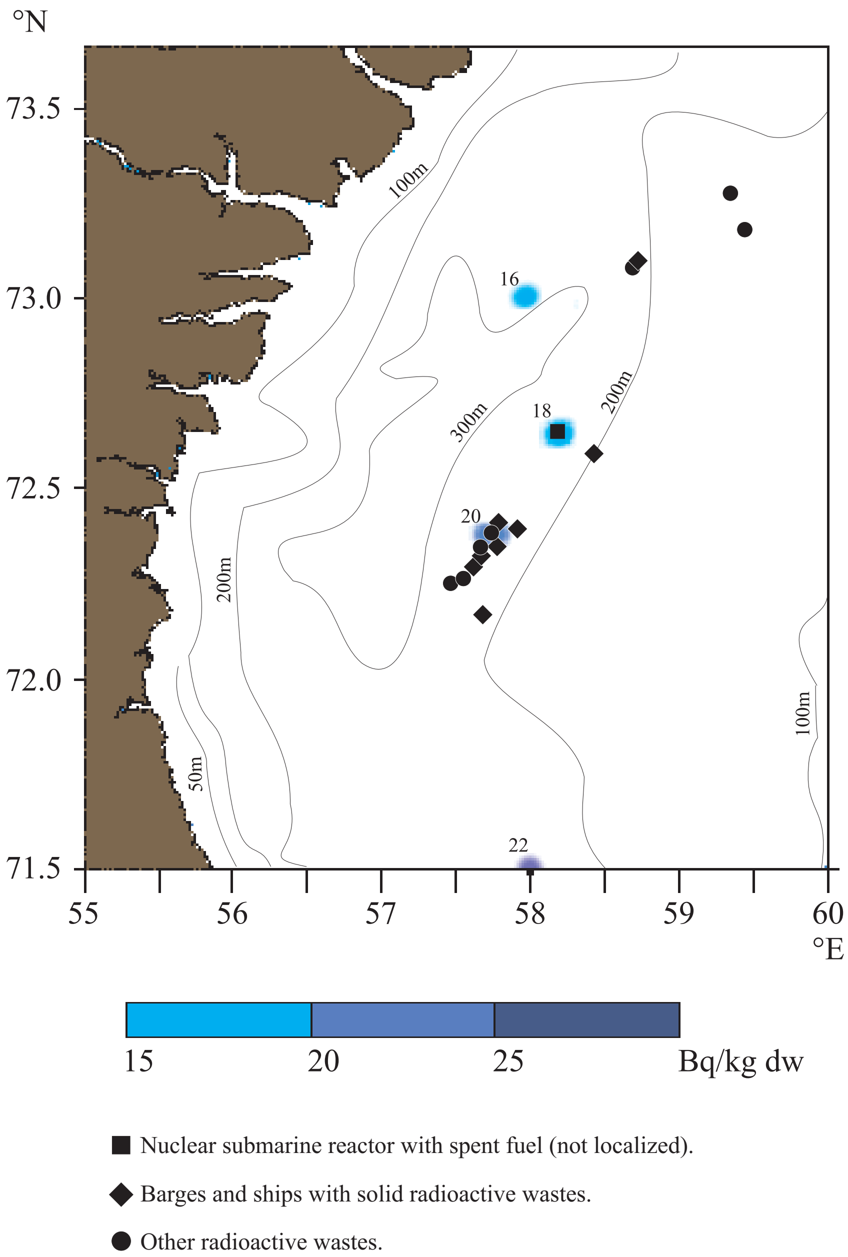


FIG. 10. Novaya Zemlya Trough dumpsite. The figures refer to ^{137}Cs levels in surface sediments at various sampling points. Locations of nuclear submarine reactor with spent fuel (■), barges and ships with solid radioactive wastes (◆), other radioactive wastes (●), according to the White Book [5]. None of the dumped objects has been localized.

productive cycle lasts only about three months and primary production is not exceptionally high even during that period. It is suggested [28] that, in midsummer, production in Arctic waters could reach 1 g $C{\cdot}m^{-2}{\cdot}d^{-1}$ but, where the ice cover was maintained, it could be as low as 0.03 g $C{\cdot}m^{-2}{\cdot}d^{-1}$. During winter, production cannot be detected. With the extremely brief summer, annual production is deemed to be poor although determinations of annual biological productivity are very scarce. An assessment of the annual rate of new production, or the fraction of total biological productivity which depends on the resupply of nutrients to the continental shelves, can be made on the basis of chemical measurements. The regional average new production derived from total carbonates measured in the upper halocline was estimated to be 45 g $C{\cdot}m^{-2}{\cdot}a^{-1}$ [29]. A second estimate for new production was derived from the apparent oxygen utilization rates, which were determined from modelling profiles of chlorofluoromethanes, salinity, temperature and oxygen in the water column [30]. The value obtained for shelf production, subject to several qualifications, was between 8 and 21 g $C{\cdot}m^{-2}{\cdot}a^{-1}$. These numbers can be compared to values for total production in the Arctic Ocean ranging between 12 and 98 g $C{\cdot}m^{-2}{\cdot}a^{-1}$ [31].

As a general rule, at any latitude, inshore waters tend to be more productive than offshore waters. Occasionally, offshore waters can show high rates of production comparable to inshore areas, but these are due to temporary enrichment and are not sustained; annual production is always greater in inshore areas. This also applies to the Arctic Ocean, and biological production will generally be more important in the marginal seas than in the central basin.

The fate of pelagic biological production is to be grazed by marine animals and/or sedimentation of particles. Grazing of phytoplankton by zooplankton is the basic step in any pelagic food web. Still, in polar regions, herbivorous activity by pelagic crustaceans (copepods, euphausiids) does not appear to be as important as passive sinking (as far as controlling phytoplankton biomass and distribution is concerned). This seems especially true in shallower regions and marginal ice zones, where there are large amounts of new production which are apparently not used within the euphotic zone by herbivores. This material apparently sinks from the surface layer and supports a large and active benthic community [32]. There are deeper regions, however, in which grazing is relatively more important.

3.1.2.2. Marine food webs in the Arctic

Marine food webs in polar regions are short, with only two to three carbon transfers between diatoms and apex predators. The apex predators include baleen and toothed whales, seals, walruses, bears and seabirds of several families. In the Arctic Basin, sea ice is greatly consolidated except at its periphery and over much of its surface never melts. Negative effects of this ice coverage in terms of production and energy transfer through the food webs are the prevention of access of land/air predators to the water column and, as already stated, the inhibition of light penetration and thus, of the photosynthetic processes necessary for pelagic food webs [33]. Positive effects of ice include the plankton blooms associated with ice margins, the provision of substrate from which micro- and macronekton seek refuge against predators, and a rich community living below the marine ice that provides important contributions to pelagic food webs [34].

Another habitat feature that strongly affects the trophic levels in the Arctic is the physiographic setting. The Arctic basin is characterized by broad, relatively shallow continental shelves and most of the surface is ice covered year-round; this results in a relatively unproductive ocean. The Bering Sea provides a narrow connection to the Pacific Ocean, and the East Greenland and Labrador Seas connect the Arctic with the Atlantic Basin. These connections are important sources of productive waters which flow to small portions of the Arctic Ocean. They also exert a major influence on the regional zoogeography of the Arctic marine biota, i.e. there is marked longitudinal as well as latitudinal change in faunal composition. The intense cold that pervades the region during winter as a result of the surrounding land masses and the important spatial and temporal changes in temperature and salinity of the surface waters (see the physicochemical description above) result in dramatic large scale east–west and north–south seasonal migrations of many species.

In subpolar waters, many of the fish and invertebrates that are the important links between primary consumers and higher levels of the food web are also species of great commercial importance, e.g. capelin (*Mallotus villosus*), herring (*Clupea harengus*), sandlace (*Ammodytes hexapterus*) and squid (*Illex*) in the North Atlantic, low Arctic zone and walleye pollock (*Theragra chalcogramma*) and herring in the Bering Sea low Arctic area [35]. Thus, a great deal is known about the life cycles, abundance and distribution of these important species. In high Arctic waters, however, the important secondary consumers are relatively less well known. Most of what is known about Arctic cod (*Boreogadus saida*), a key species in Arctic food webs, has been gained through studies of predators, especially ringed seals (*Phoca hispida*) and thick billed murres (*Uria lomvia*) [36].

In the Arctic Ocean, two trophic webs have been identified, one associated with the shallow nearshore and the other with the pelagic offshore habitats [35]. Arctic cod, the pagophilic amphipod (*Apherusa glacialis*), euphausiids (*Thysanoessa*) and copepods (*Calanus*) are central to the pelagic food web and are important in that of the nearshore. In the latter, however, organisms of the epibenthos contribute substantially to energy flow. Arctic cod in their first year feed principally on copepods but as juveniles and adults they increasingly turn to amphipods such as *Parathemisto* [36] for their nourishment. Although a number of mammals are components of the Arctic marine food webs year-round, especially polar bears (*Ursus maritimus*), ringed seals and bearded seals (*Erignathus barbatus*), there are only two avian predators: black guillemot (*Cepphus grylle*) and ivory gull (*Pagophila eburnea*) [37]. Polynias (areas within the ice pack almost always clear of ice and with enhanced productivity at their ice edges) are particularly important to the development of these Arctic Ocean food webs.

Several apex predators winter in large numbers in the marginal annual ice of the Bering, Labrador, East Greenland and Barents Seas. Where large polynias occur, these predators can also be found within the pack ice. Included among the mammals are the bowhead whale (*Balaena mysticetus*), narwhal (*Monodon monoceros*), white whale (*Delphinapterus leucas*), ribbon seal (*Histriophoca fasciata*) and walrus (*Odobenus rosmarus*) [37]. Major components of avian fauna are the ivory gull, as well as the glaucous, Iceland, slaty backed, and Ross's gulls (*Larus glaucescens*, *L. leucopterus*, *L. schistisagus*, and *Rhodostethia rosea*) and thick billed murres (*Uria aalge* and *U. lomvia*) [38]. At the Bering Sea ice edge, walleye pollock, capelin, euphausiids and a pelagic amphipod (*Parathemisto libellula*) are the mainstays of the midwater food web; walrus feed on benthic molluscs, and ribbon seals on demersal fish [39]. Copepods, again, are important primary consumers. At high Arctic ice edges, Arctic cod, copepods (*Calanus*), *Parathemisto* and pagophilic amphipods are important in the intermediate trophic levels [40]. Farther south, Arctic cod, capelin and *P. libellula* are the major prey of upper level predators.

During the summer, a number of migrants increase the numbers of predators. Most of them seek out ice free waters exclusively. For example, harp seals move north to feed in the epibenthos on the shelves of the eastern Canadian Arctic, as do grey whales, fin whales and other baleen whales and fur seals which frequent oceanic waters along continental slopes. Also moving into polar

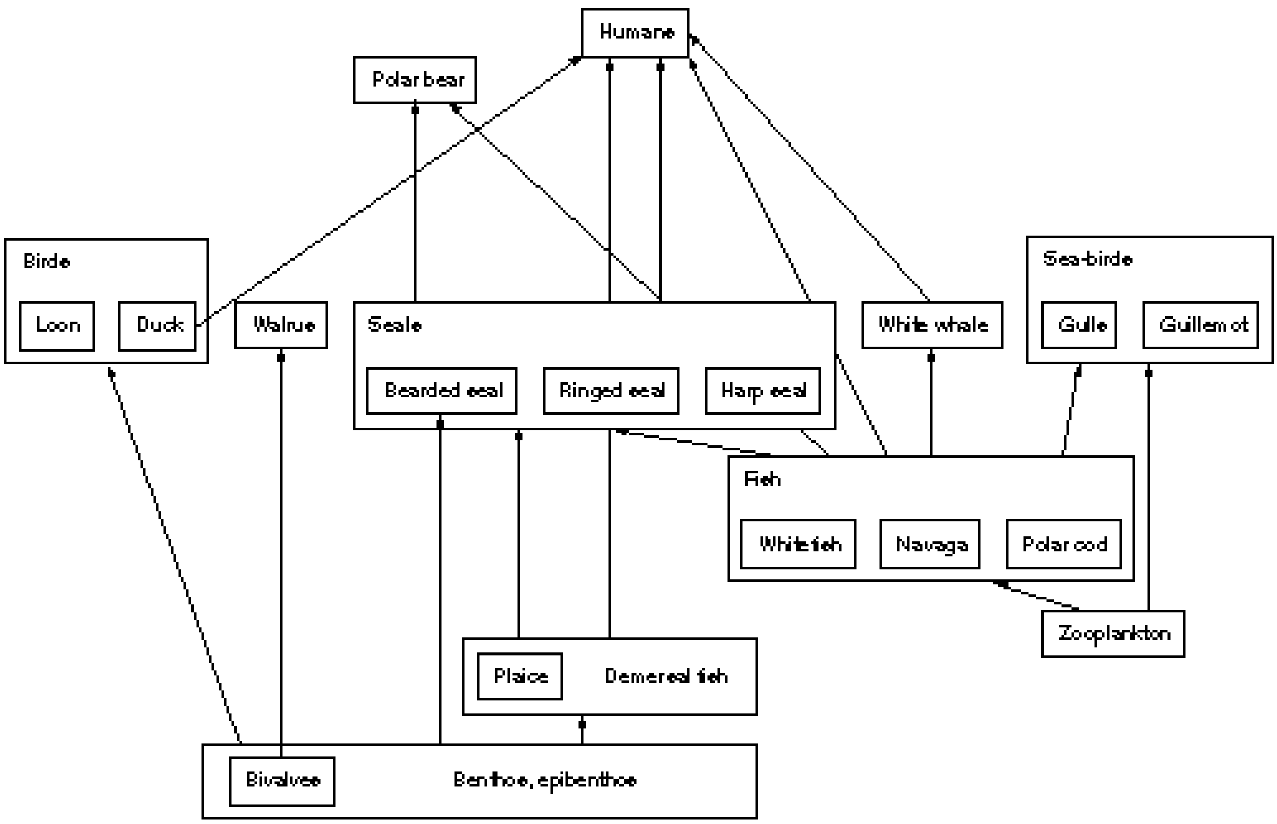


FIG. 11. Food web of the upper trophic levels in the shallow Kara Sea and potential food sources for humans (from Ref. [43]).

waters are many species of seabirds, the most abundant of which are oldsquaw and eider ducks in the nearshore (where they feed in the epibenthos), as well as shearwaters, northern fulmars, kittiwakes, murres, auklets and dovekies.

Arctic cod is paramount in the food web of high Arctic areas, but key prey vary in low Arctic areas. For example, in the eastern Barents Sea during the summer, Arctic cod is the main prey of seabirds, but in the western Barents Sea, which is strongly influenced by the warm North Atlantic Current at that time of year, sandlace, capelin and herring are important food items to predators [41].

The central zone of the Kara Sea is low in productivity, with few and small sized fish. Echinoderms dominate as predators; therefore, the energy transfer to the higher trophic levels through the food web there is small. The trophic chain to higher organisms and humans in the more coastal environment of this Sea is shown in Fig. 11, which highlights the major energy flows in the shallow waters of the Sea, the potential seafood sources for humans and the paths through which radionuclide or other contaminants in the environment may reach humans.

3.1.2.3. Fish and fisheries in the Barents and Kara Seas

The estimated average annual primary production of the Barents Sea of 110 g $C\cdot m^{-2}a^{-1}$ is far more than the average for the centre of a warm ocean (<50 g $C\cdot m^{-2}\cdot a^{-1}$) and the ice free parts of the Antarctic ocean (<20 g $C\cdot m^{-2}\cdot a^{-1}$) but slightly less than that estimated for the Bering Sea and the continental shelf off Norway [42]. This primary production is again the basis for the animals grazing on it, i.e. the secondary (zooplankton) production. The main zooplankton populations consist of copepods (mainly *Calanus finmarchicus*, *C. hyperboreus* and *C. glacialis*) and krill (mainly the smaller species *Thysanoessa inermis*, *T. raschii* and *T. longicaudata*). Amphipods also play an important role in the Arctic ecosystem. Of these, *Parathemisto libellula* is the most abundant and important. As a predator on smaller zooplankton, it forms an important link to the next step in the food web, i.e. seals, polar cod and diving birds.

The Barents Sea ecosystem contains some of the largest fish stocks of the world, e.g. the capelin (*Mallotus villosus*), which had a maximum registered stock size in the mid-1970s of about 7.5 million tonnes [43], the Northeast Arctic cod (*Gadus morhua*) with a stock size of about 6 million tonnes just after the Second World War [44] and, in part, the Norwegian spring spawning herring (*Clupea harengus*), which had a maximum stock size of 10 million tonnes before it collapsed in 1970 [43].

The annual Barents Sea harvest is from 2 to 3.5 million tonnes, and the catch is mainly divided between Norway and the Russian Federation. There are strong interactions between these stocks, and variations in the year class strength of the various fish stocks have a marked influence on other components of the ecosystem [45].

In addition to the direct harvest of the area, the Barents Sea is important as a feeding ground for fish populations harvested further south on the Norwegian shelf. The Norwegian shelf area from 62°N northwards is a spawning ground for the most important fish populations of the Northeast Atlantic. Fish eggs and larvae are transported via the Norwegian Coastal Current into the Barents Sea, where fish fry may benefit from abundant food.

The modest diversity of the Kara Sea ichthyofauna, which is due to the constraints of severe climatic conditions, contrasts sharply with the rich assortment of commercial fish in the Barents Sea [46]. The Atlantic boreal and arctoboreal species circulate within a limited distribution area of the Kara Sea, mainly its southwestern sectors. Altogether, the Kara Sea hosts 24 families of fish with 53 representatives. While polar cod (*Boreogadus saida*) is the most abundant and widespread species in the Kara Sea, navaga (*Eleginius navaga*) and polar plaice are usually found in its coastal and brackish waters.

The fish population is very small within the central area, which is characterized by brown muds and may justifiably be called the ‘fishless zone’ of the Kara Sea. This is explained by the generally lower productivity of this body of water and by the brown mud possessing conditions unfavourable to fish life. The bottom fish community of the deep waters consists primarily of small sized members of the Cyclopteridae, Zoarcidae and Cottidae families (*Artediellus*, *Ulcina*, *Licodes*, *Liparis* and *Triglops*). However, even these small sized fish are extremely rare. The only Barents Sea species also found in the brown mud is the long rough dab (*Hippoglossoides platessoides*) that lives in small numbers as immature specimens or mature dwarfs. They are small in size and grow slowly. For example, a six to seven year old dab is 15.5 to 17.5 cm long in the Kara Sea while at this age a member of the same species in the Barents Sea measures 30 to 31 cm.

Close to the shore, the situation is different and fish are far more abundant. The mouths of the great rivers Ob, Yenisey and Pyasina abound in fish. Suitable conditions for local brackish water fisheries exist in the southern part of the Sea, off the mainland and along the coast of Novaya Zemlya; industrial fisheries have been developed in the relatively productive Ob–Yenisey sector [47]. The northern char (*Salmo alpinus*) is found in the mouth of the rivers of Novaya Zemlya. Other major fish are the Arctic Sea whitefishes (*Coregonus*), frostfish (*Osmerus*), navaga, Arctic cods (*Gadidae*) and the Polar dab and goby. Many other fish caught there belong to the coregonids and salmonides (beardie, *Stenodus leucicthus nelma*, grayling and others). Polar cod (*Boreogadus saida*) are fairly frequently caught off the Novaya Zemlya coast, especially within the regions of the Kara and Matochkin Straits.

Separate statistics on fisheries for the Kara Sea do not exist. This Sea is included by the FAO in the vast fisheries area 18, i.e. 'The Arctic Sea', whereas by ICES it is included in the Barents Sea zone (ICES area 1). National fishery data exist for the largest gulfs of the Sea: in 1990, the total catch in the Ob estuary was 1526 tonnes of sea and brackish water fish and 836 tonnes of freshwater fish, whereas the total fish catch was 228 tonnes in the Yenisey basin [17]. Whitefish were the most abundant species in the catches at both places.

3.2. RADIOACTIVITY OF THE ARCTIC SEAS

3.2.1. Marine radioactivity database

The IAEA Marine Environmental Laboratory (IAEA-MEL), within the IASAP programme, has been a central facility for the collection, synthesis and interpretation of data on marine radioactivity in the Arctic Seas. The data have been included in the IAEA-MEL Global Marine Radioactivity Database (GLOMARD), which has been developed to store all data on marine radioactivity in seawater, sediments and biota [48]. The database is designed to serve the following functions: (i) to provide immediate and up to date information on radioactivity levels in the seas and oceans, (ii) to provide a snapshot of activities at any time in any location, (iii) to investigate changes in radionuclide levels with time; and (iv) to identify gaps in available information.

The format of the data has been rigorously prescribed to meet programme objectives and ensure maximum utility. The degree of detail is extensive (general sample information including type, method of collection and location as well as physical and chemical treatment) to allow the data to be validated and its quality assured. In addition, the database has links to IAEA-MEL's in-house analytical quality control database, allowing immediate checks on laboratory practice. Information is stored in such a way as to facilitate data retrieval and analysis.

The database provides a critical contribution to the evaluation of the environmental radioactivity levels of radioactive substances in the region and to the assessment of the radiation doses to local, regional and global human populations and to marine biota.

The database enables the following steps to be carried out:

(a) Evaluation of nuclide ratios — the database allows the identification of the individual radionuclide contributions to activity levels in the region, which is critical given the multiple nature of the source terms.
(b) Investigation of time trends — given the temporally varying nature of the known sources of radionuclides in the region, the database helps to estimate source contributions to the environmental concentrations and thus increases the sensitivity with which any small residual change or trend may be detected.
(c) Inventory calculations — the ability to carry out budget calculations may again permit the detection of any imbalances.
(d) Model validation — to provide reliable predictions of the impact of real or hypothetical discharges, it is necessary to use validated models, and this requires access to the existing appropriate experimental data (either in the form of time series of observation or a snapshot of activities).

About 6000 data entries from the Arctic Seas have already been included, with initial emphasis on the extensive joint Norwegian–Russian data, IAEA-MEL's own measurements and data obtained from other institutions.

3.2.2. Pre-1992 radionuclide concentrations

Results of radionuclide measurements in water, sediment and biota of the Arctic Ocean are sparse. The density of data is so low that sophisticated methods of data analysis cannot be used for data evaluation. The evaluation of time trends may be carried out only for the Norwegian, Barents and Kara Seas.

3.2.2.1. Water

Figure 12(a) shows the evolution of yearly averaged surface concentrations of ^{90}Sr and ^{137}Cs with time in Barents Sea surface water. The only well documented record available is for ^{90}Sr [49] and partly for ^{137}Cs [50, 51]. ^{90}Sr levels are gradually decreasing from an average of 19 Bq/m in 1964 to the present value of some 4 Bq/m. A slight increase in concentrations at the end of the 1970s and the beginning of the 1980s may be associated with Sellafield peak releases in the mid-1970s. This is, however, much better documented by ^{137}Cs records,

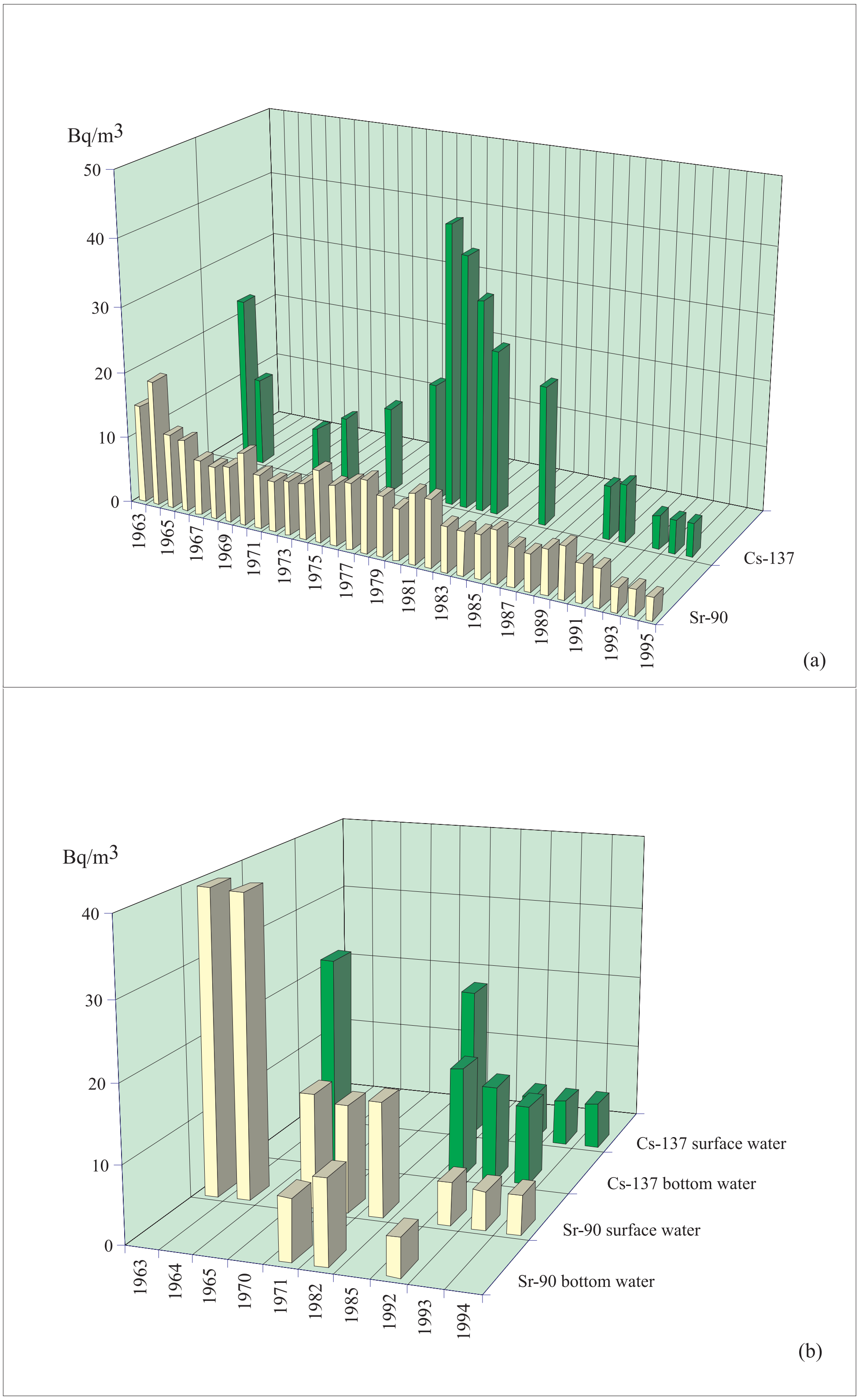


FIG. 12. Yearly average concentrations of ^{90}Sr and ^{137}Cs in surface water in the (a) Barents and (b) Kara Seas.

although values are also missing for several of these years. An approximate transit time from Sellafield to the Barents Sea can be estimated to be four to five years.

The data from GLOMARD show that there has been a decrease in anthropogenic activity levels in the Kara Sea in recent years (Fig. 12(b)). For example, the ^{137}Cs

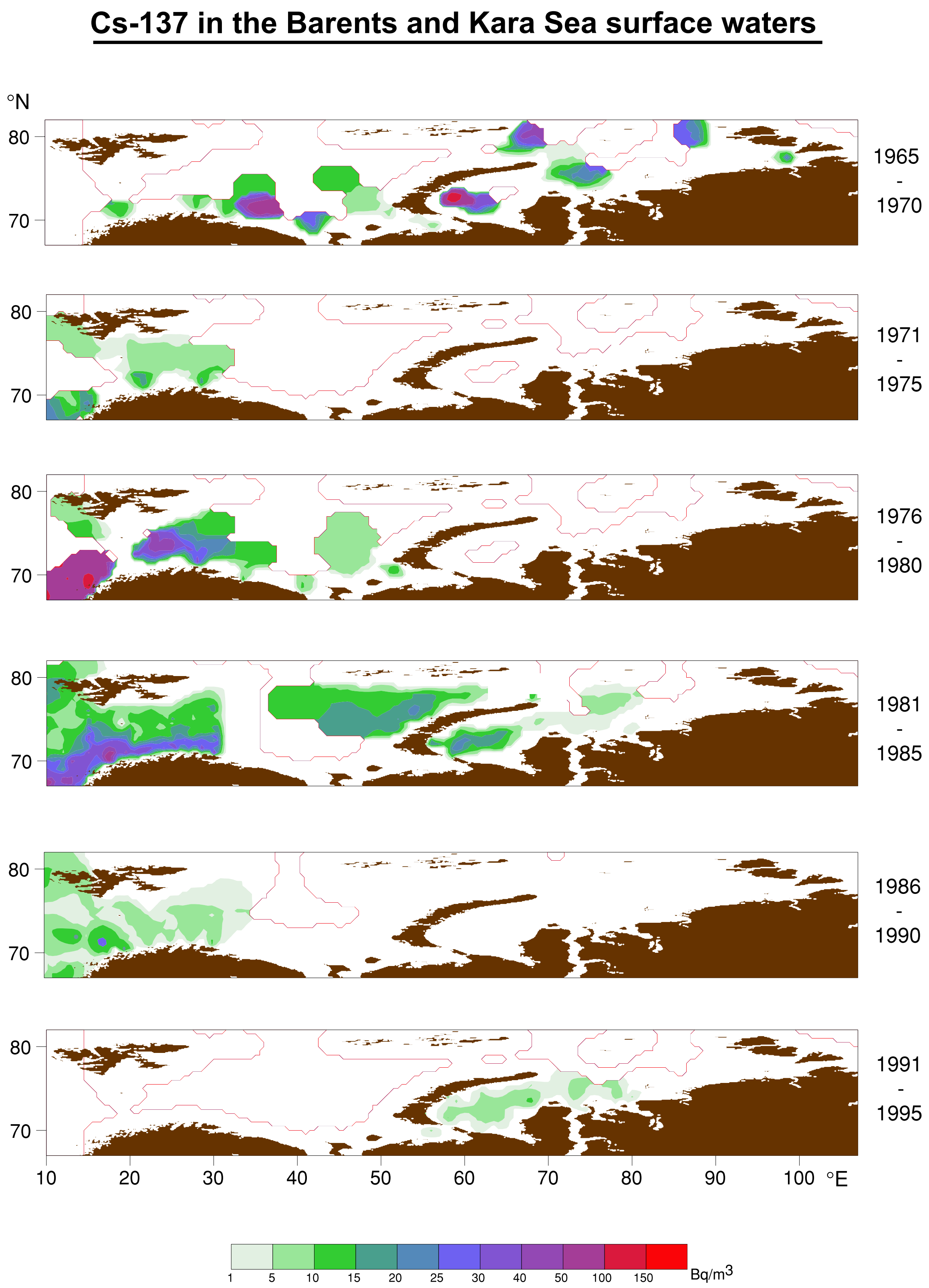


FIG. 13. Contours of ^{137}Cs levels in Barents and Kara Sea surface waters. White areas: no data available.

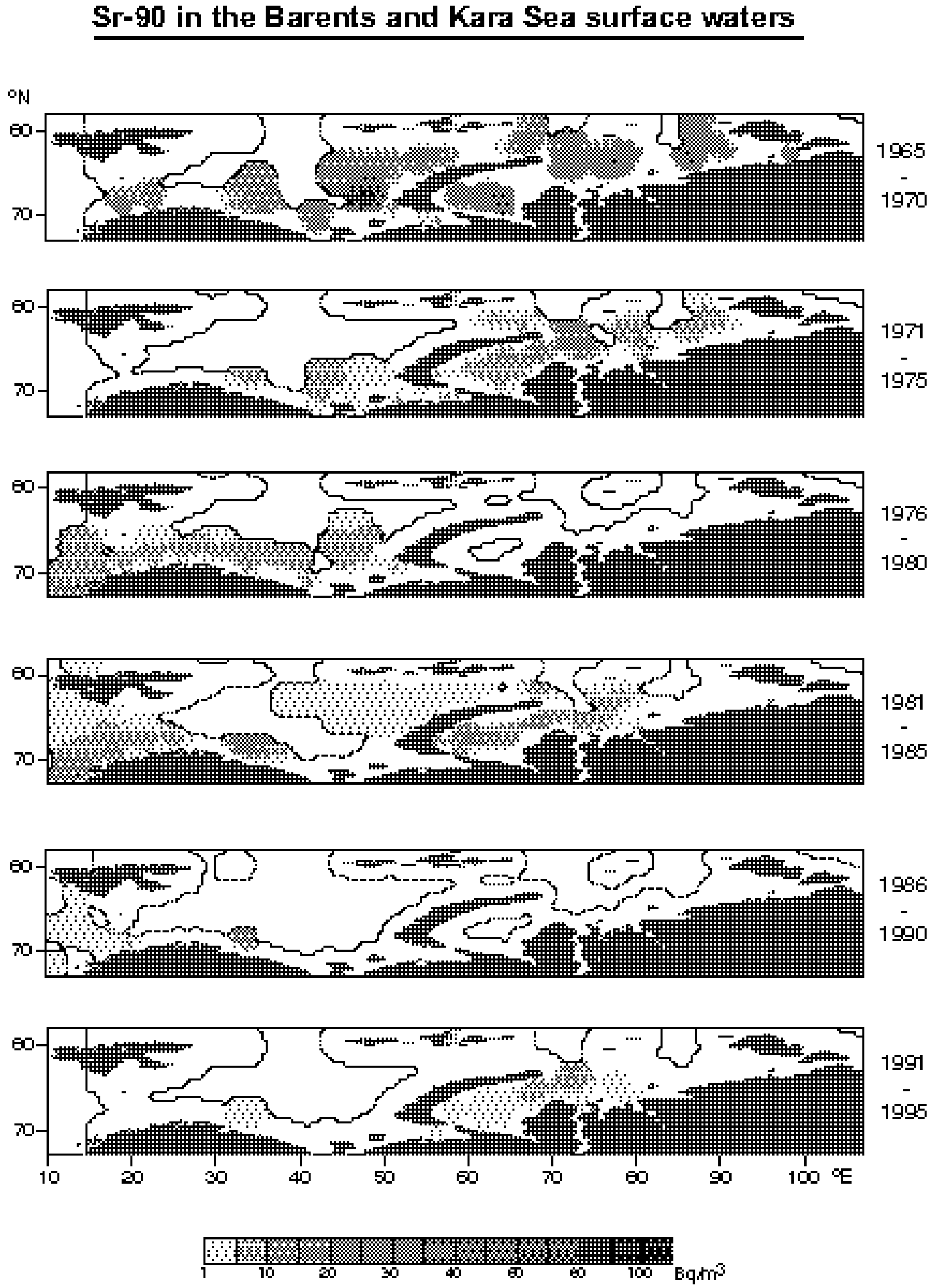


FIG. 14. Contours of ^{90}Sr levels in the Barents and Kara Sea surface waters. White areas: no data available.

content of surface water decreased from 18–27 Bq/m^3 in 1982 in the south-west Kara Sea [49] to 3–8 Bq/m^3 at present. This trend may reflect a considerable decrease in weapons testing fallout and probably also the reduction in ^{137}Cs discharges from the Sellafield reprocessing plant. ^{90}Sr in surface waters shows similar trends, the levels having decreased from 7–21 Bq/m^3 in 1982 [49] to present values of 3–11 Bq/m^3.

In the histogram shown in Fig. 12(b) for the Kara Sea, high ^{90}Sr concentrations (about 39 Bq/m) can be observed in the 1960s, which may be associated with local fallout [52]; medium concentrations (below 15 Bq/m) in 1970, 1971 and 1982; and low concentrations (about 5 Bq/m) have been observed in recent years. The ^{137}Cs data are only available from 1982, and the value for that year (about 20 Bq/m) is consistent with mean ^{90}Sr values. Present concentrations are much smaller (about 6 Bq/m), and therefore one can speculate that the 1982 value may represent the peak Sellafield signal in the Kara Sea. The approximate transit time from Sellafield to the Kara Sea would then be six to seven years. No pre-1992 data are available for $^{239+240}$Pu concentrations in Kara Sea waters.

The spatial distribution of ^{137}Cs in the Barents and Kara Sea surface waters for time intervals in the period 1965–1995 is shown in Fig. 13. The effect of the transport of ^{137}Cs from Sellafield to the Barents Sea (especially in 1976–1980 and 1981–1985) and the recent decrease in ^{137}Cs concentrations can be seen.

Because of the above temporal trends taken in the context of residence time consideration, deeper waters (>50 m) in the open Kara Sea show higher concentrations of ^{137}Cs and $^{239+240}$Pu than surface waters, by a factor of two to three.

The spatial distribution of ^{90}Sr in Barents and Kara Sea surface waters depicted in Fig. 14 also shows a considerable decrease of ^{90}Sr levels in the open Kara Sea. Remarkably higher ^{90}Sr concentrations have been observed in the central and eastern Kara Sea than in the Barents Sea, which may be due to discharges of radioactive wastes from the nuclear installations to the Ob River, to runoff of global fallout from the catchment areas of the Siberian rivers and discharges from reprocessing plants in Western Europe.

3.2.2.2. Sediment

Pre-1992 radionuclide data for sediments in the Kara Sea are very sparse. The highest ^{137}Cs levels (around 200 Bq/kg dry weight (dw)) were observed in the 1970s in the central and eastern Kara Sea. Data from the 1980s show 4–20 Bq/kg dw for ^{137}Cs in surface sediment [49] which can be compared with the present values of 18–32 Bq/kg dw [53–56]. No pre-1992 data are available for $^{239+240}$Pu concentrations in Kara Sea sediments.

3.2.3. Present radionuclide concentrations in the Kara Sea

3.2.3.1. Dumpsites

The extensive sampling and measurement programme carried out in the framework of the Joint Norwegian–Russian Environmental Co-operation conducted radionuclide analyses of water, sediment and biota sampled at the major dumpsites in the Tsivolka, Stepovoy and Abrosimov Fjords and the Novaya Zemlya Trough. The results show that radionuclide concentrations are generally low, similar to those observed in the open Kara Sea. However, very localized contamination of sediment has been observed from leakage of waste containers in the Abrosimov and Stepovoy Fjords (Table VI).

Abrosimov Fjord

An example of a profile of ^{60}Co, ^{90}Sr, ^{137}Cs and Pu isotopes in sediment collected in 1994 at the container dumpsite in Abrosimov Fjord is shown in Fig. 15. The concentrations of all radionuclides detected were higher by a factor of 10 to 1000 than those at uncontaminated sites. Contamination by fission products (^{137}Cs up to 30 kBq/kg and ^{90}Sr up to a few kBq/kg dw), activation products (^{60}Co up to a few hundreds of Bq/kg dw) and actinides ($^{239+240}$Pu up to 18 Bq/kg dw) have been observed [6, 57–60]. The ^{238}Pu/$^{239+240}$Pu activity ratio, a strong indicator of plutonium origin in the marine environment [61], ranges from 0.3 to 0.7. This ratio differs significantly from the value for the global fallout (0.03) and suggests a waste origin for plutonium in the sediment core. The sampling site where the highest contamination is observed does not contain reactor compartments, only low level wastes packed in containers. This observation implies that leakage probably occurred from dumped containers owing to their poor quality. However, the leakage has not led to a measurable increase of activity levels in the outer part of the Fjord. The highly localized character of the contamination suggests that leakage probably occurred in particulate form. Some radioactive particles have also been identified in the sediment samples [6, 60].

Figure 7 shows a map of Abrosimov Fjord with localized dumped objects and ^{137}Cs concentrations measured in surface sediments. The contours of ^{137}Cs levels are calculated on the basis of available data [6, 59, 61, 62]. A similar distribution has been observed for

TABLE VI. RANGE OF CONCENTRATIONS OF RADIONUCLIDES IN KARA SEA WATERS AND SURFACE SEDIMENTS (0–2 cm) (1992–1994) [6, 55, 62]

	^{137}Cs			^{90}Sr			$^{239+240}Pu$			^{60}Co
	Water (Bq/m^3)		Sediment (Bq/kg dw)[a]	Water (Bq/m^3)		Sediment (Bq/kg dw)[a]	Water (mBq/m^3)		Sediment (Bq/kg dw)[a]	Sediment (Bq/kg dw)[a]
Site	Surface	Bottom		Surface	Bottom		Surface	Bottom		
Abrosimov Fjord	4–7	4–9	9–8400	2–4	2–4	0.3–3500	3–7	3–5	1–18	< 1–66
Stepovoy Fjord	3–9	6–32	7–103 000	2–7	3–26	0.4–300	2–5	2–18	0.1–15	0.1–3100
Tsivolka Fjord	4–6	6–14	4–30	4–6	3–4	0.4–1	4–10	5–8	0.03–0.5	< 1–4
Novaya Zemlya Trough	4–7	7–14	7–30	2–3	2–4	0.8	3–4	7–12	1	< 2
Open Kara Sea	3–8	8–20	2–33	3–11	4–6	0.3–0.8	2–8	5–16	0.4–1.3	

[a] dw = dry weight.

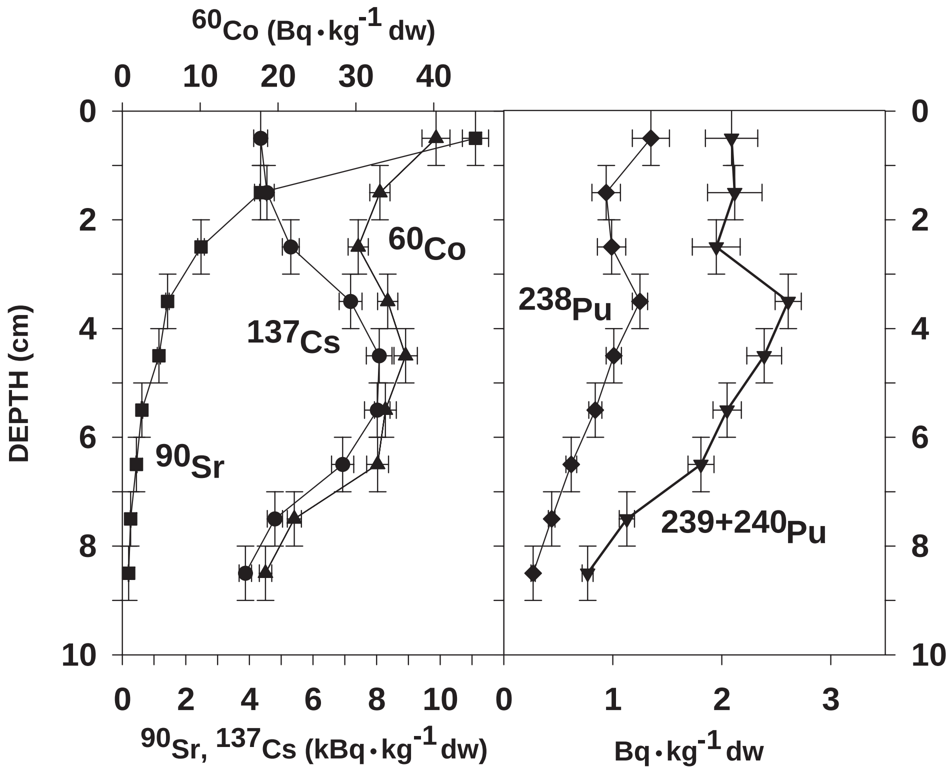


FIG. 15. Radionuclide profiles in a sediment core collected at Abrosimov Fjord in 1994.

^{60}Co, ^{90}Sr and $^{239+240}$Pu. However, ^{3}H, ^{90}Sr, ^{137}Cs and $^{239+240}$Pu levels observed in the waters of the fjord are within the typical ranges for the open Kara Sea. This would imply that leakage was not continuing in 1994 when the water samples were taken.

Radionuclide inventories in sediments for ^{137}Cs are up to 3000 kBq/m^2 at locations where leakages have been observed. Similarly ^{60}Co and $^{239+240}$Pu inventories are up to about 3 and 0.2 kBq/m, respectively [62].

Stepovoy Fjord

Perhaps the single most persuasive piece of confirmatory evidence of background radionuclide levels in the fjord is IAEA-MEL's sea-bed gamma spectrum from the sediment surface at the Stepovoy Fjord dumpsite (Fig. 16). This spectrum obtained by using IAEA-MEL's underwater survey system, which includes a propane cooled HPGe detector [27, 59, 63], is one of the first sets of high resolution gamma spectra ever recorded in situ in the marine environment. The spectrum shows at a glance the predominance of the gamma ray lines from naturally occurring (background) radionuclides, namely from ^{40}K and the U and Th decay series. The only identifiable anthropogenic radionuclide is ^{137}Cs [27].

Higher concentrations of ^{137}Cs, ^{60}Co, ^{90}Sr and Pu isotopes (^{137}Cs up to 110 kBq/kg, ^{60}Co up to 3 kBq/kg, ^{90}Sr up to 0.3 kBq/kg and $^{239+240}$Pu up to 10 Bq/kg) have been measured only at very localized places around dumped containers (Fig. 8). As in Abrosimov Fjord, radioactive particles were present in the contaminated sediments. However, no leakage has led to measurable increases in the radionuclide concentration in the outer part of the Fjord.

Measured concentrations of ^{90}Sr, ^{137}Cs and $^{239+240}$Pu in the inner part of the fjord bottom water are higher by a factor of three to five than in a surface water, indicating a leakage from containers. In the outer part of the fjord, the concentrations are typical of those in the open Kara Sea.

Radionuclide inventories in sediments may vary for ^{137}Cs from below 1 kBq/m^2 in uncontaminated locations to 110 kBq/m^2, suggesting that leakage has occurred. Similarly, ^{60}Co and $^{239+240}$Pu inventories are up to 26 and 0.2 kBq/m^2, respectively.

Tsivolka Fjord

Sediment cores analysed from Tsivolka Fjord have shown ^{137}Cs and $^{239+240}$Pu concentrations comparable to

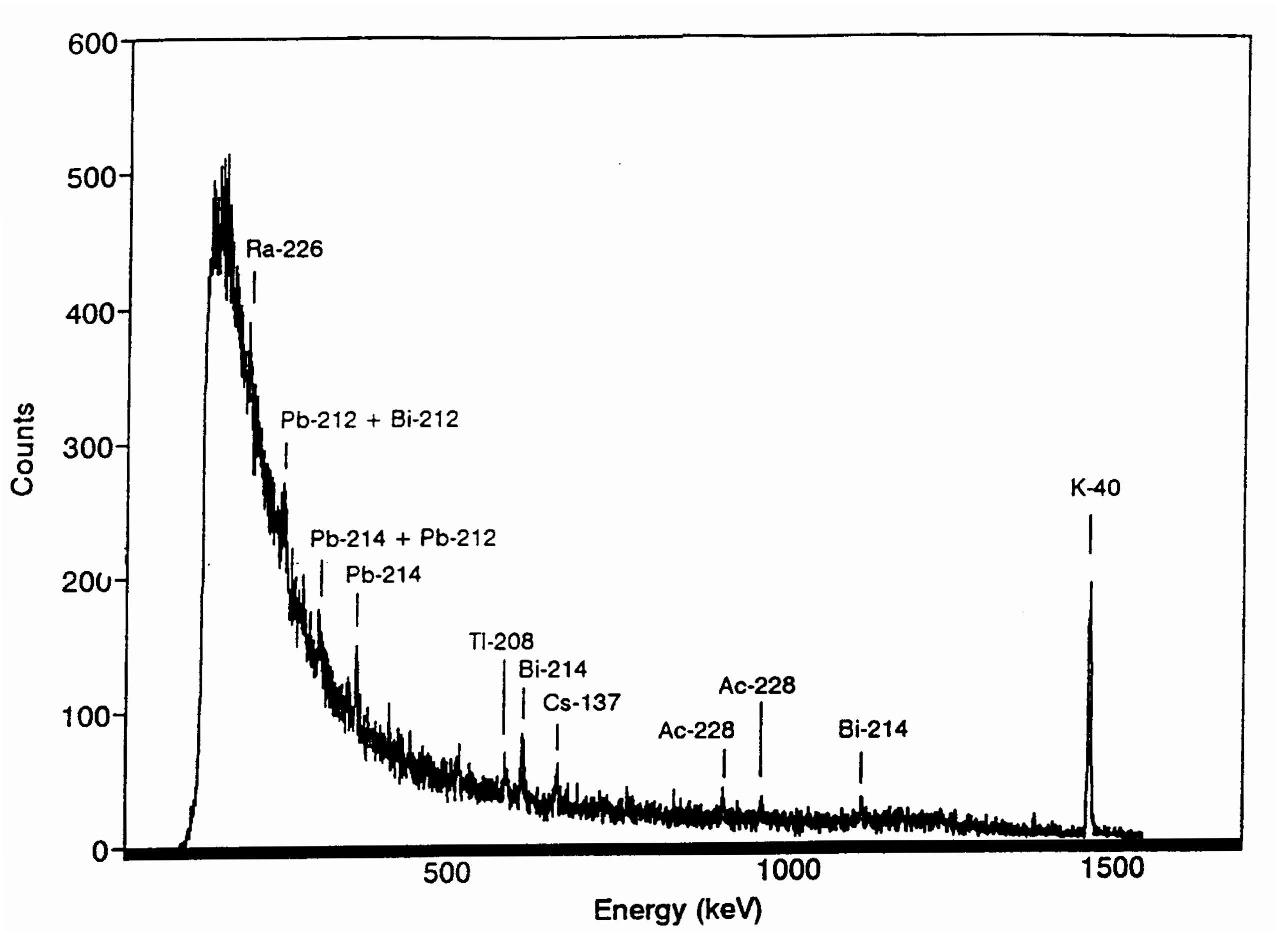


FIG. 16. In situ sea-bed gamma spectrum measured by an underwater HPGe spectrometer in Stepovoy Fjord in 1993. Spectrum accumulation time was 9500 s.

those of open Kara Sea sediment. However, the presence of traces of ^{60}Co suggests a local source of contamination. The observed radionuclide profiles indicate a fast deposition of sedimentary material and effective mixing by physical and/or biological processes.

The spent nuclear fuel of the icebreaker's reactor has not been localized yet, and no possible leakage associated with this waste has been reported (Fig. 9). Radionuclide concentrations in water are typical of the open Kara Sea.

Radionuclide inventories in sediment range from 0.3 to 3 kBq/m^2 for ^{137}Cs, and for ^{60}Co and $^{239+240}$Pu up to some 0.3 and 0.02 kBq/m^2, respectively [62].

Novaya Zemlya Trough

No local contamination of sediment has been observed. However, the nuclear reactor dumped in the Novaya Zemlya Trough has not yet been properly localized (Fig. 10).

As mentioned previously, concentrations of ^{137}Cs and $^{239+240}$Pu are expected to be higher in deep waters. While the ^{137}Cs concentration in surface water varies between 4 and 7 Bq/m^3, bottom water ^{137}Cs concentrations are 7–14 Bq/m^3. ^{137}Cs and $^{239+240}$Pu inventories in sediment are in the range of 0.3–0.8 and up to approximately 0.02 kBq/m^2, respectively [6, 55, 62].

3.2.3.2. The open Kara Sea

The concentrations of anthropogenic radionuclides in Kara Sea water sediment and biota are generally very low. ^{137}Cs data show almost constant spatial distribution. However, ^{90}Sr data, especially the ^{90}Sr/^{137}Cs activity ratio in sea water, vary across the Kara Sea. The ratio correlates inversely with salinity and indicates the importance of the ^{90}Sr contribution from the Ob river.

The ^{137}Cs inventory in the water column ranges from 1 to 4 kBq/m^2 and shows a relatively smooth and linear correlation with water depth. This implies that there are no point radioactive sources near the sampling stations [7, 55].

The sediment column inventories, ranging from 0.5 to 0.9 kBq/m^2, show an inverse correlation with water depth, suggesting an enhanced rate of nuclide scavenging in the higher particle fluxes associated with shallower waters [55].

The total ^{137}Cs inventories in the open Kara Sea range from 1.5 to 4.5 kBq/m^2. In comparison with the amounts of radioactive fallout deposited in the northern hemisphere [64], it can be estimated that the contribution of global fallout to the ^{137}Cs inventory in the Kara Sea should be about 0.5 kBq/m^2. This value is lower than the average measured value weighted over the mean water depth in the Kara Sea (1.5 kBq/m^2), suggesting that the difference may reflect contributions from local fallout and from discharges from the Sellafield reprocessing plant. Indeed, in the Pechora Sea, the observed ^{137}Cs concentrations are much higher [55, 65, 66], reflecting the close proximity of the Chernaya Bay underwater nuclear test site.

^{210}Pb was assayed in order to estimate sediment mixing and accumulation rates and to reconstruct radionuclide levels in the past [55]. The data confirm that mixing plays a dominant role in controlling the radionuclide concentrations in surface sediments of the Kara Sea. The estimated sedimentation rate is from 1 to 4 mm/a. The ^{137}Cs depth distribution is affected by sediment mixing (bioturbation) and suggest post-depositional migration of caesium, as the penetration of ^{137}Cs in sediment is deeper than would be expected from the ^{210}Pb age determination.

The $^{239+240}$Pu inventory in sediments is between 0.01 and 0.03 kBq/m^2. The $^{239+240}$Pu/^{137}Cs inventory ratios are about 0.03. ^{238}Pu/$^{239+240}$Pu ratios in sediments are between 0.02 and 0.05, suggesting a global fallout origin for the plutonium in sediment [55].

3.2.3.3. Ob and Yenisey estuaries

The Ob and Yenisey rivers are important suppliers of fresh water to the Kara Sea and possibly of contaminants (e.g. ^{90}Sr and ^{137}Cs) from land based sources. The data clearly show ^{137}Cs and $^{239+240}$Pu depletion from water at low salinity [67] due to scavenging and fast sedimentation, resulting in relatively high sediment inventories (1–5 kBq/m^2 for ^{137}Cs).

3.2.3.4. Biota

No data are available on pre-1992 radionuclide concentrations in biota of the Kara Sea. From post-1992 data, it can be concluded that typical concentrations of ^{137}Cs in the benthic fauna of the Kara Sea are around 1 Bq/kg dw. ^{137}Cs and $^{239+240}$Pu concentrations in gammarids were found around 1.5 and 0.01 Bq/kg dw, respectively. Brittle stars (*Ophiuroidea*) showed $^{239+240}$Pu concentrations around 0.1 Bq/kg dw. Higher values of ^{137}Cs were observed for Polychaeta (*Spiochaetopterus*) between 6 and 17 Bq/kg in the tubes. For fish (polar cod) caught in the open Kara Sea and in Abrosimov Fjord, ^{137}Cs concentrations were about 1 Bq/kg dw. Algae samples showed ^{137}Cs levels below 3 Bq/kg dw [6, 7, 68].

Generally, the concentrations of ^{137}Cs, ^{90}Sr and $^{239+240}$Pu in biota samples from the Kara Sea are very low (the latter two radionuclides are often below the detection limits of 0.5 Bq/kg and 0.01 Bq/kg dw, respectively).

3.2.3.5. Conclusions on radionuclide concentrations in the Kara Sea

On the basis of the extensive joint Norwegian–Russian data [6, 26, 60] and of IAEA-MEL and other results stored in the GLOMARD database on measurement of concentrations of several radionuclides (^{3}H, ^{60}Co, ^{90}Sr, ^{99}Tc, ^{129}I, ^{137}Cs, ^{238}Pu $^{239+240}$Pu and ^{241}Am) in water, sediment and biota sampled at the major dumpsites in Abrosimov, Stepovoy and Tsivolka Fjords and the Novaya Zemlya Trough as well as in the open Kara Sea and the Ob and Yenisey estuaries, it can be concluded that, with the exception of limited areas in the Abrosimov and Stepovoy Fjords, only minor contamination exists relative to background levels. In situ gamma spectra recorded at the dumpsites indicate that the contamination has a localized character and, at sites outside the disposal areas, no contribution from local sources can be observed [25, 27]. The most marked contamination of sediment appears to be associated with leakage from dumped containers but this is confined to the immediate vicinity of the containers.

The open Kara Sea is relatively uncontaminated, the main contributions being due to direct deposition and catchment runoff from global fallout caused by nuclear weapons tests, discharges from reprocessing plants in Western Europe and the former USSR, local fallout from tests performed at Novaya Zemlya and Chernobyl fallout [69–71].

3.2.4. Intercomparison exercises

The IAEA-MEL organized intercomparison exercises for the laboratories performing analyses on samples collected during the 1992–1994 joint Russian–Norwegian expeditions to the Kara Sea, covering the following relevant categories of environmental matrices:

TABLE VII. IASAP SELECTED K_d COEFFICIENTS FOR MODELLING COMPARED WITH EXPERIMENTAL DATA AND IAEA REFERENCE VALUES

Element	Fjords		Kara Sea[b]	IAEA, 1985 [72][c]	IASAP selected values[d]
	Stepovoy[a]	Abrosimov[a]			
Americium	$(0.1–1) \times 10^6$	$(0.1–3) \times 10^6$	$(0.1–4) \times 10^5$	$1 \times 10^5 – 2 \times 10^7$	1×10^6
Plutonium	$(0.5–1) \times 10^5$	$(0.5–1) \times 10^5$	$(0.2–5) \times 10^5$ [e]	$1 \times 10^4 – 1 \times 10^6$	1×10^5
Cobalt	$(1–2) \times 10^6$	$(1–2) \times 10^3$	$(0.5–5) \times 10^4$	$2 \times 10^4 – 1 \times 10^6$	1×10^6
Europium	$(1–2) \times 10^5$	$(1–2) \times 10^5$	1×10^5	$1 \times 10^5 – 2 \times 10^6$	7×10^5
Strontium	$(0.1–1) \times 10^2$	$(0.1–1) \times 10^2$	$(0.1–5) \times 10^1$ [e]	$1 \times 10^2 – 5 \times 10^3$	1×10^2
Caesium	$(3–6) \times 10^2$	$(2–3) \times 10^2$	0.15×10^2	$1 \times 10^2 – 2 \times 10^4$	5×10^3

[a] Laboratory experiments.
[b] Shipboard experiments.
[c] Coastal sediments.
[d] Used in the model calculations in Section 5, with a few exceptions (see text).
[e] Estimated K_d ranges based on measured radionuclide concentrations in separate water and sediment samples.

TABLE VIII. IASAP SELECTED CONCENTRATION FACTORS FOR MODELLING COMPARED WITH EXPERIMENTAL DATA AND IAEA REFERENCE VALUES

Element	Fish (muscle)	Sea birds	Marine mammals	IAEA, 1985 [72] Fish (muscle)	IASAP Selected values for fish[a]
Plutonium	$1 \times 10^3 – 4 \times 10^3$	$<2 \times 10^1 – 1.5 \times 10^2$	3×10^0	$5 \times 10^{-1} – 1 \times 10^2$	4×10^1
Caesium	$3 \times 10^1 – 3 \times 10^2$	$4 \times 10^1 – 1.1 \times 10^3$	$1.3 \times 10^1 – 1.8 \times 10^2$	$1 \times 10^1 – 3 \times 10^2$	1×10^2
Strontium	$2 \times 10^1 – 9 \times 10^1$	–	$4 \times 10^{-1} – 3.0 \times 10^0$	$3 \times 10^{-1} – 1 \times 10^1$	4×10^0
Nickel	1×10^2	7×10^2	–	$5 \times 10^1 – 1 \times 10^3$	1×10^2
Antimony	–	–	1×10^{-1}	$1 \times 10^2 – 1 \times 10^3$	–
Iodine	–	–	1×10^{-1}	–	–

[a] Used in the model calculations in Section 5, with a few exceptions (see text).

sediment, sea water and seaweed. For each exercise, results were requested to be reported for at least ^{90}Sr, ^{137}Cs and $^{239+240}Pu$ — the basic group of radionuclides which have been routinely analysed and reported for Kara Sea environmental samples. The radionuclide concentrations in the intercomparison samples were representative of the levels generally encountered in the Kara Sea. Detailed results of the intercomparison exercises are reported in the Joint Russian–Norwegian Experts Group Reports on the 1992, 1993 and 1994 expeditions [6, 26, 60].

As these exercises were organized for a small number of laboratories, their purpose was not to define reference values for radionuclide concentrations in the respective samples, but rather to check the agreement between results reported by the participants. On the basis of the intercomparison exercises, it was concluded that, generally, the reliability of the analytical methods used by the participating laboratories was appropriate and that the laboratories produced radionuclide concentration data which were in reasonably good agreement. For all three intercomparison materials, the best concordance amongst individual results was obtained for ^{137}Cs (around 10% standard deviation from the mean for sediment and seaweed, 5% for sea water). Good agreement was also obtained for ^{90}Sr in sea water (10% standard deviation from the mean). Owing to the very low ^{90}Sr concentration in sediment and the small number of results reported for seaweed, it proved difficult to evaluate the participants' performance in these cases. As for $^{239+240}Pu$, the agreement is not as good (up to 25% standard deviation from the mean), reflecting the analytical difficulties associated with its determination in environmental matrices.

3.3. DISTRIBUTION COEFFICIENTS AND CONCENTRATION FACTORS

Obtaining basic information on the potential uptake of contaminants by marine particles and biota is fundamental to an evaluation of the transport pathways and risks posed in the Kara Sea. The concepts of distribution coefficient (K_d) and concentration factor (CF) have been used to intercompare the relative degree of uptake of marine contaminants by different sediment types and marine biota, respectively. K_d coefficients are used in models of radionuclide dispersion to establish the equilibrium partitioning of radionuclides between particulate matter and sea water. Concentration factors are used to establish the partitioning of radionuclides between marine biota and sea water, regardless of the pathways through which these radionuclides are accumulated.

The actual values for K_d and CF are not known for many areas of the ocean. Furthermore, they vary spatially and temporally and are often affected by environmental and biological variables. Most of the information on these two variables has been collected from the temperate regions of the world and has been published by the IAEA [72]. For the Arctic, data are particularly sparse. In this report, K_d coefficients were estimated on the basis of radionuclide and stable element geochemical data and the proportions of the elements in the particulate phase that are likely to be exchangeable with the aqueous phase. CFs were computed by using field measurements of radionuclide and/or stable element concentrations in organisms as well as data derived from laboratory radiotracer experiments.

Concern was expressed that K_d coefficients and CFs for the Arctic may differ considerably from those applicable to temperate regions. A group of consultants evaluated all available information, published and unpublished, on K_d coefficients and concentration factors to fauna in the arctic marine environment. It was concluded that they do not differ significantly from the values set out in the IAEA report [72]. In addition, the IAEA-MEL carried out field and laboratory experiments using sediment from the Kara Sea and the fjords to determine site specific K_d values (details will be reported in an IAEA TECDOC currently in preparation [73]). Selected K_d values developed for the Novaya Zemlya Fjords and the open Kara Sea are given in Table VII. Selected values of CFs developed from measured radionuclide and stable element concentrations in biota and sea water are given in Table VIII for fish, sea birds and marine mammals. In those cases where data are not available, the values in the IAEA report [72] are recommended to be used.

In the various models used in IASAP (see Section 5.3.2) the selected values for the K_d and CF given in Tables VII and VIII have been applied. However, in a few cases, other values have been used.

4. THE RADIOACTIVE SOURCE TERM

4.1. SOURCE TERM DEVELOPMENT

The assessment of the potential risks posed by possible future release from the dumped wastes focused on the high level waste objects containing the majority of the radioactive waste inventory. Thus, the source term comprises a total of eleven nuclear submarine pressurized water reactors (PWRs), four with spent nuclear fuel; three icebreaker PWRs, with spent nuclear fuel from one of them in a separate container; and two nuclear submarine liquid metal reactors (LMRs), both with spent nuclear fuel [74–78]. Three of the PWRs and one of the LMRs are known to contain damaged fuel. In order to provide appropriate release rate scenarios the characteristics of the dumped reactors and their operating histories were examined. These scenarios were chosen to represent the possible range of release rates to the environment: a best estimate scenario and two accident scenarios.

4.1.1. Characteristics of the steam generating installations

In each case, the steam generating installation, including the steam generators and circulation pumps, was located in an isolated reactor compartment. The reactor pressure vessels were aligned vertically and surrounded by a water filled steel shield tank in the region of the reactor core. Additional radiation shields were located above the shield tank and round each reactor.

Each PWR, including that of the nuclear icebreaker Lenin, consisted of a cylindrical carbon steel pressure vessel with an approximate diameter of 1.4 to 2 m, a height of 3.4 to 5 m and walls 100 to 120 mm thick. Nuclear submarine cores were assumed to be loaded with U–Al alloy fuel, where the uranium was enriched to 7.5 or 20%, totalling 50 kg of ^{235}U. The icebreaker cores were loaded with varying quantities of UO_2 sintered ceramic fuel enriched to 5.0% ^{235}U and clad in Zr–Nb alloy or stainless steel.

Each LMR consisted of a cylindrical stainless steel pressure vessel of a diameter of 1.8 m, a height of 3.7 m and walls 30 mm thick. LMR cores were loaded with U–Be sintered ceramic fuel containing 90 kg ^{235}U, a fuel enrichment of 90% and clad in stainless steel.

To reduce heat and radiation effects on each reactor pressure vessel and thus extend their operating lives, stainless steel thermal shields were employed. A summary of currently available information on the dumped steam generating installations is shown in Table IX [74–76, 78].

4.1.2. Reactor operating histories

Currently available information on the operating histories of the nuclear submarine steam generating installations is limited to the years of startup, shutdown and fuel burnup. The longest and shortest periods of steam generating installation operation were of the order of five years and one year, respectively. Fuel burnup for the PWRs varied from a low of 12.5 GW·d to a high of 38.8 GW·d. In the case of the LMRs, fuel burnup for the second core load was only 875 MW·d. Information on the operating history of the icebreaker steam generating installation OK-150 is much more extensive. There were two fuel loads associated with the first installation: the first ran from 1959 to 1962 and the second lasted from 1963 to 1965. The total integrated power productions for the first installation were equal to 40.3 GW·d for the left board (N1 PWR), 32.2 GW·d for the centre (N2 PWR) and 35.5 GW·d for the right board (N3 PWR) reactors.

4.1.3. Radionuclide inventories

Radionuclide inventories associated with the nuclear submarine LMRs and the icebreaker PWRs were based on their detailed core operating histories and calculated neutron spectra. For the nuclear submarine PWRs, lack of information due to classification issues necessitated using the model for the icebreaker steam generating installation. Thus, fission product activities for the nuclear submarine PWRs are based on the quotient of the nuclear submarine and icebreaker burnups. Actinide activities are based on the product of the quotients of the icebreaker and nuclear submarine enrichments, and the nuclear submarine and icebreaker burnups and activation products are based on the calculation procedure similar to the one used for the icebreaker. A summary of the current radionuclide inventory of all dumped reactors and the icebreaker spent nuclear fuel is shown in Table X, and a breakdown by individual isotopes is provided in Table XI [74–76, 78].

It should be noted that the values quoted here are considerably lower than the original values initially provided in the 'White Book' relating to the radioactive inventory of these objects dumped in the Kara Sea [5]. This is because the present study and supporting data

TABLE IX. CURRENTLY AVAILABLE INFORMATION FOR THE STEAM GENERATING INSTALLATIONS OF THE MARINE REACTORS DUMPED IN THE KARA SEA [74–76, 78]

Factory No.	Reactor		Fuel initial conditions		Steam generating installation				RPV[a] with SNF[b]
	Type	Position	^{235}U load (kg)	Enrichment of U (%)	Startup date	Shutdown date	Burnup (MW·d)	Disposal date	
901	PWR[c]	Left board	50	20	1961	1961	1710	May 1965	Yes
	PWR	Right board	50	20	1961	1961	1670	May 1965	Yes
285	PWR	Left board	50	7.5	1961	1964	2780	Oct. 1965	No[d]
	PWR	Right board	50	7.5	1961	1964	2730	Oct. 1965	Yes
254	PWR	Left board	50	20	1958	1962	3080	1965	No
	PWR	Right board	50	20	1958	1962	3880	1965	No
260	PWR	Left board	50	20	1959	1962	1720	1966	No
	PWR	Right board	50	20	1959	1962	1940	1966	No
OK-150[e]	PWR	Left board	129[f]	5	Aug. 1959	Oct. 1965	40 300	Sep. 1967	No
	PWR	Centre line	75[f]	5	Aug. 1959	Feb. 1965	32 200[g]	Sep. 1967	No[h]
	PWR	Right board	75[f]	5	Aug. 1959	Oct. 1965	35 500	Sep. 1967	No
421	PWR	Right board	50	20	1968	1968	1250	1972	Yes
601	LMR[i]	Left board	90	90	Dec. 1962	May 1968	1580[j]	Sep. 1981	Yes
	LMR	Right board	90	90	Dec. 1962	Jun. 1968	1580[j]	Sep. 1981	Yes
538	PWR	Left board	50	20	1961	1963	1680	1988	No
	PWR	Right board	50	20	1961	1963	1440	1988	No

[a] Reactor pressure vessel (RPV).
[b] Spent nuclear fuel (SNF) with thermal shields, hardware and Furfurol(F).
[c] Pressurized water reactor (PWR).
[d] Thermal shields and hardware only.
[e] The icebreaker steam generating installation.
[f] For the second fuel load.
[g] Burnup for the second fuel load was 14 200 MW·d.
[h] Thermal shields, hardware and approximately 60% of spent nuclear fuel were discarded in a special container.
[i] Liquid metal reactor (LMR).
[j] Burnup for the second fuel load was 875 MW·d.

TABLE X. ESTIMATED 1994 RADIONUCLIDE INVENTORIES OF FISSION PRODUCTS, ACTIVATION PRODUCTS AND ACTINIDES IN THE MARINE REACTORS DUMPED IN THE KARA SEA [74–76, 78]

Factory No.	Activity in 1994							
	Fission products		Activation products		Actinides		Total	
	Activity (Bq)	Percentage (%)	Activity (Bq)	Percentage (%)	Activity (Bq)	Percentage (%)	Activity (Bq)	Percentage (%)
901	7.2×10^{14}	15	6.0×10^{12}	0.13	3.4×10^{12}	0.073	7.3×10^{14}	16
285	6.3×10^{14}	14	1.3×10^{13}	0.27	8.1×10^{12}	0.17	6.5×10^{14}	14
254	–	–	9.5×10^{12}	0.20	–	–	9.5×10^{12}	0.20
260	–	–	5.7×10^{12}	0.11	–	–	5.7×10^{12}	0.11
OK-150[a]	1.8×10^{15}	39	2.3×10^{14}	5.0	8.3×10^{13}	1.8	2.2×10^{15}	46
421	2.9×10^{14}	6.1	2.9×10^{12}	0.062	2.8×10^{12}	0.061	2.9×10^{14}	6.2
601	5.3×10^{14}	10	3.0×10^{14}	6.5	3.6×10^{11}	0.008	8.4×10^{14}	18
538	–	–	4.5×10^{12}	0.096	–	–	4.5×10^{12}	0.096
Total	4.0×10^{15}	86	5.7×10^{14}	12	9.7×10^{13}	2.1	4.7×10^{15}	100

[a] The fission products, actinides and 27% of activation products were in a shielding assembly which was discarded in a reinforced concrete and stainless steel container.

TABLE XI. ESTIMATED 1994 INVENTORY OF INDIVIDUAL ISOTOPES IN THE MARINE REACTORS DUMPED IN THE KARA SEA [74–76, 78]

Isotope	Activity in 1994 (Bq)	Percentage of group (%)	Percentage of total (%)
Fission products			
^{3}H	5.2×10^{13}	1.3	
^{85}Kr	3.9×10^{13}	0.97	
^{90}Sr	9.5×10^{14}	24	
^{90}Y	9.5×10^{14}	24	
^{99}Tc	2.8×10^{11}	0.01	
^{125}Sb	1.7×10^{11}	>0.01	
^{129}I	3.9×10^{8}	>0.01	
^{137}Cs	1.0×10^{15}	25	
^{137}Ba^{m}	9.9×10^{14}	25	
^{147}Pm	2.5×10^{12}	0.06	
^{151}Sm	2.4×10^{13}	0.60	
^{155}Eu	9.6×10^{10}	>0.00	
			86
Activation products			
^{14}C	4.9×10^{11}	0.09	
^{60}Co	1.4×10^{14}	25	
^{59}Ni	6.3×10^{12}	1.1	
^{63}Ni	3.4×10^{14}	61	
^{205}Pb	1.9×10^{8}	>0.00	
^{207}Bi	1.7×10^{10}	>0.00	
^{208}Bi	6.2×10^{9}	>0.00	
^{210}Bi	3.4×10^{9}	>0.00	
^{152}Eu	6.0×10^{13}	11	
^{154}Eu	1.1×10^{13}	2.0	
			12
Actinides			
^{238}Pu	1.6×10^{12}	1.7	
^{239}Pu	6.2×10^{12}	6.4	
^{240}Pu	2.7×10^{12}	2.8	
^{241}Pu	7.8×10^{13}	81	
^{241}Am	8.3×10^{12}	8.6	
			2
Total	4.7×10^{15}		100

became available after the publication of the White Book. The White Book provides a figure of 89×10^{15} Bq (2400 kCi) as the total activity in the dumped objects. The present study shows that at the time of dumping, the original inventory for all the radioactive isotopes was actually close to 37×10^{15} Bq. While the present study was being drawn up, the State Institute for Applied Ecology of the Ministry of Environmental Protection and Natural Resources of the Russian Federation (SIAE) provided to the IASAP an alternative inventory using different assumptions for submarine reactor configurations [79]. After examination, the IASAP team decided to continue to use the data from the original reports, even though the SIAE results tended to be higher and, therefore, more conservative. The reason for continuing to use the original data was first and foremost that the core models used in the IASAP estimate for the icebreaker and submarine factory No. 601 represent the actual configurations, whereas the SIAE models do not. Secondly, even though there are differences between the core configurations of the nuclear submarine and icebreaker PWRs, the icebreaker model is more representative of the true core configurations than the WWER-1000 model used by the SIAE.

4.1.4. Disposal operations

4.1.4.1. Submarine pressurized water reactors

With the exception of the one PWR from submarine factory No. 421 and two from No. 538, all nuclear submarine PWRs were dumped in their separated reactor compartments. Before disposal, the primary circuit loops and equipment of all PWRs were washed, dried and sealed. However, there is no indication that the seals were hermetic. The four reactor pressure vessels containing spent nuclear fuel were filled with Furfurol(F), a hardening compound containing epoxy polyamide, a mixed filler and furfuryl alcohol.

4.1.4.2. Submarine liquid metal reactors

Spent nuclear fuel remained in the two LMRs. Before disposal, a number of actions were taken to secure the reactors and reactor compartment for disposal including the use of some 2 m^{3} of the Furfurol(F) and 250 m^{3} of bitumen [76]. The submarine was towed to Stepovoy Fjord in 1981 and sunk at a depth of some 30 m.

4.1.4.3. Icebreaker pressurized water reactors and the fuel container

Initially, all undamaged and damaged spent nuclear fuel was removed from the three reactor pressure vessels of the icebreaker Lenin. However, as a consequence of an accident which occurred in 1965 during repair works, only 94 of the 219 technical fuel channels from the centre (N2) reactor vessel could be removed for normal disposal. The N2 core barrel, containing the remaining 125 technical fuel channels and all five thermal shields, was removed and placed into a cylindrical shielded containment vessel. This container consisted of two concentric steel cylinders with a 550 mm layer of concrete between them. The core barrel and the spent fuel of the N2 reactor (Configuration A), now in the container, was surrounded with Furfurol(F). The container, hereafter called Container B (or the fuel container), was then hermetically sealed and stored on land for two years.

Before disposal, the reactor pressure vessels and their primary circuit loops and equipment were washed, dried and sealed, and the ceiling of the reactor compartment was equipped with special pressure relief valves. Container B, with the spent fuel, was transferred from its land storage location to a specially prepared stainless steel caisson, hereafter called Container C, on a floating pontoon. Once Container B had been placed in Container C, it was surrounded by Furfurol(F), and Container C was sealed.

In September 1967, the icebreaker and a pontoon carrying Container C with the spent nuclear fuel were towed to Tsivolka Fjord on the coast of Novaya Zemlya. There the reactor compartment and pontoon were dumped approximately 1 km apart at depths of 50 m.

4.2. MODELLING STRATEGY [78]

4.2.1. Methods and assumptions

Radionuclide release from the dumped steam generating installations was assumed to be driven by corrosion of the materials forming the reactor structure and nuclear fuel [80]. Applying the best available predictions for corrosion rates in an Arctic environment [81–84], models were then developed to predict the release rates of the fission product, actinide and activation product inventories in the reactors. Using the inventory and construction data, corrosion rates were applied to simple computer models of the protective barriers to produce radionuclide release rates for three scenarios labelled A, B and C:

(1) Scenario A: the ‘best estimate’ discharge scenario.
(2) Scenario B: the ‘plausible worst case’ scenario, a situation where a disruption, e.g. collision or munitions explosion, causes a complete breach of the containment surrounding the spent nuclear fuel from the icebreaker. This is assumed to occur at the year 2050 under this scenario.
(3) Scenario C: the ‘climate change’ scenario refers to a major environmental disruption where global cooling followed by glaciation scours out the fjords. Subsequent warming would then release activity directly into the Kara Sea from the affected or crushed reactor cores. This release is assumed to occur in the year 3000, one thousand years from now, under this scenario.

Scenario A provides the release rate produced by corrosion processes alone; Scenario B, applied only to the icebreaker steam generating installation, estimates corrosion processes up to the year 2050 followed by a catastrophic release of all remaining radionuclides; and Scenario C models corrosion up to the year 3000 followed by a sudden release caused by a glacier riding over the dumped material. In each case Scenario C is identical to Scenario A up to the year 3000, when a release peak occurs and all remaining radionuclides are released.

It was assumed in all models that all material corroded is immediately released to the environment. This is a highly conservative assumption, as most of the corroded material will be heavy and insoluble and hence will remain inside the hull or reactor pressure vessel until corrosion is well advanced. Much of the released material will be absorbed by the surrounding seabed sediments. These assumptions ensure that the IASAP models are a ‘worst case’ estimate of release rates.

4.2.2. Model construction

Construction details of the steam generating installations were analysed to determine the likely ingress routes for sea water, and the time at which it occurs after dumping. For all installations, corrosion of the outer surface of the reactor pressure vessel and reactor components begins immediately after dumping, as the reactor compartments are free to flood. The first release is therefore almost immediate, with activation products released at the rate of corrosion of the carbon steel making up the components of the steam generating installation; this release is dominated by ^{60}Co and thus rapidly decays over the first 25 years after dumping.

Once sea water has gained entry to the interior of the reactor, the stainless steel cladding on the inside surface of the reactor pressure vessel and the thermal shields, which contain most of the activation products, will corrode at an *effective corrosion rate.* It is a function of actual material corrosion rate and the effectiveness of the containment barriers; this is discussed in Section 4.2.3.1. Fission product and actinide release will begin as soon as water reaches the fuel. Fuel pin cladding has been ignored in all cases since, with considerable fuel damage, the cladding would not present any barrier to the corrosion and release of fuel material. Radionuclide release will then continue until all active material has gone. Since model time-scales were of the order of hundreds or thousands of years, any radionuclide in the core inventories with a half-life of a year or less was ignored unless there was a case of an isotope pair in secular equilibrium.

4.2.2.1. Submarine pressurized water reactors

For the submarine PWRs, the fastest ingress routes into the core are:

(1) via the control rod Furfurol(F) filling vent into the top of certain of the control rod channels;
(2) via the primary circuit inlet and outlet tubes, which had been cropped and welded shut with a 10 mm steel plate; and
(3) ingress into the control rod channels, via the 10 mm steel caps on the control rod channels.

Later ingress routes include reactor pressure vessel head welds and general corrosion of the pressure vessel structure.

For the U–Al alloy fuel assemblages, a faster rate of release was given to some of the more mobile radionuclides of the fission product inventory, typically 20% of the total activity.

4.2.2.2. Submarine liquid metal reactors

After the elliptical sealing cap over the reactor and bitumen filler has failed, the initial release of activation products is produced by corrosion of the thermal shields due to water ingress through the emergency cooling tubes. Some early release of fission products and actinides occurs from the port steam generator, which contains damaged fuel from the left board reactor. Water ingress to the core is initially via the emergency protection rod channels, and later by external corrosion of the core once the thermal shields have disappeared. Activation products in the Pb–Bi heat transfer medium in the steam generators and core are also released.

4.2.2.3. Icebreaker pressurized water reactors and the fuel container

Although the geometry of the structure containing the spent nuclear fuel and the core barrel from the N2 reactor is complicated, modelling the release rates was relatively straightforward [80]. Using pitting corrosion rates applicable to the material of each container acting on the lid welds, it has been assumed that no radioactive material is released from the whole unit until the retaining weld of the Container C lid has corroded off, followed by the retaining weld of the Container B lid. Once seawater penetrates Container B, the five stainless steel thermal shields within the core barrel of the N2 PWR and the oxide fuel begin to corrode.

The damaged fuel from the N2 PWR was UO_2 pellets, enriched to 5%, 4.5 mm in diameter, of a density of 10 g/cm^3, enclosed in Zr–Nb alloy cladding. As soon as the fuel is in contact with sea water, an immediate release of 20% of the fission product and actinide inventory is assumed from the fuel grain boundaries. The remaining fission products and the actinides are released through dissolution of fuel grains at a pessimistic rate of 30×10^{-7} $g \cdot cm^{-2} \cdot d^{-1}$ [84]. For fuel pins with a 4.5 mm diameter and 10 g/cm^3 density, this corresponds to a constant dissolution rate of 0.0011 mm/a.

4.2.3. Corrosion rates

The model results obtained by the IASAP study rely mainly on predicted corrosion rates for materials used in the construction of the steam generating installations. Estimates of corrosion rates were derived from many sources and, in almost all cases, provided ranges of values. Where relevant, pitting corrosion rates in welded material were used. To generate a workable set of models, a single value of corrosion rate at the conservative (i.e. fast) end of the range was defined as the *best estimate corrosion rate* for each material except the filler substances, and this set of corrosion rates was used to obtain release rates. Analysis of the sensitivity of the calculated release rates to the best estimate corrosion rate was then carried out by applying different corrosion rates to the models and obtaining a set of release rate curves; this is further explained in Section 4.3 (results and analyses).

Little information was available about the long term behaviour of the filler substances in sea water and under irradiation. Bitumen is known to become brittle and crack below room temperature, and concrete is, under almost all conditions, porous to water. Since Furfurol(F) is a patented material, detailed information about its make-up and behaviour was not available to IASAP, and long term behaviour is in any case difficult to predict. It is known that the compound Furfurol(F) is a mixture of epoxy resin, amine type solidifier, mineral filler, shale distillate and furfuryl alcohol [85]. An effective lifetime of 500 years is quoted in the White Book [5] for this material. However, in the absence of reliable data on the performance of Furfurol(F) in such environmental conditions, a conservative life span of 100 years in the radiation environment is assumed in the model, and this was

TABLE XII. CORROSION RATES AND FILLER LIFETIMES USED FOR IASAP STEAM GENERATION INSTALLATION MODELS [81–84]

Material	Best estimate corrosion rates (mm/a)	Filler lifetime[a] (a)	Biofouling factor[b]	Sensitivity ranges (mm/a) (a)
Stainless steel (bulk)	0.020		2(1–3)	0.01–0.1
Stainless steel (pitting)	0.50	–	–	0.25–0.75
Mild steel (bulk)	0.075		2(1–3)	0.038–0.11
Mild steel (pitting)	0.166	–	–	0.08–0.33
U–Al alloy	0.03[c]	–	–	0.015–0.045
UO_2	0.0011	–	–	0.0001–0.01
U-Be ceramic[d]	0.001	–	–	0.0001–0.01
Pb-Bi coolant	0.01	–	–	0.001–0.1
Boron carbide	0.01	–	–	0.005–0.05
Europium hexaboride	0.01	–	–	0.001–0.1
Bitumen[e]	–	100	–	50–500
Furfurol(F)[e]	–	100	–	25–500
Concrete[e]	–	100	–	50–500

[a] Lifetime is the period during which the filler provides a physical barrier.
[b] For steels, bulk corrosion rates at outer surfaces were increased by a factor of two to account for biofouling.
[c] For fission products with low solubility and actinides; rate used for soluble fraction of fission products is ten times the slow rate.
[d] Uranium–beryllium alloy in beryllium oxide ceramic matrix.
[e] Filler materials were given a lifetime in preference to a corrosion rate.

supported by a preliminary evaluation carried out at the Kurchatov Institute [86]. Hence, for the calculation of the source term, it was assumed that, at the time of dumping, the fillers were fully effective as barriers to water ingress and radionuclide release, but quickly began to degrade by shrinkage, embrittlement and cracking and would eventually become ineffective after a 100 year lifetime.

Best estimate corrosion rates and degradation rates for materials used in reactor construction are summarized in Table XII. Also included are the parameter ranges applied in the sensitivity analyses.

4.2.3.1. *Containment barriers*

In most cases, the active material contained in the steam generating installation was enclosed by an outer containment barrier such as the submarine hull or reactor pressure vessel. Although breached and allowing ingress of sea water, these containment barriers were assumed to provide a restriction in the oxygen transport into the reactor interior, slowing the true corrosion rate. This effect was modelled by introducing a factor k_c, where:

$$k_c = \frac{\text{area of breach in containment}}{\text{total surface area of containment}} \quad (1)$$

The value of k_c will vary through the corrosion process; for example, most of the PWRs were dumped with their reactor compartments. Their bulkhead penetrations had been sealed with steel end plates. When these corrode and allow ingress, a stepwise increase in the value of k_c models the increase in size of the breach. The factor k_c is scaled to give a value of unity once bulkhead and steel plates have fully corroded. Where the corroding material is surrounded by more than one barrier, the k factors are multiplied together to provide a cumulative factor.

Furfurol(F) has the effect of inhibiting the radionuclide release from the spent nuclear fuel encased in it by the factor k_f. The multiplying factor k_f is zero until the material is dumped, and then ramps to unity over the ensuing 100 years:

$$\begin{aligned} k_f &= 0.01\ t \text{ when } t \le 100 \text{ years} \\ k_f &= 1 \text{ when } t > 100 \text{ years} \end{aligned} \quad (2)$$

where t is the time in years from commencement of corrosion.

Thus the effective corrosion rate for a material inside a Furfurol(F) filled reactor pressure vessel is:

$$X_{eff} = k_c k_f X \quad (3)$$

where

X_{eff} is the effective corrosion rate (mm/a);
k_c is the k factor for reactor pressure vessel containment;
k_f is the k factor for Furfurol(F); and
X is the best corrosion rate (mm/a).

Concrete was used in the dumping of the spent nuclear fuel from submarine factory No. 421. Again, with little information on the type of concrete or its behaviour in sea water, a factor $k_{concrete}$ with values identical to that of k_f was introduced to represent the effect of the concrete barrier. The reactor compartment of the dumped submarine factory No. 601 was partly filled with bitumen before dumping, and the model for release from this unit used a factor k_b, again with a value identical to k_f.

4.2.3.2. Release rates of spent nuclear fuel

Once sea water has entered the core region of a reactor vessel with spent nuclear fuel, corrosion of the cylindrical fuel pins will occur, allowing a release of fission products and actinides. The release rate will depend on the size of the fuel pin. The volume of fuel alloy released per year will be given by

$$v(t) = 2\pi h X_{eff} (R - X_{eff} t) \qquad (4)$$

where

v(t) is the volume of spent nuclear fuel released per year (mm^3/a);
h is the height of the fuel pin (mm);
X_{eff} is the effective corrosion rate, as defined in Eq. (3) (mm/a);
R is the initial fuel pin radius (mm); and
t is the time from commencement of corrosion (a).

The activity of fission products and actinides released per year can be obtained from:

$$a(t) = \frac{A(t)v(t)}{V} \qquad (5)$$

where

a(t) is the activity released per year (Bq/a);
A(t) is the total activity of all the material, corroded and uncorroded, at time t (Bq); and
V is the total initial volume of the material (mm^3).

Expressing V in terms of R and h, and using Eq. (4), the activity release of fission products and actinides can be shown to be:

$$a(t) = A(t) \frac{2X_{eff}}{R^2} (R - X_{eff} t) \qquad (6)$$

where all designations have been defined previously.

4.2.3.3. Release rates of activation products

The release rates of activated material from steam generating installation material were calculated by applying effective corrosion rates to simplified geometries representing the structure in question. For the submarine and icebreaker PWRs, the majority of the activation products came from the thermal shields, the reactor pressure vessel cladding and the reactor pressure vessels themselves; these were modelled by using plane geometry for simplicity.

In the case of the LMRs, the bulk and complex geometry of the thermal shields of steam generating installations and ingress routes required a more detailed analysis, and corrosion rates were modelled by using a variety of circular corrosion geometries applied to the core, thermal shields and steam generators.

4.2.4. Release scenarios

4.2.4.1. Submarine pressurized water reactors

Radionuclide release rates for Scenarios A and C for all the dumped submarine PWR steam generating installations are shown in Figs 17–22. Figure 17, unit 421, for example, shows the release of fission products, actinides and activation products, together with the sum total release rate, from the time of dumping in 1972 until the year 3710 when the last of the steel, with its associated activation products, has corroded away. In this section all calculated events are given to a precision of ±5 years (e.g. 1972 would be treated as 1970).

Unit 421 was encased in concrete before dumping. As discussed earlier, a slowly degrading lifetime of 100 years was assumed for this containment barrier. A similar lifetime was assumed for the Furfurol(F) encapsulating the spent nuclear fuel. As the concrete barrier becomes more and more porous, activation products are released from the outside of the reactor vessel. Then the breather hole into the interior of the vessel is corroded open in the year 2005, allowing fuel and interior stainless steel corrosion to begin. The other reactor vessel penetrations and barriers begin to open up in year 2035, shown by the peak in release rates for the fission

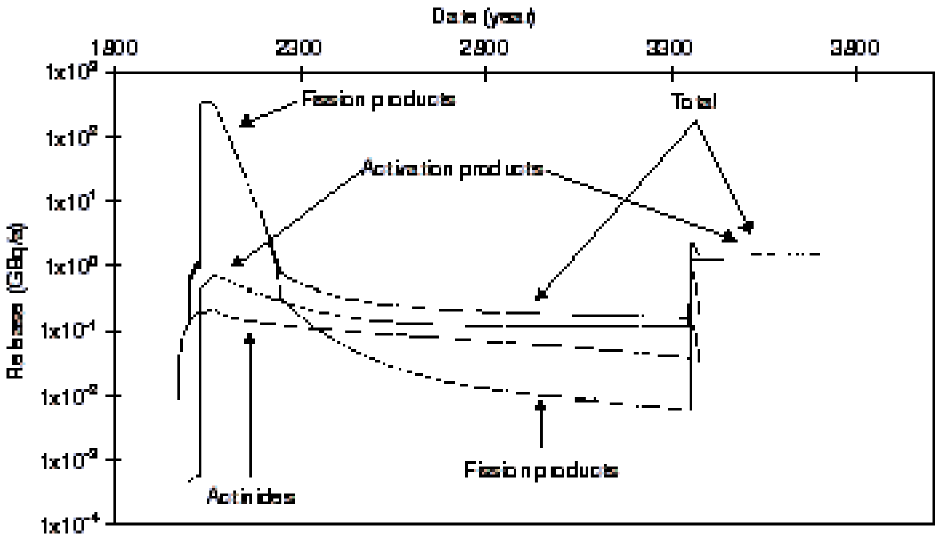


FIG. 17. Novaya Zemlya Trough, unit 421, total radionuclide release for Scenario A.

products (370 GBq/a) and actinides (0.2 GBq/a). Coupled with the continuing steel corrosion, the total peak release rate is approximately 370 GBq/a.

After this, the fission product release rate falls quickly as the more soluble atoms are released, until the year 2500. Subsequently, the rate follows the less soluble elements in the fission product inventory. The actinide release follows this slow degradation rate; as time goes on, however, the overall rate of radionuclide release is dominated by the activation products. Between the years 2500 and 3360, the release rate is of the order of 0.2 GBq/a.

In the year 3360, the reactor pressure vessel has finally disintegrated; without this containment the slow effective rate of corrosion of the interior of the structure is replaced by the faster base corrosion rate, and a short peak in fission and activation product release occurs, totalling 2.5 GBq/a. After that, the fuel rapidly disappears and is gone by the year 3385.

The internal reactor pressure vessel cladding and the thermal shields are finally corroded away by the year 3710; in this last period, the release rates are dominated by the long lived activation products at 1.6 GBq/a.

Figure 18 shows the same situation for unit 421, but for Scenario C, in which the disruption due to glacial scouring occurs in the year 3000. All the original processes detailed above occur until 3000. Then, a total of 630 GBq is released into the Kara Sea in that year from what remains of the unit.

Figures 19 and 20 show the same scenarios for the units in Abrosimov Fjord. These diagrams are a summation of the releases from all the units, with and without spent nuclear fuel, and take into account their different dumping dates. The units were dumped in their reactor compartments, which represents an outer containment barrier, and the units with spent fuel have an interior protection of Furfurol(F).

In Fig. 19, Scenario A, the peak release comes in the year 2040, with a total of 2700 GBq/a. After that, the fission product contribution to the release decreases, and the activation products begin to dominate the total release. By the year 2350, the rate is of the order of 3.3 GBq/a until the year 2690, when the reactor pressure vessels have corroded away, allowing the fuel release peak to be seen again. The fuel has disappeared by 2740 and the remaining steels by the year 3075.

Figure 20 shows Scenario C release rates for the units in Abrosimov Fjord. This shows that the fuel has already disappeared and only the activation products at 880 GBq remain to be released into the Kara Sea in the year 3000. Comparing Fig. 18 for unit 421 in Novaya Zemlya Trough with Fig. 20 for all units in Abrosimov Fjord shows how the concrete barrier of unit 421 has extended the life of the fuel material into the glacier scenario.

Figures 21 and 22 show the situation of unit 538 in Techeniye Fjord. Here, the unit was dumped without spent nuclear fuel in the year 1988 and only the activation products in the steels corrode away, causing a release. Starting at 0.8 GBq/a, the release rate falls to 0.06 GBq/a in the year 2655 when the reactor pressure vessel finally disappears. Afterwards, only the cladding and the thermal shields are left and, by 3075, they too have corroded away. The final release rate is 0.4 GBq/a. Figure 22 shows the peak in the year 3000 owing to the disintegration of the remaining material through glacier scouring.

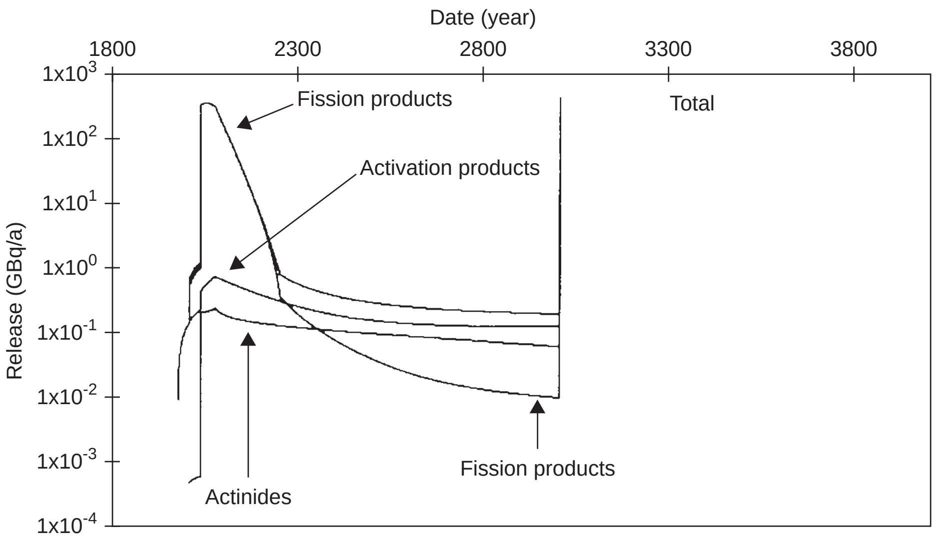


FIG. 18. Novaya Zemlya Trough, unit 421, total radionuclide release for Scenario C.

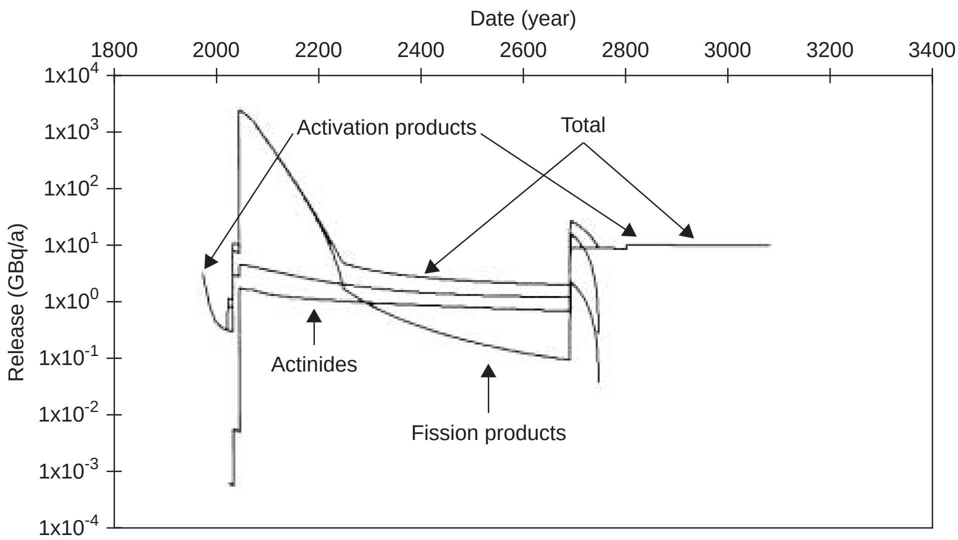


FIG. 19. Abrosimov Fjord, units 901, 285, 260 and 254, total radionuclide release for Scenario A.

4.2.4.2. Submarine liquid metal reactors

Scenario A release rates for unit 601 are shown in Fig. 23. Initially, no active material is expected to appear from this unit dumped in 1981, because of the hull and bitumen barriers. When releases start, the initial fission product and actinide release are less than 1×10^{-4} GBq/a at 2105 when the fission product and actinide inventories in the corroded left board steam generator begin to appear. Corrosion of the outer surfaces of the activated reactors by that time contributes about 8 GBq/a.

By the year 2180, fission products and actinides from the damaged and undamaged reactor cores join the release stream, and the total release rate is of the order of 5 GBq/a. By the year 3000, the rise in release rate of the nuclear fuel and thermal shields of the reactors, caused by the expanding circles of corrosion, exceeds the fall due to decay, and the release rate rises until the year 5200 when the emergency cooling tubes merge to form an annulus. The release rate then varies as the shields external to the emergency cooling tubes disappear by 6800 and the left board steam generator loses all its nuclear fuel by year 7500. By this stage, the release rate has fallen to 0.07 GBq/a.

The release rate remains steady until year 11 400 when the thermal shields disappear. Release rates then continue at 0.004 GBq/a from the fuel and the lead–bismuth coolant, as the reflector is attacked by

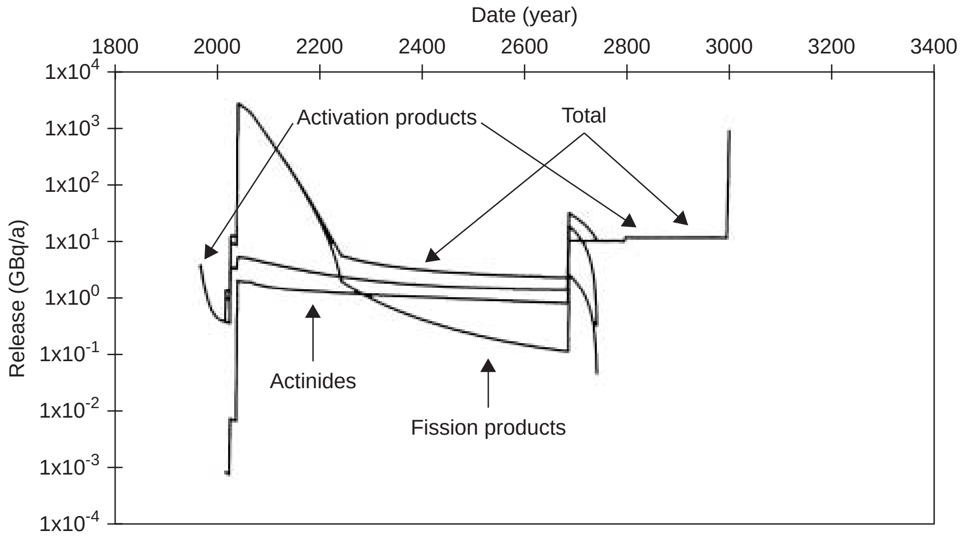


FIG. 20. Abrosimov Fjord, units 901, 285, 260 and 254, total radionuclide release for Scenario C.

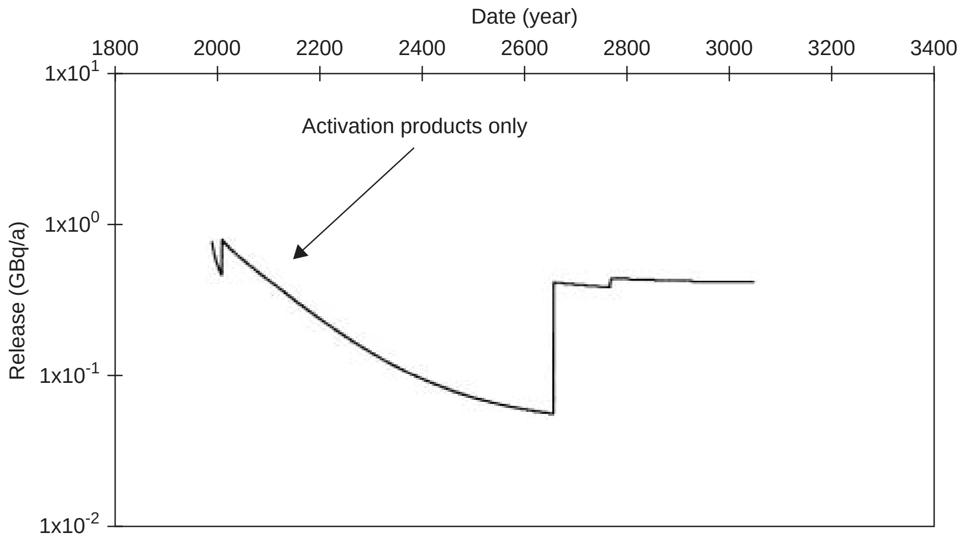


FIG. 21. Techeniye Fjord, unit 538, total radionuclide release for Scenario A, only activation products present.

corrosion. A small rise occurs in the year 18 400 as the reflector disappears and the water is able to attack the outside of the core. Release then continues at a rate of 0.007 GBq/a until all the nuclear fuel has gone at the year 36 580.

Scenario C for unit 601 is shown in Fig. 24. The peak in the year 3000 is of the order of 2100 GBq.

Despite the prediction from the model that the first release will not occur until early in the 21st century, a sample of sea-floor sediment was taken by the Joint Norwegian–Russian Expert Group in 1993 showing europium isotopes ^{152}Eu and ^{154}Eu and higher than expected levels of ^{137}Cs [6]. As the sample site was close to the unit 601 submarine, the report suggested that these radionuclides might have originated in the LMRs.

Although this is a possibility, the first predicted pathway into the damaged spent nuclear fuel is via the welded cap of the steam generator. To open this path requires the corrosion of two stainless steel welds of 20 mm as well as the degradation of the Furfurol(F) inside the steam generator and the bitumen surrounding the whole steam generating installation. The spent fuel has, however, a mixture of fission products and actinides, including ^{155}Eu and ^{137}Cs.

A larger source of europium, but with no caesium, lies in the control rod channels of the reactor pressure vessel structure itself. Europium was used as a neutron

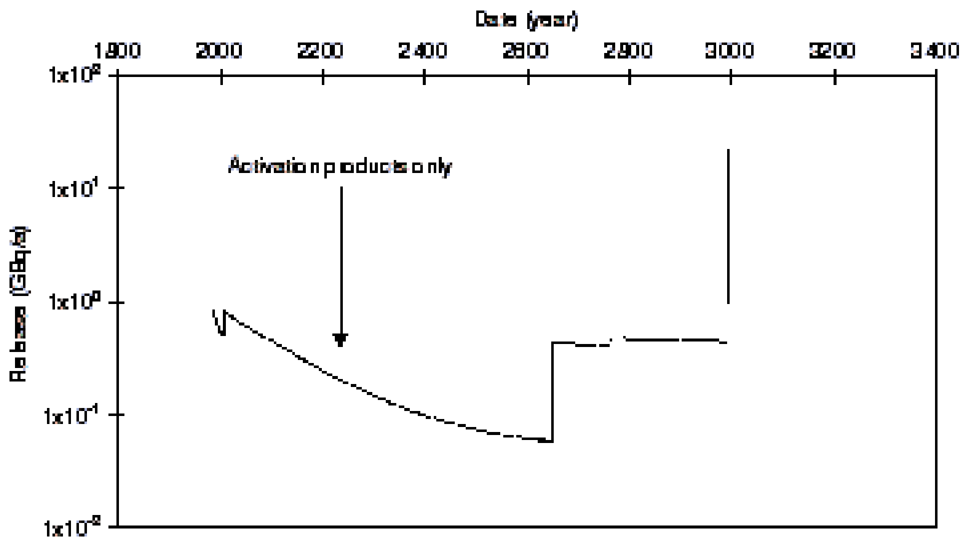


FIG. 22. Techeniye Fjord, unit 538, total radionuclide release for Scenario C, only activation products present.

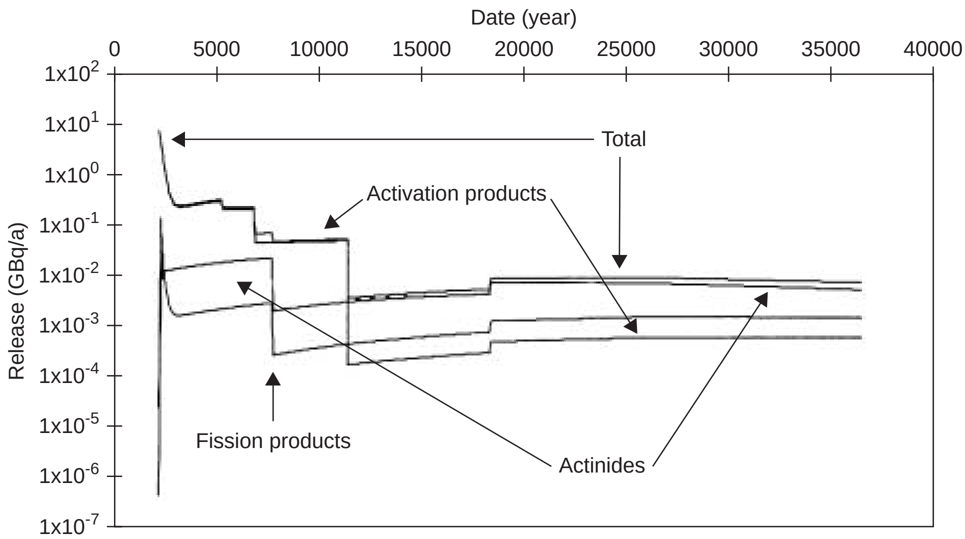


FIG. 23. Stepovoy Fjord, unit 601, total radionuclide release for Scenario A.

absorber. This pathway, however, is not predicted to come into effect until the end of the 21st century.

The model results do not agree with observation. This might be because:

(1) The reactor compartment was contaminated by ^{152}Eu, ^{154}Eu and ^{137}Cs before disposal;
(2) The europium and caesium in the sediment come from another source;
(3) The bitumen and Furfurol(F) barriers have been breached and corrosion of the steam generator, reactor cap and control rod cap welds has been much faster than anticipated; the steam generating installation is in fact leaking.

If (1) or (3) were the case, other isotopes from the LWR would have been observed in the sediment sample. It is suggested that the observations are explained by contamination coming from another source in Stepovoy Fjord.

4.2.4.3. Icebreaker pressurized water reactors and the fuel container

Release rates for fission products, actinides and activation products into Tsivolka Fjord are shown in Figures 25 to 30. As the dumped reactor compartment and fuel container are resting close to each other in the Fjord, the release rates shown are of both units added together.

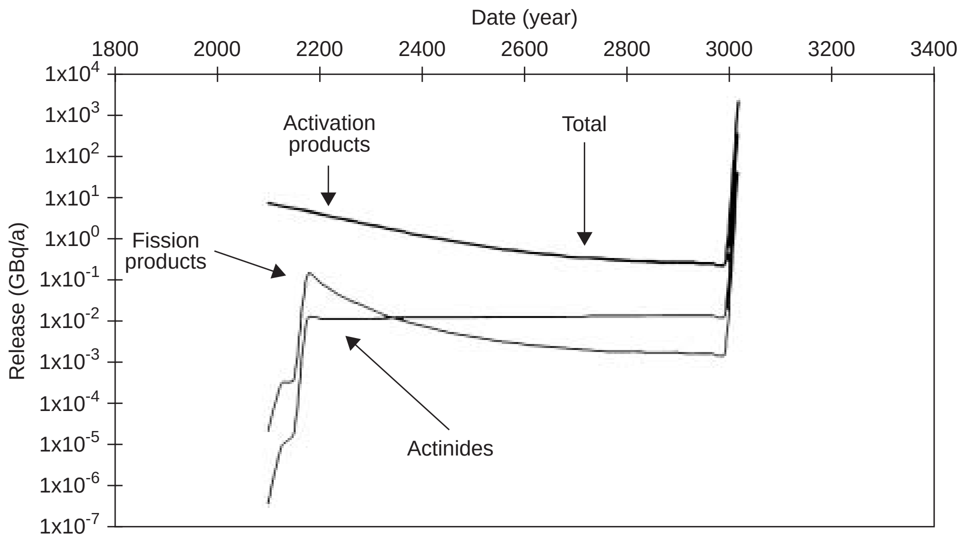


FIG. 24. Stepovoy Fjord, unit 601, total radionuclide release for Scenario C.

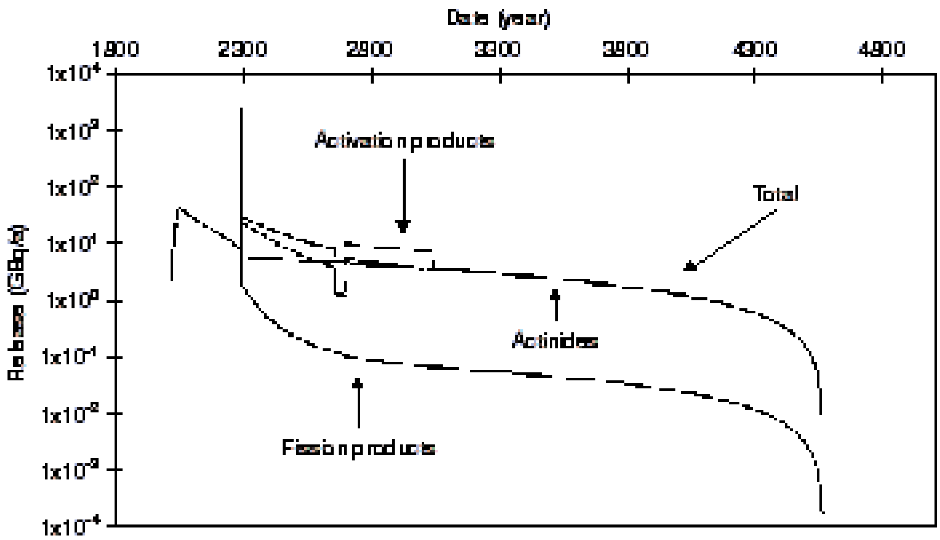


FIG. 25. Tsivolka Fjord, icebreaker reactor compartment and fuel container, total radionuclide release for Scenario A.

Fission product and actinide release

Figure 25 shows the overall picture of the Scenario A release. Figs 26 and 27 break this down to illustrate the fission product and the actinide releases separately.

Figures 26 and 27 show the peak generated as the containers on the pontoon are finally breached in the year 2305. 500 GBq of fission products and 1.6 TBq of actinides are immediately released to the Kara Sea from the cracks and porosity of the damaged fuel. In the following year, the rate of release reverts to the calculated corrosion rate of the oxide fuel, and fission product release is 1.7 GBq/a and actinides 5.7 GBq/a. The fuel slowly corrodes away and the activity of the fuel itself decreases; in the year 3300, the release rate for fission products is 50 MBq/a, and for actinides it is 2.7 GBq/a. The fuel has finally disappeared by the year 4570.

Activation product release

The graph in Fig. 28 for the activation product release rate is more complicated as various k factors come into play on the two dumped units. Again using Scenario A, the hull surrounding the reactor pressure vessel compartment is first breached in the year 2030, commencing outer wall pressure vessel corrosion release, with a small contribution from material inside the pressure vessels through gaps which are assumed to

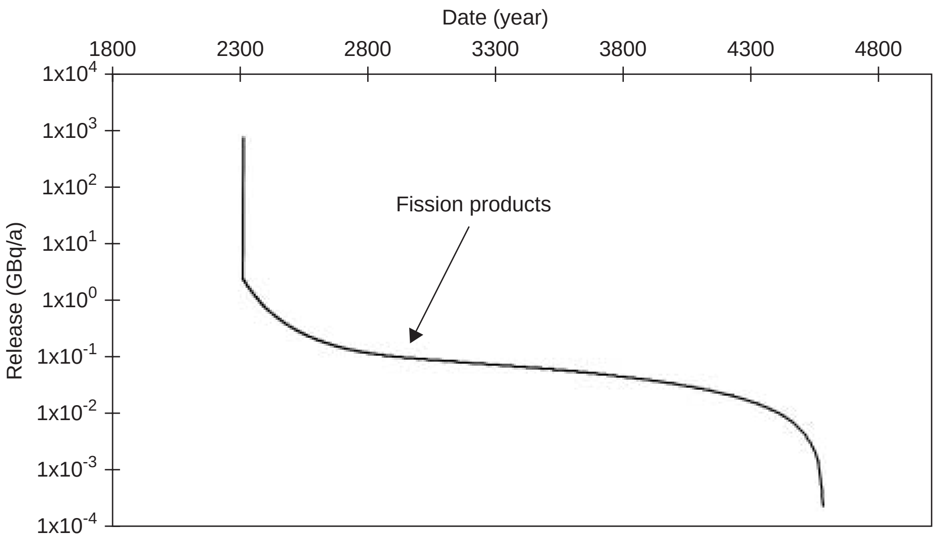


FIG. 26. Tsivolka Fjord, icebreaker reactor compartment and fuel container, fission product release for Scenario A.

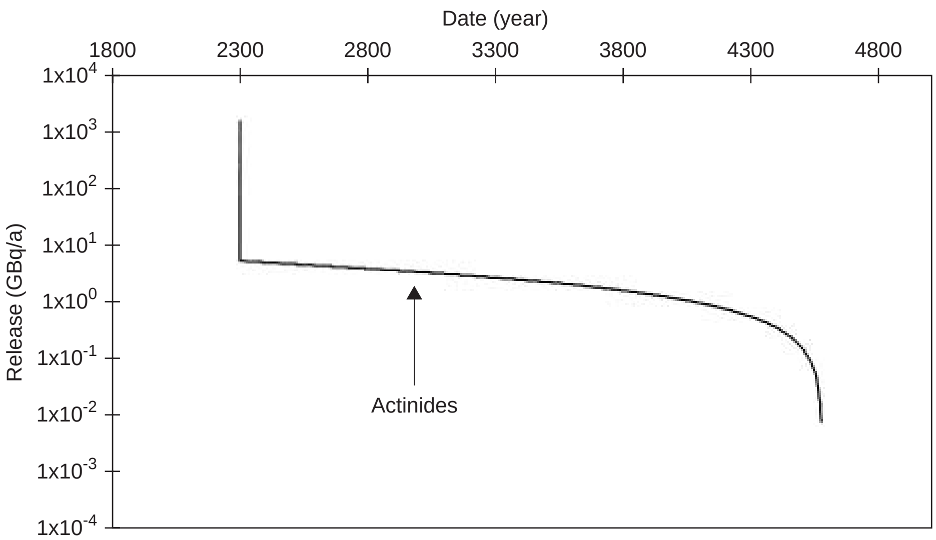


FIG. 27. Tsivolka Fjord, icebreaker reactor compartment and fuel container, actinide release for Scenario A.

exist in the bolted flanges. The release rate starts at 2.3 GBq/a, but five years later the hull fails completely and the release rate jumps to 23 GBq/a. Caps on main coolant pipes, control rod tubes and other penetrations fail in the year 2055, increasing the release rate to 41 GBq/a. Until the year 2300, the release rate falls, largely because of the decay of the ^{60}Co component of the total activation activity.

In the year 2305, the container B on the pontoon is breached and the thermal shields of reactor N2 are exposed to corrosion at the full best estimate corrosion rate. This shows up as an increase in the overall rate to 22 GBq/a. These shields are corroded away by the year 2665, and the rate drops to 3.6 GBq/a as the only remaining activated material resides in the reactor pressure vessel assemblies.

The reactor pressure vessels within the reactor compartment disappear by the year 2700 and their cladding by 2795. This exposes the remaining material in the thermal shields to corrosion at the full best estimate corrosion rate and the release rate jumps to 6.0 GBq/a. The rate falls away gradually for the final 250 years until all the activated material has gone by the year 3050. As shown above, there is still an appreciable amount of fuel material left to corrode away after the activated steels have disappeared.

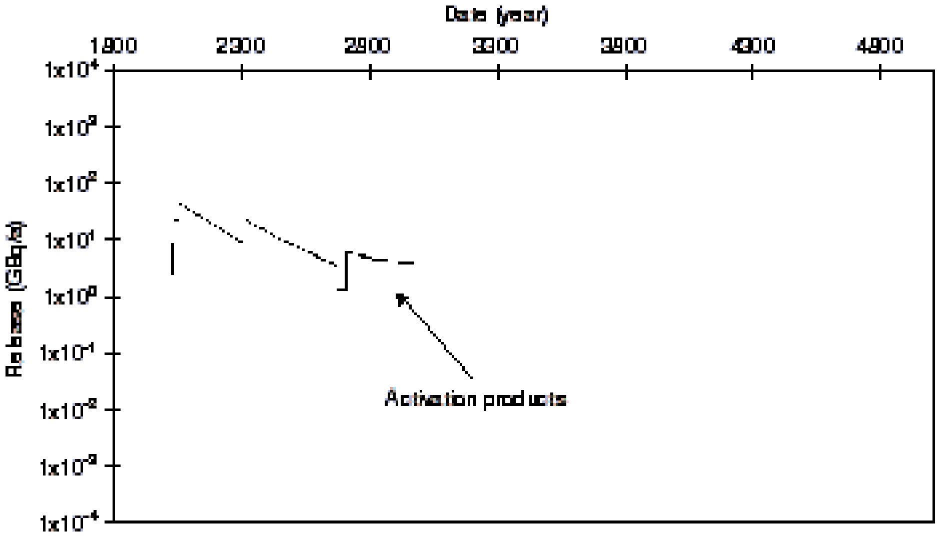


FIG. 28. Tsivolka Fjord, icebreaker reactor compartment and fuel container, activation product release for Scenario A.

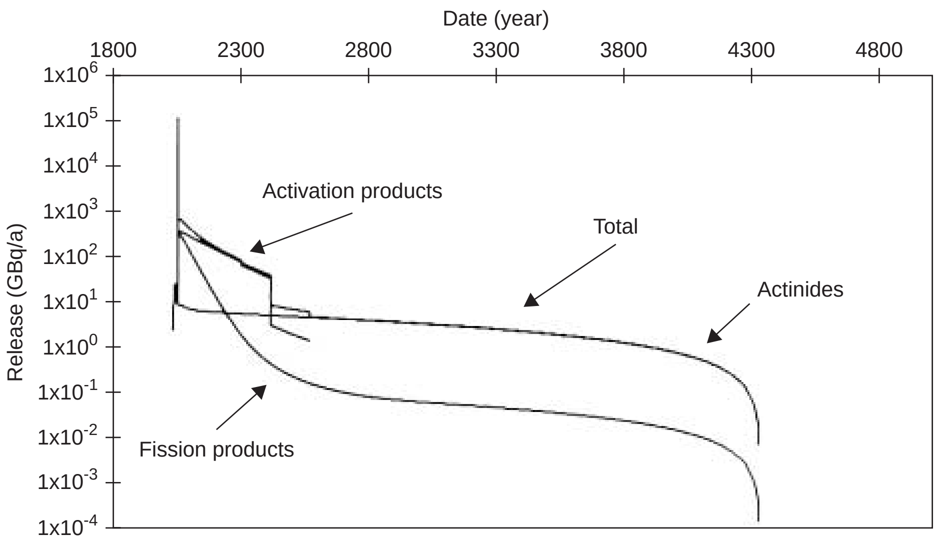


FIG. 29. Tsivolka Fjord, icebreaker reactor compartment and fuel container, total radionuclide release for Scenario B.

Total release rates

Figure 25 shows the total release in Scenario A, the summation of Figs 26 to 28. The initial release is dominated by the activation product material from the reactor compartment, with the peak between the years 2035 and 2055 of the order of 22 GBq/a. Release rates drop until the year 2305, when containers A and B holding the damaged nuclear fuel open up. A fission product and actinide spike of 2100 GBq/a dominates the release in that year.

In the year 3050, when the last thermal shields disappear, the decrease in the total release rate is marked, as the rate drops to 3.4 GBq/a. This leaves only the fuel to continue on to the year 4570.

Figure 29 shows Scenario B. A collision or munitions explosion is postulated in the year 2050 near the dump site in Tsivolka Fjord. This accident is assumed to breach Containers B and C on the pontoon, and also to break off the reactor pressure vessel lids in the reactor compartment. The contents of all the units are then exposed to best estimate corrosion rates, with no containment barriers.

The initial corrosion rate is observed until the year 2050; then the accident causes a release peak of 110×10^3 GBq/a. Because of the increased corrosion

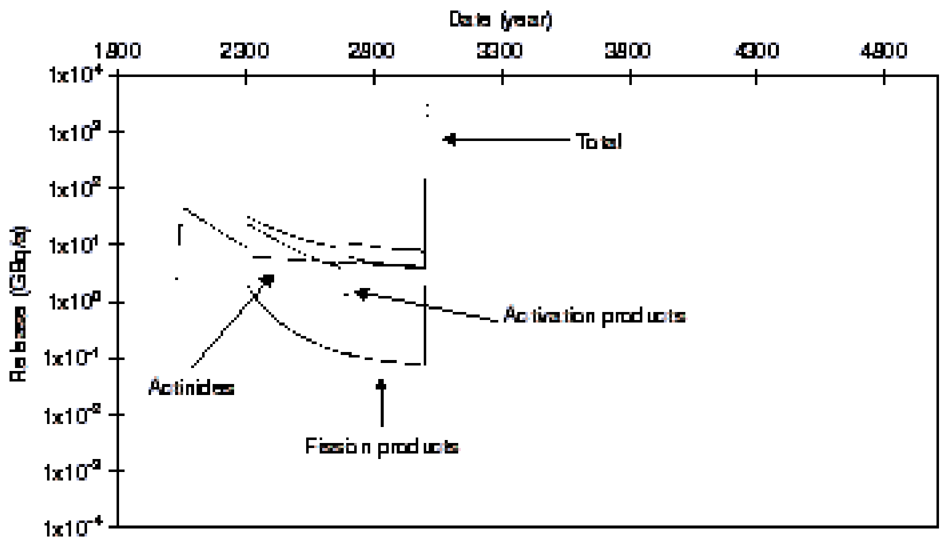


FIG. 30. Tsivolka Fjord, icebreaker reactor compartment and fuel container, total radionuclide release for Scenario C.

rates, the activated components are now totally corroded away by the year 2565 (as opposed to 3050 in Scenario A), and the fuel disappears by 4320 (4570 in Scenario A).

Figure 30 shows Scenario C, the disruption due to glacier scouring in the year 3000. The situation up to that time is assumed to be that of Scenario A, then the remaining active material (3000 GBq) is released to the environment in that year.

4.2.4.4. Total release rates for the Kara Sea

Figures 31 and 32 show the cumulative total radionuclide release rates from all the dumped PWR steam generating installations. The LMR of unit 601 is excluded from this diagram because of the much extended time frame for its corrosion. Figure 31 shows Scenario A, the peaks of 3000 GBq/a due to the release of fuel between the years 2030 and 2050, the peak of 2100 GBq/a, due to the opening of containers protecting the icebreaker fuel in the year 2305 and the sharp falls when the thermal shields corrode away and cease to contribute to the total release rate.

Figure 32 shows the same Kara Sea total, but for Scenario C, the glacier scenario. Here, the release in the year 3000 totals 4500 GBq/a.

4.2.5. Potential criticality of reactors

Three scenarios were reviewed to investigate the possibility of any of the reactor cores achieving a critical state:

(1) Corrosion of a large proportion of the control rod material before the nuclear fuel has substantially gone. This could have the same effect on core reactivity as control rod withdrawal during a reactor startup, and cause criticality.

(2) The corrosion process or certain forms of attempted remedial action cause a structural change within the core, such as the nuclear fuel falling to the bottom of the reactor pressure vessel or control rods being displaced, resulting in some or all of the fuel and core material reaching a critical mass.

(3) Ingress of water into the core will increase neutron moderation and hence increase reactivity. In particular, in the case of the LMRs, corrosion of the Pb–Bi coolant allows water to enter the core and surround the fuel pins, which will provide a much greater amount of moderating material than was present during core operation, and hence increase the reactivity (a measure of how close an assembly of nuclear fuel is to criticality) of the core.

Although the values of the parameters used were at the limits of the IASAP ranges and gross assumptions were made on the critical mass, the thickness of the stainless steel structure and the solubility of uranium, the review showed the following:

(1) The possibility of sufficient nuclear fuel being available to form a critical assembly after substantial corrosion of the control rods or loss of the stainless steel supporting structure cannot be ruled out. The risk of this occurring increases with greater fuel enrichment and lower core burnup: hence the reactor most at risk is the right board reactor of submarine factory No. 421, and to a lesser extent those of submarine factory No. 901. Using the most extreme

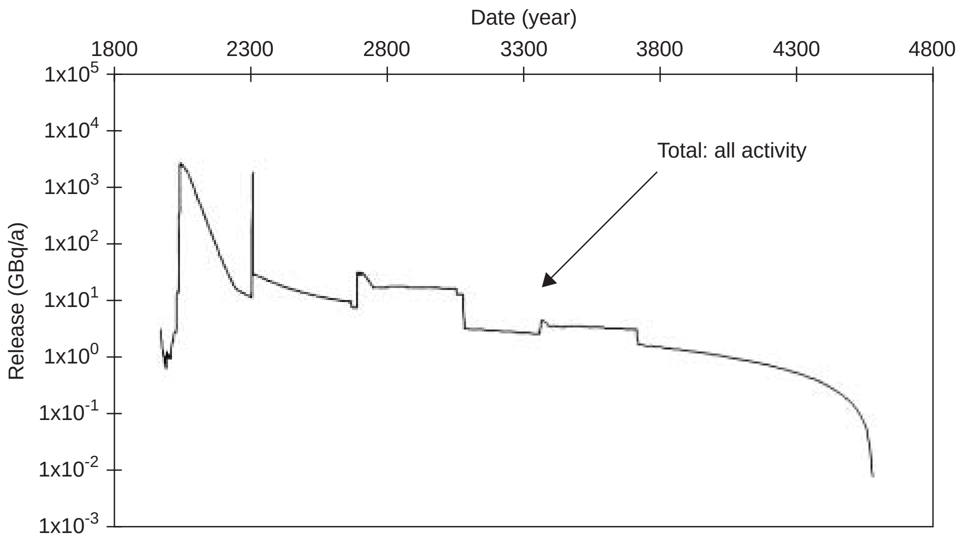


FIG. 31. Kara Sea, all units except 601, total radionuclide release for Scenario A.

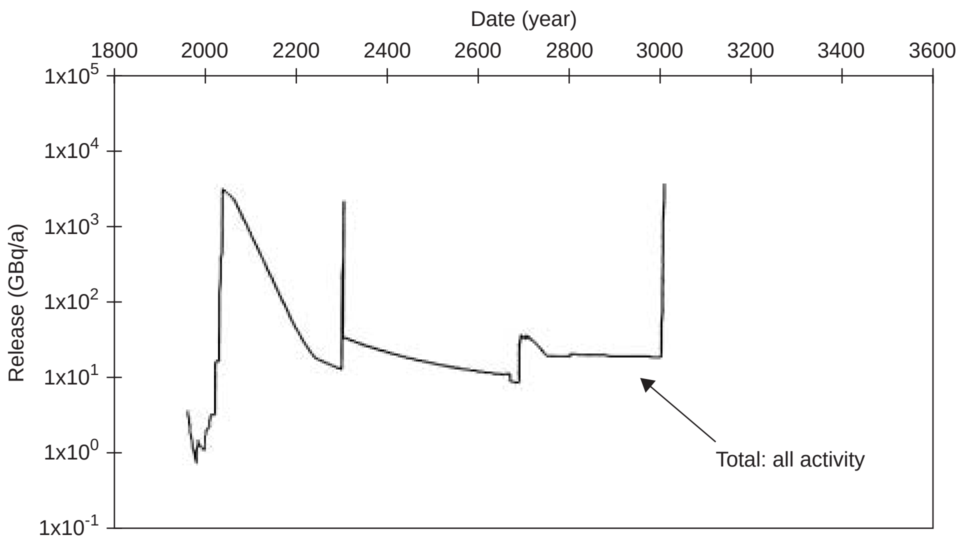


FIG. 32. Kara Sea, all units except 601, total radionuclide release for scenario C.

IASAP corrosion values, such an event could occur by about 2730.

(2) The behaviour of the Pb–Bi thermal expansivity has a large impact on the potential for criticality, and the time at which it occurs.

(3) Each of the three scenarios set out above is extremely unlikely to cause a critical assembly to be formed. The combination of all three, however, makes the possibility less remote, particularly for the reactors of submarine factory No. 421, 901 and 601. Analysis of such a combination indicates that around the year 5000 the reactor cores in submarine factory No. 601 could slowly achieve criticality.

If criticality is possible and occurs through slow corrosion of the control rods or water ingress, or a combination of the two, the approach to criticality will be extremely slow so that the possibility of prompt criticality and any kind of explosion or structural damage can be entirely ruled out. Instead, the onset of criticality will cause a slow rise in fission rate and an increase in nuclear fuel temperature. The conditions in the core are likely to be such that the heat generated is easily dissipated, particularly as the rate of reactivity addition is so slow, and the temperature is unlikely to rise significantly. This may cause a slight rise in corrosion rates and increased flow through the reactor pressure vessel. Since the cores will be water moderated, the rise in temperature will probably cause a reduction in reactivity, and there is a possibility that self-regulation could occur. As corrosion continues, the critical state is likely to be short lived compared with the total lifetime of corrosion, with further structural corrosion and loss of fuel leading to a reduction in reactivity and eventual subcriticality. Such behaviour is unlikely to have a significant effect on total release rates.

A structural change resulting in prompt criticality is an extremely remote possibility, but cannot be ruled out. If it occurred owing to corrosion of the structure resulting in toppling and control rod displacement, the following large release of energy would cause the structure to further disintegrate and radionuclide release would be accelerated, possibly causing the total remaining inventory to be released to the environment. If it occurred because of attempted remedial action which involved movement of the reactor pressure vessel, the consequences could be much more serious.

A prompt criticality with core disassembly in the far distant future would involve very little radioactivity compared to the present radionuclide inventory in these cores. By the year 2700, nearly all current fission product inventory in the cores would have decayed. Also, the amount of fission products produced in a prompt critical excursion is relatively small. For example, the amount of ^{137}Cs generated in a 10^{18} fission criticality excursion (about the same as the SL-1 accident in the United States of America) would be 44 MBq [87].

4.3. RESULTS AND ANALYSES

4.3.1. Reliability

The principal sources of error in the predicted release rates are:

(1) the information on steam generating installation structure and materials;
(2) the inventory;
(3) the values of best corrosion rates;
(4) the degree of pessimism used in the models.

4.3.1.1. Information on the steam generating installation structures and materials

The reliability of the information on the submarine and icebreaker steam generating installation structures and materials centres on the reactor cores, the thermal shields, the reactor pressure vessels, and their associated support structures. Core details concerning the submarine LMRs are the most comprehensive, with information on the fuel rod materials, dimensions, and pitch, core control rod and emergency protection rod materials and locations, and overall materials distribution. For the icebreaker PWRs, the information is essentially limited to the fuel rod materials, dimensions and pitch, and the configuration of the core control rods and emergency protection rods within the cores. In both cases, the core details are substantially more reliable than the core details for the submarine PWRs, where essentially everything is assumed.

Information on the thermal shields and the reactor pressure vessels is reasonably complete concerning the materials, dimensions and locations of one with respect to another. The most detailed description covers the submarine LMRs. No information on the support structures for the reactor cores, thermal shields and reactor pressure vessels has been provided to any significant degree. As such, this is the area where the largest number of assumptions have been made and where the data are least reliable. Given the concerns about possible reactor criticality in the event of remedial action, the lack of details on the support structures may influence future decisions.

4.3.1.2. Radionuclide inventory

The reliability of the radionuclide inventory centres on the details of the reactor cores, their associated reactor operating histories and the core models used in the calculations. The reliability of the core details has been described in Section 4.1.1. Suffice it to say that the details of the submarine LMR and of the icebreaker core are the most reliable. Reactor operating histories obtained from the Kurchatov Institute [74, 75] and the Institute of Physics and Power Engineering [76] for the submarines and icebreaker, when compared with those from Russian Navy records [79], were found to differ by no more than a factor of two and on average yielded a comparative ratio of 1.0 $\pm$ 40%. As such, the reactor operating histories used in the calculations were considered reliable.

Core models used in the calculations have been discussed in Section 4.2.2. Models used for the submarine LMRs and icebreaker PWRs represent the actual configurations; however, the models for the submarine PWRs are built on the same assumptions as the icebreaker PWRs. When compared with an independent estimate of the radionuclide inventories prepared by the State Institute of Applied Ecology, Russian Federation [79], the inventories used by IASAP were determined to represent the best estimate. With respect to potential radionuclide release, while the submarine LMR and icebreaker radionuclide inventories are deemed the most reliable, the submarine PWR radionuclide inventories are considered the least reliable.

4.3.1.3. Values of best estimate corrosion rates

The reliability of the values of the best estimate corrosion rates used depends on the type of material considered. Degradation rates are best estimates based on currently available data, some of which are scant for materials such as Pb–Bi eutectic and uranium fuels in their various alloy and oxide ceramic forms. The different types of materials may be subdivided as follows:

(1) *Structural materials.* Extensive literature search has been undertaken for common structural materials such as stainless steels and mild steels, and the best estimate corrosion rates employed are believed to be on the conservative side of the range of values examined [81, 82]. Soo [83] reviewed values determined by the IASAP and considered them reasonable for the bulk and pitting corrosion rates of stainless steel. Sufficient data exist on mild steels (low alloy and carbon) to justify their use and reliability.
(2) *Pb–Bi coolant (LMR only).* In the absence of data on Pb–Bi, the best available alternative was the known corrosion performance of lead in sea water. However, there is a possibility that the behaviour of the Pb–Bi eutectic may be different, and some doubt must therefore exist on the reliability of these data. Given the significance of this material with regard to its likely inhibition of fuel corrosion in the core, and indeed to its thermal expansivity and the possible presence of voids in the core, further information on this material is essential.
(3) *Fissile fuel.* Research data on the corrosion of uranium fissile fuels in sea water is very limited. The best available data has been used for the dissolution rate of the icebreaker oxide fuel.
(4) *Filler materials.* Furfurol(F), bitumen and concrete. Alexandrov et al. [85] confirmed the lifetime of Furfurol(F) as 100 years, in a project conducted during the period of the IASAP study. Carter [84] helped in assigning values for concrete and bitumen at 100 years as well, but without better information on their make-up, these remain best estimates.

4.3.1.4. Degree of pessimism used in the models

In order to produce pessimistic but realistic values of radionuclide release, several assumptions were made:

(1) All material corroded is immediately available to the environment and is regarded as released for the purposes of the IASAP models. Making this assumption negates the need for analyses of solubility and environmental transport mechanisms through the containment and immediately demonstrates that the IASAP release rates must be pessimistic. In practice, of course, much of the corroded material will be both heavy and insoluble and will remain within the containment, reducing the true release rates. This fact has been discussed in the context of reactor criticality, where it was assumed that about 80% of the corroded fuel was insoluble and would fall to the bottom of the reactor pressure vessel. Retention of corroded material could further reduce the release rate as its presence may slow down or even stop the flow of water through the reactor, retarding or preventing further corrosion and release of material.
(2) Best estimate corrosion rates used for release rate calculations were chosen to give fast release rates, adding further pessimism to the models. Also, the best corrosion rates applied to the steam generating installation materials were constant, whereas some studies indicate that true corrosion rates will decrease with time as a corrosion resistant layer forms on the surface of material under attack.
(3) Fuel pin cladding has been ignored in the IASAP release rate models. Inclusion of the cladding, even when the fuel has been substantially damaged, will retard or reduce the release rates.
(4) The filler materials were assigned a lifetime and rate of degradation at the end of which they ceased to be barriers to corrosion or transport of corroded material. There is little information available on the long term behaviour of the filler materials in sea water and in a radiation environment; if filler lifetimes are longer, the release rates in the first few hundred years will be less. In reality, the filler material will not simply disappear at the end of its lifetime and may retard or prevent release of material.

4.3.2. Sensitivity

By its very nature, the IASAP release rate model is sensitive to the chosen corrosion or degradation rates for the steel components and for the barrier materials. Although an extensive literature study was undertaken and advice sought from materials specialists from the contributing countries in order to provide the best estimates, steel corrosion rates in the ocean can result in a

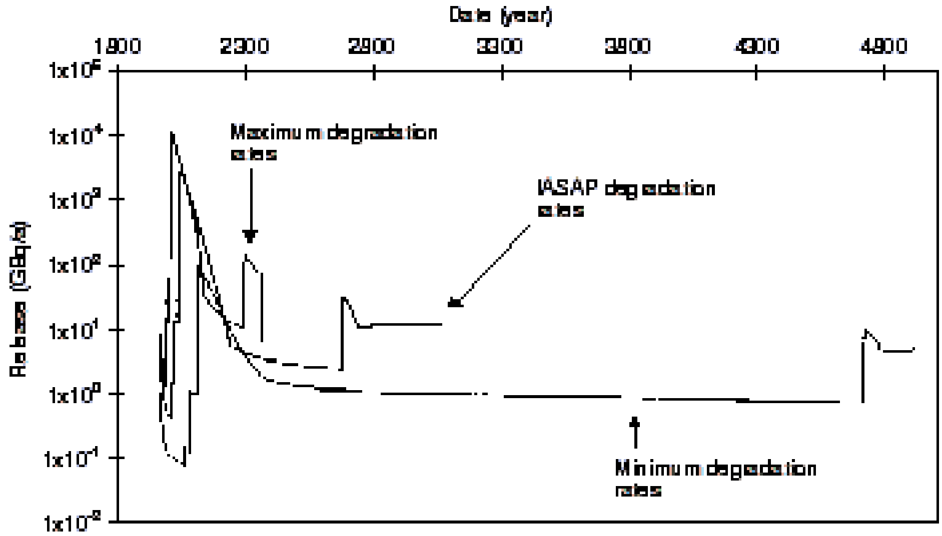


FIG. 33. Scenario A, Abrosimov Fjord: Sensitivity of radionuclide release to changes in corrosion rates and barrier lifetimes. Release rates with maximum, minimum and 'best corrosion rates' applied.

wide spread of values, and barrier effectiveness is not well defined.

The model can be run using a range of the rates. One or more of the parameters can be varied, for example, by using a maximum stainless steel pitting corrosion value and a 500 year Furfurol(F) lifetime, as opposed to the 100 year lifetime selected for the purpose of this study. To illustrate this, the model was run with three sets of corrosion rates and barrier lifetimes. The rates or lifetimes were all chosen at the upper and lower ends of the published or advised values (see Table XII) and compared with the best estimate corrosion rate values.

Figure 33 shows the result of one such run, using the Abrosimov Fjord units and Scenario A. Applying the maximum rates, all the units have disappeared by the year 2370; using the best estimate corrosion rate values, the time is extended to 3075 and using the slowest rates, the units survive until about 5000. As a general principle, there is a factor of three between release rates peak values for the highest degradation rates and the best estimate corrosion rate values and a factor of 30 between the best estimate corrosion rate and the slowest values. The maximum rate peak is 12 000 GBq/a, the IASAP peak is 2700 GBq/a and the slow corrosion peak is 90 GBq/a.

4.4. ISSUES RELATING TO POSSIBLE REMEDIAL MEASURES

Before any remedial measures are taken, it is assumed that a site survey will be undertaken to establish the structural condition of units containing spent nuclear fuel and evidence of radionuclide leakage. In view of their relatively low levels of activity, this section does not address the issue of remedial measures for radioactive material not containing nuclear fuel. An estimate is provided below of the physical condition of units containing spent nuclear fuel to assist in any assessment of necessary or feasible actions such as: reinforcement of existing containment barriers around nuclear fuel, or recovery of dumped nuclear fuel for land storage.

4.4.1. Reinforcement of existing barriers

To inhibit any monitored leakage from units containing spent nuclear fuel or to enhance current containment arrangements, it may be possible to pump fillers inside existing barriers. Additional barriers may also be erected around the units, such as capping materials applied to potential leakage paths. Factors to be considered include:

(1) The risk of disturbance to units containing spent nuclear fuel that may damage existing barriers, perhaps inducing leakage and weakening of containment structures.
(2) The risk of disturbance causing a change in orientation of nuclear fuel in reactor pressure vessels sufficient to displace core control rods and initiate criticality, to the extent that prompt criticality may occur. The resulting power excursion due to prompt criticality could cause structural damage and immediate radionuclide release to the environment and prejudice the safety of personnel at the scene. Even

if slow criticality occurs, the increase in temperature may accelerate corrosion rates and cause early breakdown of containment barriers. If this occurs with fuel in sealed containers, they may be ruptured by an accompanying rise in pressure, leading to premature breakdown of barriers to radionuclide release.

(3) The risk of disturbance causing displacement of nuclear fuel in reactor pressure vessels due to corrosion weakened or damaged condition of fuel supporting structures. In this case the possibility of a critical mass being formed at the bottom of a pressure vessel, beneath the core control rods, cannot be ruled out. This could result in either slow or prompt criticality similar to the situation caused by displacement of core control rods discussed under item (2) above.

4.4.2. Recovery of spent nuclear fuel for land storage

In the event that a decision is made to recover dumped spent nuclear fuel for removal to a land disposal site, the factors to be considered are of a rather wider range:

(1) The risk that through deterioration in the condition of units containing spent nuclear fuel the barriers fail during recovery operations, releasing radionuclides either at the scene or during subsequent recovery and transport to the land disposal site.

(2) The risk that disturbance to fuel supporting structures or change in orientation of nuclear fuel induces criticality, as described in Section 4.4.1 (2) and (3) above.

(3) A nuclear safety hazard associated with the handling and transport of nuclear fuel to its final disposal site, including the radiation dose to personnel involved throughout the process.

(4) A comparison of the potential hazard to the environment of nuclear fuel in its land disposal site against retention at its sea dump site.

Items (3) and (4) are discussed in more detail in Sections 6 and 7.

4.4.3. Structural integrity of spent nuclear fuel containers

Table XIII provides an estimate of the structural integrity of spent nuclear fuel containers in 1996 and in 2015, specifying the implications for recovery purposes. This represents a theoretical assessment only and should therefore not suggest any remedial action without prior confirmation, by way of site survey, of the actual condition of the units containing nuclear fuel.

4.5. CONCLUSIONS

(1) The radioanuclide inventory of the marine reactors and associated spent nuclear fuel dumped in the Kara Sea was calculated on the basis of the information on nuclear fuel quantity, enrichment and burnup. Inventory data were split into three categories: fission products, activation products and actinides.

The total activity in 1994 was calculated to be 4.7×10^{15} Bq. Fission products made up 86% of this total, activation products 12%, and the actinides 2%. The major component of this total inventory comes from the container holding the spent nuclear fuel from the icebreaker reactor.

The inventory figure finally established in this study for all these units was considerably lower than the one originally published in the White Book. The White Book quoted 89×10^{15} Bq as the total activity in the dumped reactors and spent nuclear fuel from the icebreaker. The Working Group established that the original inventory at time of dumping was 37×10^{15} Bq.

(2) A computer model was developed on the basis of best estimates of the corrosion rates of the steam generating installation materials, barrier lifetimes, information on the structures and containment of the spent nuclear fuel and methods used for dumping. Calculations were made for the release of radionuclides into the Kara Sea, using several scenarios.

Release rates for all the radionuclides were calculated from the objects in each fjord. For example, using the best estimate scenario and summing the contributions from all the fjords with pressurized water reactors, it was shown that a release rate peak of 3000 GBq/a would occur around the year 2040 when the reactor containments are partially breached, and there would be another peak of 2100 GBq in the year 2305, when the icebreaker fuel container corrodes open. For a large part of the time, however, release rates would lie between 20 GBq/a and 2 GBq/a. Some very low levels of activation product releases might be expected a decade or so into the next century owing to the corrosion of the outer walls of the reactor pressure vessels.

The liquid metal cooled reactors, with their very slow corrosion rates and large mass of solidified Pb–Bi heat transfer, were treated differently and separately. Using this second model, it was shown that the release rate would peak at 8 GBq/a by the year 2105 but, for

TABLE XIII. ESTIMATES OF THE CONDITION OF DISPOSED SPENT NUCLEAR FUEL CONTAINERS FOR REMEDIAL ACTION CONSIDERATIONS

Factory No. and reactors	Disposal date	Disposal site	Containment structure	Condition		Conclusions
				1996	2015	
901 2 PWR	1965	Abrosimov Fjord	a. Reactor compartment (RC) — hull b. Mild steel bulkheads c. Reactor pressure vessel (RPV)	a. 25% corroded – sound b. Unsound c. Little corroded – sound	a. 40% – weakened b. Unsound c. 5% corroded – sound	RC bulkheads believed to be ineffective as containment barrier in 1996. It may not be possible to use hull for recovery purposes. RPV remains intact.
285 1 PWR[a]	1965	Abrosimov Fjord	a. RC – hull b. Mild steel bulkheads c. RPV	a. 25% corroded – sound b. Unsound c. Little corroded – sound	a. 40% – weakened b. Unsound c. 5% corroded – sound	RC bulkheads believed to be ineffective as containment barrier in 1996. It may not be possible to use hull for recovery purposes. RPV remains intact.
OK-150	1967	Tsivolka Fjord	a. Pontoon b. Outer container — steel box with welded lid c. Inner container — steel clad concrete box with welded lid	a. Unsound[b] b. 30% corroded – sound c. 20% corroded – sound	a. Unsound[b] b. 45% corroded – sound c. 35% corroded – weakened	Pontoon may not be used for recovery purposes. Containers remain intact.
421 1 PWR	1972	Novaya Zemlya Trough	a. Barge b. Concrete enclosure bulkheads c. RPV	a. Unsound[b] b. 25% degraded c. Little corroded – sound	a. Unsound[b] b. 45% degraded c. 5% corroded – sound	Barge may not be used for recovery purposes. Effectiveness of concrete barrier likely to be severely degraded. RPV remains intact.
601 2 LMR	1981	Stepovoy Fjord	a. Submarine hull b. Bitumen inside reactor compartment c. RPVs and steam generators	a. 10% corroded – sound b. 15% degraded – sound c. Sound	a. 25% corroded – sound b. 35% degraded – cracked c. Sound	RC and RPV remain intact. Effectiveness of bitumen barrier considered severely degraded. Otherwise no structural constraints on remedial actions.

[a] A defuelled PWR is also included in this reactor compartment.

[b] Judged unsound on the basis that the pontoon and barge are known not to have been built for this specific purpose and are believed to have been old and at the end of their serviceable life.

Assumptions: a. The concrete and bitumen have effective lifetimes of 100 years, with a linear decrease in effectiveness over this period.
b. Best corrosion rates, as used in the models of this study.
c. A survey of the dump sites will be necessary to assess the actual condition of the discarded objects before any decisions on remedial actions.

most of the remaining long lifetime, release rates would remain close to 7×10^{-3} GBq/a.

For this best estimate scenario, the calculations were extended until all components had corroded away; for the pressurized water reactor units, this is expected to happen by the year 4570; the submarine liquid metal cooled reactors were shown to take until the year 36 600 to disintegrate.

With another postulated scenario — glacial action in the year 3000 — it was calculated that 4500 GBq would be released to the Kara Sea simultaneously from all the pressurized water reactors and their spent nuclear fuel, and 2100 GBq from the liquid metal reactors.

These release rate data, for all isotopes, was then used for dispersion and exposure pathway modelling.

(3) The errors, uncertainties and conservatism in the model were discussed, and note was taken, inter alia, of the assumption that all activity is released and dispersed immediately in the sea. This makes the estimated release rates overly pessimistic, as much of the corroded material will slump to the bottom of containment structures or will be buried in surrounding sediments and therefore not pass into the fjords for circulation into the Kara Sea and beyond.

Corrosion rates were best estimates found in the literature but some uncertainties remain, especially the rate assigned to the solidified Pb–Bi of the liquid metal reactors.

Any future studies should obtain and use actual corrosion rates and on-site observations of barrier material effectiveness from samples of the actual objects themselves, providing such investigation does not breach the containment barriers.

(4) The risks were considered of a criticality incident in the dumped fuelled reactors following corrosion of the components or remedial action. The possibility of such an incident was found to be very low but could not be ruled out. This should be borne in mind if remedial action is contemplated.

(5) With regard to the availability of design information for the submarine and icebreaker steam generating installations, a great deal of information is still lacking about the support structures for the reactor cores, thermal shields and reactor pressure vessels. While sufficient information is available on the disposal of the spent nuclear fuel from the icebreaker to lessen concern on that score, the absence of design information is a hindrance to criticality studies and the evaluation of remedial action for the submarine reactor pressure vessels that contain spent nuclear fuel.

Sufficient information has been provided for the reactor cores of the liquid metal cooled submarine and the icebreaker pressurized water reactor cores to develop preliminary core models; however, information on cores of the submarine pressurized water reactors has been so limited as to require several assumptions with respect to their composition and layout. Less is known about the U–Al alloy fuels of the pressurized water reactors than about the icebreaker fuel and its configuration in the reactor structure. Thus, inventories and release rates associated with the submarines with pressurized water reactors, and specifically the reactors containing spent nuclear fuel, have the greatest relative uncertainty and would be likely to require further and more detailed evaluation, should remedial actions be considered.

(6) As a further verification of the validity of the corrosion and release scenarios assumed in this analysis, it would be useful to undertake on-site investigation of the integrity of the objects, looking for any leaks which have started earlier than anticipated by the model. First, the condition of the welds and caps which seal important leakage paths should be investigated. These would include the icebreaker fuel container lid weld, the control rod cap welds on the top of the pressurized water reactors, and the state of the concrete capping over unit 421. Hatches into reactor compartments might provide access to small remotely operated vehicles to look for radionuclides leaking from the nuclear fuel and for activation products corroding from the internal reactor steels.

(7) Assuming the barrier strategy is correctly modelled and there is no disturbance to the objects, release rates will be very low for most of the time these structures remain corroding away on the Kara Sea floor.

5. ENVIRONMENTAL MODELLING FOR RADIOLOGICAL IMPACT ASSESSMENT

5.1. INTRODUCTION

The IASAP team of scientists developed models for the assessment of the radiological consequences of the releases of radionuclides from the dumped wastes in the Arctic. In parallel, they developed a number of complementary models which reflect different modelling approaches and accommodate the requirements within IASAP of providing predictions at very diverse space- and time-scales; the stated IASAP objectives require assessment of impact at near field (local), intermediate field (regional) on relatively short time-scales, and far field (global) over much longer time-scales of thousands of years.

Within such a collaborative programme, where the main modelling need is for predictive capabilities, it is common practice to include a model intercomparison and an evaluation of the different types of model on both theoretical and practical grounds. This would also include an evaluation of the data and modelling uncertainties and their subsequent effects on the predictions. The complementary nature of the models provides a mechanism of incorporating model uncertainties within the overall uncertainty analysis of the predictions.

As a result of these considerations, the team devised and implemented an extensive model comparison exercise. Results from this intercomparison will be briefly presented and discussed in this section but only insofar as they affect the dose estimates.

5.1.1. Aims and objectives of modelling

Representatives of both the IAEA and seven Member States were involved in the modelling. Within the general programme of assessment of the present and future radiological impact of the dumped waste in the Kara Sea, two specific objectives were identified:

- Development of predictive models for the dispersal of radioactive contaminants both within and from the Arctic Ocean and an assessment of their reliability; and
- Evaluation of the contributions of dominating transfer mechanisms to the contaminant dispersal and, ultimately, the risks to human health and the environment.

In the first phase of the work, predictive models were prepared either by extending existing models or by developing new models. It was also established that different models would be particularly suited to assessment at different scales (specifically at local and regional spatial and short time-scales, and global spatial and long time-scales). We shall turn again to this important point in the discussion of the models and the comparison of their predictions.

To facilitate this work, and as part of the model intercomparison exercise which would follow, the IASAP group collated and processed available information on the Arctic region.

In the final stage, it was envisaged that the results from the model intercomparison exercise would be used as a basis on which to evaluate the estimates of concentration fields when detailed source term scenarios were used. Additionally, the results were also used to assess the uncertainties in ensuing dose calculations.

5.1.2. Brief oceanographic basis of the modelling work

The group devoted substantial effort towards defining a synthesis of existing data sources for the Arctic. It identified specific processes as being peculiar to the area and thus potentially important for model development. A full description of these processes with references can be found in Section 3; the main features discussed include:

(a) a general oceanographic and geophysical description of the areas of interest, specifically the bays on the eastern coast of Novaya Zemlya, the Kara and Barents Seas and the Arctic Ocean;
(b) the sedimentology of the area; and
(c) a description of the Arctic marine ecosystem.

It should be emphasized that the creation of a quantitative and qualitative description of the oceanographic features of the area was a significant undertaking. Sources for much of the required information were limited and difficult to locate. However, the need for a detailed description was clear: in many previous assessment programmes, the Arctic had only been considered in a marginal way, and thus it was necessary to extend and modify the models used until then. In addition, there was also a need to incorporate region specific features by using modelling tools which had previously been developed for climate studies (modelling the growth and transport of ice) and to apply hydrodynamic models which already existed as part of climate modelling programmes. Only the key features of relevance to the modelling and model development are selected and described below. In summary, the main features of concern are ice formation and transport, major fresh water runoff, seasonal effects, upwelling and dense water formation, fish migration, and, last but not least, the lack of detailed data for the area. Not all these processes could easily be incorporated within the existing modelling frameworks but, wherever possible, some evaluation of their influence was carried out. The lack of data also acted as a significant hindrance in the development of site specific models, e.g. it proved impossible to find sufficient data to develop models for specific fjords such as Tsivolka Fjord. As such, difficulties in data sources are a considerable source of uncertainty because it was not possible to validate all aspects of the models fully and completely.

5.1.2.1. Significant features of the Kara Sea

The particular features of interest for the Kara Sea are:

Runoff

Approximately 1300 km^3 of fresh river water enters the area annually (mainly from the Ob and Yenisey rivers). There is a distinct maximum flow in the summer (80% of the annual runoff occurs between June and September) with much reduced flow in the winter.

For compartmental models, fresh water inflow is incorporated but only to provide flow balance. For hydrodynamical models, both the volume transport and the input of low density fresh water have a significant impact on the flow field.

Bottom water formation

Bottom water formation is thought to occur in the northern Kara Sea and along the east coast of Novaya Zemlya during ice formation mainly in leads or polynias (open areas in the ice) due to haline convection. The so-called shelf brine water is presumed to fill the deeper troughs and trenches and to flow out to the Arctic Ocean. Information on these processes in the Kara Sea is very limited. Vertical mixing and thermal convection occur during the entire winter and are mainly responsible for the homogenization of the water column. The hydrodynamic models usually parametrize these processes by turbulent mixing, but this does not produce vertical flows or bottom water.

Ice cover and transport

Sea ice begins to form in September and melts in June. Icebergs are also found in the area. The ice drift is generally northward to the Arctic basin. There is extensive ice exchange between the Kara and Barents Seas. The whole Siberian shelf is an important area for ice formation, with the ice being transported westward by the transpolar drift. This is of particular relevance to models that require as inputs surface fluxes of heat and salt and transfer of momentum from the atmosphere to the water column (hydrodynamic models) due to the influence of ice cover on such phenomena. In this way, dispersion of radionuclides is also affected, although direct dispersion by sea ice is not modelled. It should be noted that the transport of radionuclides in sea ice with incorporated sediment and the importance of this mechanism are assessed in a later section.

5.1.2.2. Features of the Barents Sea

The Barents Sea exchanges water with the Arctic and Atlantic Oceans as well as with the Kara and White Seas. It also receives river runoff. It is never completely covered with ice, but can be characterized by intensive ice formation and water mass transformation. Detailed models and databases for this sea already exist, and we therefore do not discuss this area in any detail.

5.1.2.3. The Arctic Ocean

The Arctic area as a whole presents a modelling challenge given the extreme conditions, including the features already highlighted in connection with the Kara and Barents Seas. It is almost entirely land-locked and connects with the rest of the world's oceans through the Fram and Bering Straits; the central Arctic is permanently covered by sea ice. The circulation within the central Arctic is increasingly well understood; most of the Atlantic water enters the Arctic through Fram Strait, while a current branch also brings Atlantic water into the basin over the Barents and northern Kara Sea shelves.

Several detailed hydrodynamic models for the Arctic exist for climate work. One such model was utilized in the model intercomparison work and was also used to provide estimates of ice fluxes for later calculations.

5.1.2.4. The fjords of Novaya Zemlya

Only limited site specific information is available for the fjords of Novaya Zemlya where material had been dumped. The most valuable information came from the Russian–Norwegian joint cruises, from which crude topography could be inferred and a limited number of temperature–salinity profiles could be used to provide very broad oceanographic descriptions. In several bays, the team of scientists were unable to access sufficiently detailed information to allow the development of site specific models.

5.1.3. Productivity of the Barents and Kara Seas

The Kara Sea is a typical Arctic shelf sea. Its climate is essentially polar and is harsher than that of the Barents Sea. Tentatively, the Kara Sea is considered three to five times less productive than the Barents Sea, with the most productive zones in its south-west sector and in the Ob and Yenisey estuaries. In the open Kara Sea there is no commercial fishing. Sea mammals are of some commercial significance for the local populations. The marine ecosystem of the Barents Sea has been well documented and studied. It was covered in more detail in Section 3 above. Many of the species of commercial interest are migratory and use the Norwegian Sea as spawning grounds.

5.2. MODELS USED IN THE IASAP STUDY

5.2.1. Introduction

The models used in this study simulate the dispersion of radionuclides by advection and diffusion within the water column, and some include interaction with suspended material and sediment. Two modelling approaches were represented within the IASAP endeavour, namely compartmental or box models and hydrodynamic circulation models. In addition, a hybrid approach, which uses a compartmental structure but on a finely resolved spatial scale, was applied. Significantly differing modelling approaches were also taken for dealing with sedimentary processes and for modelling biological uptake. The modelling can be considered in two stages: first, the dispersal (by diffusion and advection) in the dissolved phase and, second, the interactions with sediment and biota. By not basing the IASAP study on a single model or model type, it was hoped that the overall results would prove robust. Indeed, the estimates of uncertainties in the endpoints reflect the uncertainties arising from lack of knowledge and paucity of data.

For modelling the advective and diffusive dispersal, compartmental models provide long time, spatially averaged (far field) capabilities, while the hydrodynamic models provide locally resolved, short time-scale results. Before turning to the specific models used within the project, the following subsections outline the main features of these model structures.

5.2.1.1. Compartmental models

Compartmental models are widely used in radiological assessment when there is a requirement for predictions in distant locations and over long time-scales. Such models are based on assumptions of instantaneous, homogeneous mixing within identified areas (compartments). Dispersion of the contaminants is parametrized by flows between the compartments, usually assumed to be time independent and proportional to the inventories of material within the compartments. The model typically contains more compartments in areas of high concentration gradient. Boxes may be depth stratified and may also include sediment–water interactions.

5.2.1.2. Hydrodynamic models

The second approach of hydrodynamic circulation models provides finely resolved spatial predictions based on calculated flow fields related to the driving forces in the system, such as wind, temperature and salinity. However, because of the high computational effort, such models can only be run for limited time-scales of the order of tens of years.

5.2.1.3. Advantages and disadvantages of the different modelling approaches

Compartmental models

The principal advantages of the compartmental models used here are:

- Model runs spanning long time-scales, up to thousands of years, are possible.
- The model is computationally expedient and low cost.
- The modelling is not limited to conservative radionuclides, but can be extended to deal with particle reactive nuclides (detailed submodels for dispersal in sediment and biota can be attached simply to the compartmental model).

The main disadvantages of the compartmental models used here are as follows:

- Owing to the high degree of spatial averaging implicit in the design of compartments, no concentration gradient can be created within a compartment. Concentration gradients can only be resolved

in the model by increasing the number of boxes in the region of the expected gradient. If these boxes are not included, the gradients cannot be described.
- Considerable uncertainties remain in some key parameters.

Hydrodynamic models

The principal advantages of the hydrodynamic models used here are:

- The ability to calculate two or three dimensional (2- or 3-D) flow fields based on realistic topography of the area and realistic forcing functions of wind and density.
- High temporal and spatial resolution, thus allowing for significant horizontal and vertical variations within small areas.

The main disadvantages of the hydrodynamic models used here are:

- The great numerical and computational effort involved in running such models limits the simulated time for dispersion forecasts to the order of decades.
- Much uncertainty resides in some of the key parameters, such as eddy viscosity and diffusivity coefficients.
- The considerable forcing data required for hydrodynamic models applied to the Arctic Ocean and Kara Sea: there is a major shortage of quality forcing data.
- Difficulties in the incorporation of sedimentary processes.

5.2.2. Description of the models used in IASAP

Both model types were necessary within IASAP, given their different space- and time-scale capabilities; both contributed significantly within the project. The individual modellers placed different emphases on processes occurring within the area and, consequently, the models may be considered complementary to each other in many respects. Table XIV summarizes the capabilities of each model. Particular models also contain differing data requirements. In the absence of the necessary site specific information, default values have been assigned. The descriptions below summarize the main features of the models.

TABLE XIV. SUMMARY OF MODEL PROPERTIES

Group	Model description
Denmark/Norway	Global: Compartmental model, with vertical stratification, linked to sediment submodel, predictions beyond 1000 years.
Japan	Regional: Compartmental model of Arctic region, with fine resolution grid, vertical stratification, detailed sediment model predictions up to 1000 years.
Netherlands	Regional: Compartmental model of Arctic region, linked to dynamic food chain model, predictions up to 1000 years.
Russian Federation	Regional: Compartmental model of Kara and Barents Sea, sediment submodel, linked to dynamic food chain model, predictions up to 100 years.
United Kingdom	Global: Compartmental model, with vertical stratification, linked to sediment submodel, predictions beyond 1000 years.
United States of America	Regional: 3-D multilevel baroclinic model for Arctic Ocean, no sediment model, linked to sea ice model, predictions up to ten years.
IAEA-MEL	Global: Compartmental model, with vertical stratification, linked to sediment submodel, predictions beyond 1000 years.
Germany/IAEA-MEL	Regional: 3-D baroclinic circulation model for Barents and Kara Seas, includes loss to sediment, linked to sea ice model, predictions up to 10 years.
	Local: 3-D baroclinic model for bay, predictions truncated after three years.

5.2.2.1. *Compartmental models*

Russian Federation

This regional compartmental model is designed for the Kara and Barents Seas. There is vertical stratification of the water column in the Kara Sea, including surface and deep waters of the Novaya Zemlya Trough. The removal of radionuclides from the water column to bottom sediments is calculated by using a particle scavenging model. Interactions of the sediment compartment with the water compartment include the processes of sorption, sedimentation, diffusion and bioturbation. The model is linked to a dynamic radioecological submodel, which calculates the dynamics of marine biota contamination in the Barents and Kara Seas. The effects of long distance fish migrations on radionuclide uptake are considered.

The Netherlands

This is a compartmental model covering the Kara and Barents Seas, the Arctic Ocean and far field locations. There is no vertical stratification. In the sediment layer, two boxes are distinguished to describe the downward and upward transport of radionuclides. Processes taken account of in the model are: particle scavenging/sedimentation, molecular diffusion, enhanced migration into solution due to physical and biological processes and burial. A dynamic biological submodel has also been developed and used in the dose estimation.

Denmark/Norway

This compartmental model covers the Arctic Seas and the North Atlantic, including European coastal waters. There is some vertical stratification in the water column to include surface and deep waters. The sediments are represented by two layers, a surface and deeper layer. Association of radionuclides with suspended sediment material is taken into consideration in addition to transfer to sediment through particle scavenging. Further transfer mechanisms include diffusion, bioturbation and resuspension.

United Kingdom

This is a compartmental model covering the Kara and Barents Seas, the Arctic Ocean and far field locations with no vertical stratification in the water column. Each water compartment has ambient suspended sediment concentration, with an interface compartment with suspended sediment concentrations ten times higher than in the water column and a bottom sediment compartment which is assumed to be a well mixed layer. There is no net sedimentation, but sediment can be transported in suspension between regions.

IAEA-MEL

This global scale compartmental model covers the world's oceans with increased spatial resolution in the Arctic and Atlantic regions. It includes vertical stratification of the water column in the Kara Sea. Sediment is represented as a single 0.1 m thick layer. Scavenging by particulates and removal of radionuclides from the dissolved phase to bottom sediment are described through a parametric model, with specific K_d, suspended sediment load and sedimentation rate prescribed for each water compartment.

5.2.2.2. *Hydrodynamic models*

Germany and IAEA-MEL

This is a three dimensional baroclinic circulation model, which is coupled to a free drift thermodynamic ice model. The model is implemented on a stereographic grid with an average size of 18 km and a domain which covers the Barents and Kara Seas. The vertical domain is resolved by ten layers. This model was further developed to include a sediment submodel, so that it might be used for particle reactive radionuclides.

A similar modelling approach also produced a local model for selected fjords of Novaya Zemlya.

United States of America

A three dimensional multilevel baroclinic ocean model is used; the model is linked to a Hibler ice model [88, 89]. Grid resolution is approximately 0.28° (17–35 km), with 15 vertical levels. It includes no interaction with suspended load or sediment. This model was used as a diagnostic tool to check estimates of water fluxes and to provide ice fluxes.

5.2.2.3. *Hybrid model*

Japan

A modified compartmental model has been designed to cover the local and regional fields (including the whole Arctic). The model is used to calculate the flow field based on temperature and salinity, which is then linked to diffusion calculations to derive the concentrations of radionuclides based on the sea water flow field.

The model includes adsorption and sedimentation due to the interaction with suspended matter, burying and elution due to interaction with seabed deposits and diffusion, as well as burying and bioturbation movement in seabed deposits.

5.2.3. Model validation

Clearly the process of model validation is important. Validation has generally been possible for those parts of the models that describe well characterized areas (e.g. the Irish Sea or the North Sea) and has made use of the extensive data sets available for radionuclides discharged from the Sellafield reprocessing plant. The models have been verified primarily against estimated transit times between Sellafield and northern waters. In addition, a comparison was made between literature values of water fluxes in the Arctic Ocean and those used by the compartment models; agreement was generally good.

Validation was difficult for the hydrodynamic models because of the lack of detailed information on essential variables, such as temperature, salinity fields or meteorological information for the Kara Sea.

In general, the validation process for both compartmental and hydrodynamic models was limited by the lack of appropriate data, particularly within the Arctic area.

5.3. SOURCE SCENARIOS FOR RADIOLOGICAL ASSESSMENT

Three release scenarios were used as a basis for radiological assessment:

(1) Scenario A: the ‘best estimate’ discharge scenario.
(2) Scenario B: the ‘plausible worst case’ scenario, a situation where a disruption, e.g. collision or munitions explosion, causes a complete breach of the containment surrounding the spent nuclear fuel from the icebreaker. This is be assumed to occur in the year 2050 under this scenario.
(3) Scenario C: the ‘climate change’ scenario refers to a major environmental disruption where global cooling followed by glaciation scours out the fjords. Subsequent warming would then release activity directly into the Kara Sea from all affected or crushed reactor cores. This release is assumed to occur in the year 3000, one thousand years from now, under this scenario.

Scenario A provides the release rate produced by corrosion processes. The corrosion rates used are lower than open sea values, as most of the radioactive material is partially isolated behind layers of containment. Scenario B, applied only to the icebreaker steam fuel container and generator installation, estimates corrosion processes up to the year 2050 as in Scenario A, followed by an accident which opens all containment barriers to normal seawater corrosion processes, accelerating the release rates. Scenario C models corrosion up to the year 3000, followed by a sudden release caused by a glacier riding over the dumped material. Scenario C is identical to Scenario A up to the year 3000, when a release peak occurs and all remaining radionuclides are released.

5.4. DOSE ESTIMATION AND EVALUATION OF RESULTS

For the scenarios outlined above, the IASAP team calculated maximum individual dose rates for the full range of relevant nuclides and considered decay chains where appropriate. The collective doses were calculated only for Scenario A.

5.4.1. Maximum individual dose rate

The maximum individual dose rate is of concern in all scenarios but particularly in B and C, which deal with extreme conditions. As the populations of interest are found only in the areas of the Kara and Barents Seas, results from a three dimensional hydrodynamic model are also included. It should be noted that, in some instances, owing to the source data, there may be several peaks in the dose rate at different times; Tables XV–XIX provide only the absolute maximum individual dose rate and time to maximum.

5.4.1.1. Definition of population (critical) groups

In the estimation of doses to individuals, three population groups were identified as follows:

Group 1

The first group comprises consumers of seafood, including sea mammals. Three such groups are assumed to exist, one residing on the Ob/Yenisey estuaries (significant population centres), the second on the Yamal peninsula (significant population centre) and the third on the Taymyr peninsula. This latter hypothetical group was placed at this location following the study of the potential contaminant dispersal which used the hydrodynamic model of Germany and IAEA-MEL showing maximum concentrations at this location. In the absence of detailed local data, these populations were assumed to have habits characteristic of subsistence fishing communities in other countries bordering the Arctic. The habits postulated for these populations were:

Marine consumption: fish, 500 g/d
sea mammals, 80 g/d
seabirds, 20 g/d
seabird eggs, 20 g/d

For the evaluation of external exposure, it was assumed that the population would spend 250 h per year on the shore.

Group 2

The second critical group represents individuals occupying the foreshore of the fjords containing dumped radioactive materials. These individuals are likely to be military personnel to whom food is supplied from elsewhere. The pathways to be considered include external exposure and inhalation of sea spray and resuspended sediment. Because of the harsh environmental conditions, an occupancy of 100 h/a was considered appropriate if not conservative. The concentration of radionuclides in beach material is taken to be 10% of that in seabed sediment to take into account the composition of sand and gravel. A similar assumption was made in the Marina Med Project of CEC [90].

Group 3

The third critical group is considered representative of the average local Russian population. This group was sited on the Kola peninsula, and fish consumption was taken as 50 kg/a (assumed caught in the Barents Sea) in addition to 0.5 kg/a of molluscs and 1 kg/a of crustaceans. Neither seaweed nor sea mammal consumption was considered, nor was an external exposure pathway included.

5.4.1.2. K_d values, concentration factors and dose conversion factors

Where possible, site specific values for the concentration factors and K_d values were used following the IASAP review of the available data (Tables VII and VIII of Section 3 above). Dose conversion factors from International Basic Safety Standards [9] were used. Some of the variations observed in the final dose results can be attributed to differences in the parameter values, and the importance of these parameters may also vary amongst the various models. It is worth noting that in the dynamic food chain models (Table XIII), concentration factors are not used directly [17].

5.4.1.3. Pathway exposure estimation

The pathway exposure modelling was based on the definitions of the critical groups and agreed sets of parameters. However, the various modelling approaches applied some of these parameters differently. Clearly, the fact that dynamic food chain modelling is not based on a concentration factor approach has particular implications for the modelling results for specific nuclides. There are also some differences in the assumptions used within the different models.

5.4.1.4. Results of Scenario A

Results from all sources combined

Table XV shows the estimates of the maximum individual dose rate for the identified population groups from all sources within the Kara Sea. For each population group, a range of maximum individual doses has been constructed for the peak dose calculated by each model. Whilst the decision to make a series of calculations for all sources combined was made specifically for the compartmental models which had lower spatial resolution, the results also provide an overall measure of the potential impact to the populations. It can be seen that, for the non-military populations, the maximum dose rates are very low and lie in the range of 5×10^{-12} to 6×10^{-8} Sv/a over the four locations where population groups 1 and 3 reside, with the ‘average’ Russian population on the Kola peninsula (Group 3) receiving the lowest maximum dose rate. In all these cases, the dose is mainly delivered through fish consumption, with ^{137}Cs and ^{239}Pu being the dominant nuclides. The results for the military group (Group 2) are higher although still low, with an estimated maximum dose rate of 2×10^{-5} Sv/a from external exposure and inhalation. This maximum dose rate would be delivered in the period 2100–2300.

Results from individual sources

Tables XVI(a) to XVI(d) show the results for each of the individual source locations.

It should be noted that the estimated maximum dose rate obtained for releases in particular fjords can be higher than the maximum dose rate corresponding to all sources combined. This is because some modellers give results only for combined sources and some only for individual fjords. The difference reflects the differences

TABLE XV. RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, ALL SOURCES COMBINED, SCENARIO A

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$3 \times 10^{-10} - 6 \times 10^{-8}$	$3 \times 10^{-10} - 8 \times 10^{-9}$	$3 \times 10^{-10} - 3 \times 10^{-9}$	$3 \times 10^{-8} - 2 \times 10^{-5}$	$5 \times 10^{-12} - 2 \times 10^{-9}$
In years	2100–2500	2100–2500	2100–2500	2100–2300	2000–2400
Dominant[a] pathways	Ingestion, inhalation	Ingestion	Ingestion	Inhalation, external exposure	Ingestion
Dominant[a] nuclides	Cs-137 Pu-239 Pu-240	Cs-137 Pu-239	Cs-137 Pu-239	Cs-137 Pu-239	Cs-137 Pu-239 Pu-240

[a] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVI(a). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, ABROSIMOV FJORD SOURCES, SCENARIO A

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-10} - 6 \times 10^{-8}$	$2 \times 10^{-10} - 1 \times 10^{-8}$	$2 \times 10^{-10} - 3 \times 10^{-9}$	$2 \times 10^{-9} - 7 \times 10^{-5}$	$5 \times 10^{-11} - 2 \times 10^{-9}$
In years	2100	2100	2100–2300	2100–2700	2100
Dominant[a] pathways	Ingestion, external exposure	Ingestion, external exposure	Ingestion	Ingestion, external exposure	Ingestion
Dominant[a] nuclides	Cs-137	Cs-137	Cs-137	Pu-239 Pu-240	Cs-137

[a] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVI(b). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, TSIVOLKA FJORD SOURCES, SCENARIO A

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-10} - 3 \times 10^{-8}$	$3 \times 10^{-10} - 7 \times 10^{-8}$	$3 \times 10^{-10} - 5 \times 10^{-9}$	$2 \times 10^{-8} - 6 \times 10^{-4}$	$7 \times 10^{-12} - 6 \times 10^{-10}$
In years	2300–2500	2300–2500	2300–2500	2300	2300–2400
Dominant[a] pathways	Ingestion, external exposure	Ingestion, external exposure	Ingestion	Inhalation, external exposure	Ingestion
Dominant[a] nuclides	Pu-239 Pu-240	Pu-239 Pu-240 Cs-137	Pu-239 Pu-240	Pu-239 Am-241	Cs-137 Pu-239 Pu-240

[a] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVI(c). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, TECHENIYE FJORD SOURCE, SCENARIO A[a]

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-16} - 2 \times 10^{-11}$	$2 \times 10^{-16} - 4 \times 10^{-13}$	$2 \times 10^{-16} - 1 \times 10^{-11}$	8×10^{-12}	$5 \times 10^{-19} - 7 \times 10^{-17}$
In years	2000–2200	2000–2200	2000–2200	2000	2000–2200
Dominant[b] pathways	Ingestion, external exposure	Ingestion, external exposure	Ingestion, external exposure	External exposure	Ingestion
Dominant[b] nuclides	Co-60 Ni-59	Co-60 Ni-59	Co-60 Ni-59	Co-60	Co-60 Ni-59

[a] It should be noted that dose rates of the order of magnitude 10^{-16}–10^{19} correspond to only few radioactive decays in a human lifetime and are thus radiologically meaningless. For comprehensiveness, the calculated figures are, however, included in the report.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVI(d). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, NOVAYA ZEMLYA TROUGH SOURCE, SCENARIO A

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-11} - 2 \times 10^{-10}$	$5 \times 10^{-11} - 9 \times 10^{-9}$	$2 \times 10^{-12} - 5 \times 10^{-10}$	$1 \times 10^{-14} - 2 \times 10^{-6}$	$2 \times 10^{-12} - 3 \times 10^{-10}$
In years	2000–2100	2000–2100	2000–2100	2040–2060	2000–2100
Dominant[a] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[a] nuclides	Cs-137 Sr-90	Cs-137 Sr-90	Cs-137	Sr-90	Cs-137 Sr-90

[a] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVII(a). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, TSIVOLKA FJORD SOURCES, SCENARIO B

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$3 \times 10^{-10} - 7 \times 10^{-7}$	$3 \times 10^{-10} - 3 \times 10^{-7}$	3×10^{-10} [a]	$1 \times 10^{-7} - 4 \times 10^{-3}$	$1 \times 10^{-10} - 2 \times 10^{-8}$
In years	2100	2100	2100	2100	2100–2200
Dominant[b] pathways	Ingestion, external exposure	Ingestion, external exposure	Ingestion	Ingestion, external exposure	Ingestion
Dominant[b] nuclides	Cs-137 Pu-239 Pu-240	Cs-137 Pu-239 Pu-240	Cs-137 Pu-239 Pua -240	Pu-239 Sr-90 Pu-240	Pu-239 Cs-137

[a] Only one result

[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVII(b). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, ALL SOURCES COMBINED — TSIVOLKA FJORD SOURCES, SCENARIO B, ALL OTHER SOURCES ACCORDING TO SCENARIO A

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$1 \times 10^{-7} – 3 \times 10^{-7}$	$4 \times 10^{-8} – 1 \times 10^{-7}$	1×10^{-7} [a]	$1 \times 10^{-5} – 4 \times 10^{-4}$	$4 \times 10^{-10} – 3 \times 10^{-9}$
In years	2053	2052–2055	2052	2050	2054
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[b] nuclides	Cs-137 Pu-239 Pu-240	Cs-137 Pu-239 Pu-240	Cs-137 Pu-239 Pu-240	Pu-239 Pu-240	Cs-137 Pu-239 Pu-240

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XVIII. RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, ALL SOURCES COMBINED, SCENARIO C

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$4 \times 10^{-9} – 3 \times 10^{-7}$	$4 \times 10^{-9} – 3 \times 10^{-7}$	$4 \times 10^{-9} – 3 \times 10^{-7}$	$8 \times 10^{-10} – 2 \times 10^{-4}$	$6 \times 10^{-10} – 6 \times 10^{-9}$
In years	3000–3121	3000–3121	3000–3121	2300–3000	3000–3089
Dominant[a] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[a] nuclides	Pu-239	Pu-239	Pu-239	Pu-239 Pu-240	Pu-239 Pu-240

[a] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

between the models used and the uncertainties in marine modelling. However, the doses calculated by models which were run for all sources simultaneously are within an order of magnitude of the doses which would be obtained by summing up the distinct sources (Section 7).

Abrosimov Fjord. Table XVI(a) shows the results for discharge in Abrosimov Fjord, where several reactors have been dumped. Maximum dose rates for the non-military populations lie in the range between 5×10^{-11} Sv/a at Kola (marine produce caught in the Barents Sea) and 6×10^{-8} Sv/a at Yamal (marine produce from the Kara Sea and external pathways). The maximum dose rate is delivered in the period of 2100–2300 and is due to ^{137}Cs with ingestion as the dominant pathway. However, one model gives external exposure as the dominant pathway for all groups except those at Kola and at Ob/Yenisey. Again, the maximum dose rate to military personnel, as might be expected, is higher, with a maximum value 7×10^{-5} Sv/a also delivered in the period of 2100–2700. Inhalation and external exposure are the dominant pathways.

Tsivolka Fjord. Table XVI(b) illustrates the results for Tsivolka Fjord, site of the icebreaker reactors and the fuel container. Results range from a minimum of 7×10^{-12} Sv/a at the Kola peninsula to a maximum of 7×10^{-8} Sv/a at Taymyr, with the maximum dose rate delivered in the period 2300–2500. The dominant pathway is ingestion (with one model as an exception) with ^{239}Pu and ^{137}Cs being the major contributors to the dose. For the military group, dose rates are again higher, with a maximum value of 6×10^{-4} Sv/a delivered at about the year 2300, inhalation being the dominant pathway.

Techeniye Fjord. Table XVI(c) provides the results obtained in two compartmental models for sources in Techeniye Fjord where two reactors have been dumped without fuel. Maximum dose rates to the non-military populations at Kola are low, 5×10^{-19} to 7×10^{-17} Sv/a.[2] For the other population locations, dose rates are in the range of 2×10^{-16} to 2×10^{-11} Sv/a.[2] The military group dose rate is 8×10^{-12} Sv/a. For the non-military group, ingestion is the dominant pathway, with ^{60}Co and ^{59}Ni being the dominant nuclides.

Novaya Zemlya Trough. Table XVI(d) presents the results for the Novaya Zemlya Trough. This location is significantly deeper than any of the other source locations and there is significantly more variation in the model results close to the source (some of the reasons for this are discussed in Section 5.4.4). The vertical structures of the models differ and their treatment of the trough (sometimes as a separate entity within the model) accounts for these differences. Model sensitivity to vertical structure and hence migration of radionuclides was demonstrated in the initial benchmarking exercises. We also see differences in the timing of the delivery of the dose, again reflecting different model structures.

The results for all non-military groups are low, in the range of 2×10^{-12} to 9×10^{-9} Sv/a. Fish consumption dominates here with ^{137}Cs and ^{90}Sr contributing substantially. The results for the military group are also low, i.e. 1×10^{-14} to 2×10^{-6} Sv/a.

5.4.1.5. Results of Scenario B

Scenario B was devised to simulate a ‘plausible accident’, and the disposal site of the icebreaker reactors and the fuel container, Tsivolka Fjord, was chosen for the exercise. Disruption of the source was assumed to occur at the year 2050, resulting in a changed discharge pattern after this date. Up to 2050, the discharge pattern is identical to the one found in Scenario A.

Maximum individual dose due to Tsivolka Fjord

Table XVII(a) contains the results for Scenario B. In this projection, maximum dose rates to the non-military populations are in the range of 3×10^{-10} to 7×10^{-7} Sv/a. For the military personnel, there has been an increase in maximum dose rate (up to 4×10^{-3} Sv/a at its highest), and the year of delivery immediately follows the disruptive event. Table XVII(b) shows the results for Scenario B applied to the sources in Tsivolka Fjord, including all the other sources provided in Scenario A. Again, for non-military populations, the results obtained are low.

5.4.1.6. Results of Scenario C

It should be noted that, for Scenario C, the doses arising from peak releases in the year 3000 may, as a consequence of radioactive decay, be lower than the maximum doses in earlier years.

Results of all sources combined

Table XVIII shows the results of all sources combined under the scenario which projects the total release of all remaining inventory in the year 3000. For Group 1, the maximum dose rate lies in the range of 4×10^{-9} to 3×10^{-7} Sv/a; for Group 3, the maximum dose rate lies in the range of 6×10^{-10} to 6×10^{-9} Sv/a.

[2] It should be noted that dose rates of the order of magnitude 10^{-16}–10^{-19} correspond to only few radioactive decays in a human lifetime and are thus radiologically meaningless. For comprehensiveness, the calculated figures are, however, included in the report.

These values are higher than the corresponding values for Scenario A, but remain low. The dominant pathway is ingestion, with ^{239}Pu acting as the major contributing nuclide. For Group 2, the maximum dose rate is in the range 8×10^{-10} to 2×10^{-4} Sv/a, with the dominant nuclide being ^{239}Pu and ^{240}Pu obtained through inhalation.

Results of individual sources

Tables XIX(a) to XIX(d) separately present the maximum individual dose rates from each of the sources. Three separate models were used to obtain values.

Abrosimov Fjord. Table XIX(a) represents the results for Abrosimov Fjord. Maximum dose rates for Group 1 lie in the range of 2×10^{-10} to 2×10^{-8} Sv/a. For Group 3, at Kola, results lie in the range of 2×10^{-11} to 5×10^{-10} Sv/a. In all cases, ^{137}Cs is the dominant nuclide. For Group 2 (military personnel), the maximum dose rate range is 5×10^{-9} to 2×10^{-4} Sv/a, and the dominant nuclides include ^{90}Sr, ^{239}Pu and ^{240}Pu.

Tsivolka Fjord. Table XIX(b) illustrates the results for Tsivolka Fjord. Maximum dose rates for Groups 1 and 3 lie in the range of 2×10^{-10} to 2×10^{-7} Sv/a with ^{239}Pu and ^{240}Pu dominating. For the military group (Group 2), the maximum dose rates lie in the range of 10^{-7} to 3×10^{-3} Sv/a.

Techeniye Fjord. Table XIX(c) presents the results for Techeniye Fjord. Maximum dose rates for all groups are very low ($<3 \times 10^{-11}$ Sv/a). The dominant nuclides are ^{59}Ni and ^{63}Ni.

Novaya Zemlya Trough. Table XIX(d) gives the results for the Novaya Zemlya Trough. Maximum dose rates for Group 1 are less than 4×10^{-9} Sv/a and less than 8×10^{-11} Sv/a for Group 3. For Group 2, the maximum dose rate is less than 2×10^{-6} Sv/a, and the dominant nuclides are ^{90}Sr and ^{137}Cs.

5.4.1.7. Comments

To provide some perspective on the individual dose estimates, it should be noted that the dose rate delivered through the consumption of fish, molluscs and crustaceans from ^{210}Po (a natural radionuclide) to:

(a) Population group 3, on the Kola peninsula, is estimated to be 1×10^{-4} Sv/a;
(b) Population group 1, on the Taymyr peninsula, is estimated to be 5×10^{-4} Sv/a. (This dose arises from fish consumption alone.)

These figures are based on ^{210}Po concentrations for FAO fishing area 27 (north-east Atlantic) [91].

5.4.2. Collective dose calculations

The exposure pathways to be considered include consumption of fish, crustaceans, molluscs, seaweed and mammals. To facilitate the calculation of collective doses, certain basic data were selected.

5.4.2.1. Fishery statistics

Fishery statistics used were based on worldwide FAO data for 1990 [92] and supplemented by ICES [93] data for the Barents, Norwegian Seas and Spitsbergen areas. Sea mammal catches were also included whenever possible. Some modellers, rather than basing their results on a single year that might not be representative, made use of five year average fish catches in these areas. For the calculations, it was assumed that 50% of the fish catch, 30% of the crustacean catch, 15% of the mollusc catch and 10% of the seaweed harvest would be consumed. These figures are widely accepted for radiological protection purposes and were recently reviewed in a major international assessment of the radiological implications of radionuclides in the marine waters of northern Europe [94]. In particular, it should be noted that the figure of 50% for fish catch takes account of fish caught for industrial purposes, i.e. used for the production of animal feed.

For the Kara Sea, additional assumptions were necessary because no detailed information was available. These were that 2200 tonnes of fish were caught annually in the southern Kara Sea, and 100% was consumed, as it is likely that fish are not caught for industrial purposes in this area. No crustacean or mollusc catch was included. The sea mammal catch limits for the Kara Sea were taken as actual catches corresponding to 115 tonnes per year but for the Barents Sea, a catch of 63 tonnes per year was assumed, of which 50% was consumed [95].

5.4.2.2. Truncation times

Long lived radionuclides may cause exposures over thousands of years. However, there are increasing uncertainties in calculating doses to populations beyond a few hundred years. Hence, “in decision making, less significance should be attached to collective dose estimates relating to periods beyond 500 years into the future than to those relating to shorter time periods” [96]. Therefore, after considering the postulated releases, it was decided to select truncation dates of 2050 (approximately 50 years) and 3000 (approximately 1000 years). All collective dose calculations are based on compartmental models, since the hydrodynamic models are designed to

TABLE XIX(a). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, ABROSIMOV FJORD SOURCES, SCENARIO C

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-10} – 2 \times 10^{-8}$	$2 \times 10^{-10} – 2 \times 10^{-8}$	2×10^{-10} [a]	$5 \times 10^{-9} – 2 \times 10^{-4}$	$2 \times 10^{-11} – 5 \times 10^{-10}$
In years	2040–2080	2040–2080	2080	2050–2730	2040–2080
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[b] nuclides	Cs-137	Cs-137	Cs-137	Sr-90 Pu-239 Pu-240	Cs-137

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XIX(b). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, TSIVOLKA FJORD SOURCES, SCENARIO C

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-9} – 2 \times 10^{-7}$	$1 \times 10^{-9} – 2 \times 10^{-7}$	2×10^{-9} [a]	$1 \times 10^{-7} – 3 \times 10^{-3}$	$2 \times 10^{-10} – 5 \times 10^{-9}$
In years	3000–3120	3000–3120	3120	2300–3000	2300–3100
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[b] nuclides	Pu-239 Pu-240	Pu-239 Pu-240	Pu-239	Pu-239 Pu-240	Pu-239 Pu-240

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XIX(c). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, TECHENIYE FJORD SOURCE, SCENARIO C

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$1 \times 10^{-16} - 3 \times 10^{-11}$	$1 \times 10^{-16} - 3 \times 10^{-11}$	1×10^{-16} [a]	$1 \times 10^{-12} - 3 \times 10^{-12}$	$4 \times 10^{-17} - 6 \times 10^{-13}$
In years	3000–3100	3000–3100	3100	2000–2660	3000–3080
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation, external	Ingestion
Dominant[b] nuclides	Ni-59 Ni-63	Ni-59 Ni-63	Ni-63	Ni-59 Ni-63 Co-60	Ni-59 Ni-63

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XIX(d). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, NOVAYA ZEMLYA TROUGH SOURCE, SCENARIO C

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$8 \times 10^{-11} - 4 \times 10^{-9}$	$2 \times 10^{-11} - 4 \times 10^{-9}$	8×10^{-11} [a]	$1 \times 10^{-14} - 2 \times 10^{-6}$	$3 \times 10^{-12} - 8 \times 10^{-11}$
In years	2080–3000	2040–3000	2080	2040–2060	3000–3050
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[b] nuclides	Pu-239 Pu-240 Cs-137	Cs-137 Sr-90 Pu-239 Pu-240	Cs-137	Sr-90 Cs-137	Pu-239 Pu-240 Am-241

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

deal with shorter time-scales. This discussion is continued in Section 7, where the globally dispersed radionuclides ^{14}C and ^{129}I are also dealt with.

5.4.2.3. Results

Table XX provides the evaluated collective doses to the years 2050 and 3000. Results lie in the range of 3×10^{-3} to 1×10^{-2} man·Sv and 8×10^{-2} to 9×10^{-1} man·Sv, respectively, which are generally very low. The dominant pathway is fish consumption. In both cases ^{137}Cs is the main contributor. ^{239}Pu and ^{90}Sr also give significant contributions to collective doses to the year 3000.

5.4.2.4. Comments

The collective doses to the years 2050 and 3000 estimated by different modelling groups are in both cases in good agreement and show the low collective dose delivered to the world population as a result of the dumping in the Kara Sea. The variation in the results is small and can be explained by the differences in some of the necessary input data for the different models. The same can also be said about the differences in the contributions of the marine pathways. In addition, the spatial domain of the models varies and this may also lead to minor differences in collective dose estimates.

5.4.3. Submarine No. 601

The liquid metal reactors of the submarine No. 601 in Stepovoy Fjord presented some particular difficulties to the modellers due to the very long release time under the most likely release scenario. Nevertheless, the results for both Scenarios A and C are illustrated in Tables XXI(a) and XXI(b).

5.4.3.1. Scenario A

Table XXI(a) shows that at Yamal, Taymyr, Kola and Ob/Yenisey, the maximum dose rates are very low and lie in the range of 7×10^{-14} to 6×10^{-10} Sv/a. Ingestion is the dominant pathway. The main contributing nuclides are ^{137}Cs and ^{239}Pu. For the military group, only one value is reported, 8×10^{-12} Sv/a.

5.4.3.2. Scenario C

Table XXI(b) provides the results for Scenario C. Results in all cases of public exposure are less than 3×10^{-8} Sv/a and for Group 2 less than 7×10^{-8} Sv/a. ^{239}Pu is the dominant nuclide.

TABLE XX. COLLECTIVE DOSE, SCENARIO A: ESTIMATED CONTRIBUTIONS FROM EACH COMPONENT OF THE FOOD CHAIN

	Range of collective dose (man·Sv)	
	To year 2050	To year 3000
	$3 \times 10^{-3} - 1 \times 10^{-2}$	$8 \times 10^{-2} - 9 \times 10^{-1}$
Fish	16–99%	22–99%
Crustaceans	6.5–64%	1–75%
Molluscs	20.1%	1–32%

5.4.3.3. Special nuclides

For modelling purposes, the submarine No. 601 is unlike the other sources disposed of in the Kara Sea because of the very long release period (thousands of years) and a number of more exotic radionuclides which must be considered (e.g. ^{79}Se and ^{205}Pb). Some consideration was given to such nuclides in terms of contribution to the collective dose in the year 3000. The results are outlined in Table XXI(c). The contribution of these nuclides is very small.

5.4.4. Model intercomparison

The interpretation of the dose estimates and their variation can be aided by comparison of the primary environmental modelling results and the factors which influence them. One way to do this is through model intercomparison. An extensive intercomparison was conducted, and a summary of the key results is presented below.

5.4.4.1. General comments

This discussion draws attention to some of the factors that are relevant when evaluating the final dose results. The scope and philosophy of the hydrodynamic and compartmental models are generally very different. As already mentioned, the time span of the forecast and the spatial resolution of the model types differ considerably. The compartmental model gives a ‘box’ integrated value, which is assumed representative of a region covering several thousand square kilometres whereas the hydrodynamic model gives a ‘point’ value representing a much smaller area (in the range of a few to a few tens of square kilometres). Furthermore, the hydrodynamic model is able to resolve seasonal or even tidal cycles of concentration. The comparison becomes more difficult for particle reactive nuclides owing to the effects of

TABLE XXI(a). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, STEPOVOY FJORD SOURCE NS601, SCENARIO A

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$1 \times 10^{-12} – 3 \times 10^{-12}$	$5 \times 10^{-14} – 6 \times 10^{-10}$	$1 \times 10^{-13} – 4 \times 10^{-11}$	8×10^{-12} [a]	$7 \times 10^{-14} – 3 \times 10^{-13}$
In years	2200–7700	2200–7700	2200–7700	2200–7700	2200–7700
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[b] nuclides	Pu-239 Cs-137	Pu-239 Cs-137	Pu-239 Cs-137	Pu-239	Pu-239 Cs-137

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XXI(b). RANGES OF MAXIMUM INDIVIDUAL DOSE RATES, STEPOVOY FJORD SOURCE, SCENARIO C

Population group		1		2	3
Location	Yamal	Taymyr	Ob/Yenisey	Military	Kola
Individual dose rates (Sv/a)	$2 \times 10^{-9} – 3 \times 10^{-8}$	$1 \times 10^{-10} – 3 \times 10^{-8}$	2×10^{-9} [a]	$5 \times 10^{-8} – 7 \times 10^{-8}$	$3 \times 10^{-11} – 6 \times 10^{-10}$
In years	3000–3130	3000–3130	3130	3000–3020	3000–3100
Dominant[b] pathways	Ingestion	Ingestion	Ingestion	Inhalation	Ingestion
Dominant[b] nuclides	Pu-239	Pu-239	Pu-239	Pu-239 Pu-240	Pu-239

[a] Only one result.
[b] Defined as contributing more than 20% to the individual dose rate by at least one modeller.

TABLE XXI(c). CONTRIBUTION OF SPECIAL NUCLIDES TO COLLECTIVE DOSE (man·Sv) TO YEAR 3000, STEPOVOY FJORD, SCENARIO A

Nuclide	IAEA-MEL	KEMA
Ni-63	4×10^{-4}	–
Se-79	6×10^{-6}	–
Cs-135	2×10^{-7}	–
I-129	–	5×10^{-7}
Zr-93	2×10^{-9}	4×10^{-7}
Tc-99	–	5×10^{-8}
Bi-210m	–	1×10^{-8}
Sn-121m	2×10^{-9}	–
Eu-152	6×10^{-11}	–
Cd-113m	5×10^{-11}	–
Pd-107	2×10^{-11}	9×10^{-13}
Pb-205	7×10^{-13}	–
Eu-154	9×10^{-14}	–
Total	4×10^{-4}	9×10^{-7}

numerical diffusion on such nuclides. The treatment of vertical migration of radionuclides in the water column is also different between the model types used in the IASAP. This may have substantial repercussions on predicted concentrations in the sediment and the overlying water column. The final dose calculations may then be affected, e.g. when considering ingestion pathways for fish, the concentrations in the upper layers would be important, while for molluscs and crustaceans, the bottom concentrations would be important. With regard to the importance of the sedimentary processes, a small sensitivity study of changes in sedimentary parameters was undertaken which pointed to differences between predicted sediment concentrations and showed that the degree of sensitivity to the parameters was model dependent. In addition to sediment, two quite different approaches have also been taken to modelling concentrations in biota. Several models followed the traditional approach, using concentration factors to give concentrations in biota, while others developed dynamic food chain models. A preliminary sensitivity analysis showed that a specific dynamic food chain model resulted in doses approximately 15% higher than the traditional concentration factor approach.

5.4.4.2. Design of the intercomparison

A detailed intercomparison scenario was developed with the aim of assessing quantitatively the comparability of the results from the different models (one aspect of model reliability) and to investigate the influence of the different model structures on the results.

Four nuclides (^{137}Cs, ^{239}Pu, ^{99}Tc and ^{60}Co) were considered. These nuclides were chosen owing to their varying half-lives and different particle reactivities in the marine environment. Moreover, with the exception of ^{99}Tc, they were selected on the basis of the radiological impact and their importance in the source term inventory. Two release patterns (instantaneous and fixed rate per year for ten years) were used for the sources sited in an eastern bay of Novaya Zemlya (Abrosimov Fjord, 71°N 55°E) and in the Novaya Zemlya Trough (72°N 58°E). The endpoints were selected to match the needs of the IASAP objectives, in particular, the spatial and temporal scales required for the radiological assessment. Participants provided maximum concentrations in sea water (Bq/m^3) and sediment (Bq/kg dw and Bq/m^2) within the Kara and Barents Seas (three locations within each) and at a number of other locations around the Arctic (including the Chukchi Sea, central Arctic, Beaufort Sea, Davis Strait and Iceland Sea), and the time (in years) at which the maximum occurred.

5.4.4.3. Results and conclusions of the intercomparison

Brief summaries of the results relating to an instantaneous release of ^{137}Cs and ^{239}Pu from both Abrosimov Fjord and Novaya Zemlya Trough are presented below. These two nuclides were selected because of their importance in the radiological assessment and because of their different particle reactivities. The comparison is also based only on compartmental models, since the comparison with hydrodynamic models is only possible on a much more selective basis. However, where these comparisons were possible, the results were in good agreement. The comparisons presented here are relevant to the interpretation of the variation in dose estimates. A more detailed analysis can be found in Refs [17] and [97].

Caesium-137

In general, for a release from Abrosimov Fjord, the agreement in maximum concentrations in water is better than in sediment. The level of agreement is typically about one order of magnitude for water at near source locations. The disagreements can at least in part be explained by the different model structures and some differences in the flow fields. Within the Kara Sea, the time to maximum concentration is also relatively short (typically, less than four years). Differences in agreement in the Kara and Barents Seas are due to the different definitions of boxes and flows used by the modellers. For instance, one of the differences between the models is the assumption of the existence and magnitude of the

Litke current which flows from the Kara Sea to the Barents Sea.

For sediment, the level of agreement is generally better than three orders of magnitude and tends to fall off with distance from the source. Poorer agreement at distant locations will, for radiological assessment purposes, present no difficulties, as it is near source concentrations that are important for dose delivery. Agreement for the Kara Sea is within one order of magnitude. The greater variation in sediment results is likely to be due to the parametrization of sedimentary processes and to the assumed suspended loads in the model compartments. The times to maximum radionuclide concentration in sediment are also longer than those in sea water (up to 12 years).

Such patterns of variation are repeated when the other source location is considered. In general, results for the Novaya Zemlya Trough show more variation, again partly explained by the different spatial and vertical resolutions of the models within the Kara Sea.

Plutonium-239

In general, the level of agreement in maximum water concentrations for ^{239}Pu is comparable to that for ^{137}Cs. For releases from either Abrosimov Fjord or Novaya Zemlya Trough, the level of agreement is better than two orders of magnitude in both the Kara and Barents Seas. The results for maximum sediment concentrations also show agreement better than two orders of magnitude near source, falling off with distance. The fall-off with distance is more pronounced for ^{239}Pu than ^{137}Cs. This again emphasizes the need, within the compartmental structure, for sufficient boxes (i.e. sufficient spatial resolution) to reproduce the concentration gradient. There is a noticeable sensitivity to model structure of time to maximum concentration in sediment. This may have some impact on relative contributions of particle reactive radionuclides and exposure pathways in the assessment of radiation dose rate.

5.5. OTHER TRANSPORT MECHANISMS

One further exceptional transport pathway was also considered during the radiological impact assessment of the solid wastes dumped in the Kara Sea, specifically, the transport of contaminated sediment in sea ice.

5.5.1. The effect of sea ice

One special feature of the area which was not explicitly modelled is the transport of sediment bound radionuclides by sea ice. The best available information identifies this mechanism as a potential pathway for redistribution of the radionuclides but there is a lack of quantitative information necessary for conducting an actual calculation. Therefore, an evaluation of the magnitude of this particular pathway is addressed by a scoping calculation. The results of the scoping calculation are based on a number of broad assumptions which are further identified below.

5.5.1.1. Sediment transport in ice

A considerable amount of sediment or particulate material is transported from shallow Siberian shelves by the transpolar ice drift towards the Fram Strait. Through the melting of ice, the material is released to the marine environment of the Greenland/Iceland Seas. Estimates of the total annual sediment export by sea ice through Fram Strait range from 7 million tonnes up to 150 million tonnes [98]. The Laptev Sea is one of the most important source regions and contributes approximately 28% of deep sea sediment accumulation in the European Nordic Seas. The transpolar drift is by far the shortest pathway for contaminant dispersal from the Arctic. The sediment usually accumulates at the surface as patches or surface layers on the ice. The highest concentrations of sediment can usually be found on the surface of multiyear ‘old’ ice, with observed values in the central Arctic ranging from 10 mg/L to 56 000 mg/L [99].

5.5.1.2. Estimating the transport from the Kara Sea

The ice export from the Kara Sea to the Arctic Ocean has been estimated from three different locations within the Kara Sea on the basis of the US Naval Research Laboratory’s three dimensional model of the Arctic [100]. The three areas considered were two along the Taymyr coastline, and the third on the eastern coastline of Novaya Zemlya; estimated ice transport quantities for these areas were 167, 209 and 128 km^3/a. Assumptions were made concerning the sediment load of the sediment loaded ice and the proportion of ice passing through the Fram Strait. It was then possible to estimate the magnitude of this pathway for the transport of radionuclides from the areas of the dumped wastes first to the Arctic Ocean and then via the transpolar drift to the Fram Strait.

5.5.1.3. A simple scoping calculation

Assuming a concentration of 60 Bq/kg ^{137}Cs in sediments (corresponding to a maximum concentration of ^{137}Cs from sources in Abrosimov Fjord, as predicted by the three dimensional model 3DMEL), a sediment load in ice of 3 kg/m^3, an ice flux of 128 km^3/a and, ignoring the possibility that only 20% of ice is dirty [101], 4×10^{11} kg of sediment are exported to the Arctic per year, resulting in a net transport of 24 TBq/a. (It must be noted that measurements of radionuclide concentrations in sea ice sediment of up to 70 Bq/kg have been reported [102]). Extending this calculation, 3×10^3 km^3/a of ice is transported from the Arctic through the Fram Strait, compared with 2×10^5 km^3/a of sea water. If we assume (unrealistically) that all such ice had been formed in the Kara Sea under the conditions described above, and that all ice so formed drifts to the Fram Strait (it is estimated that there is a 70% probability of drift to Fram Strait [103], then 7×10^{12} kg of sediment would be transported per year to the Nordic Seas, corresponding to a radionuclide transport of 0.4 PBq/a of ^{137}Cs. Considering the dissolved transport of radionuclides (the corresponding model maximum concentration in sea water is 2100 Bq/m^3), then approximately 410 PBq/a of ^{137}Cs would be transported. Thus, the transport of radionuclides in sea ice sediment is three orders of magnitude less than the corresponding transport in the dissolved phase. These figures scope the magnitude of this rather unusual transport pathway and show that, in this context, it is dominated by the dissolved transport of radionuclides.

5.6. IMPACT OF DUMPING ON POPULATIONS OF WILD ORGANISMS

Estimates of the incremental radiation exposure arising from radionuclides released from the packaged waste materials dumped into the Kara Sea provide the necessary basis for an assessment of the potential impact of the practice on populations of wild organisms. Dosimetry models that allow the estimation of radiation dose rates to a variety of aquatic organisms, from both internal and external sources, have been developed [104–108] and utilized and, where necessary, extended for the present assessment. It is clearly not possible to make an assessment for every organism native to the Arctic Seas, but it is necessary to include a sufficient number of types both to provide a realistic indication of the range of possible incremental exposures and to explicitly recognize the particular species, e.g. seals and whales, typical of this environment. The organisms considered and the geometrical models adopted to represent them are summarized in Table XXII. Most of the dosimetry models have been used in previous assessments, and the relevant references are also indicated. The model for the whale is new to this assessment, and it is sufficiently large in relation to the ranges of α and β particles that the corresponding dose rates are $D_\alpha(\infty)$ and $D_\beta(\infty)$ [105]. For γ radiation, the absorbed fraction has been extrapolated from the data published by Brownell et al. [109].

Since, other factors being equal, the highest concentrations and therefore the highest doses are likely to occur in the immediate vicinity of the dumped wastes, the peak concentrations estimated for the release from the wastes dumped in Abrosimov Fjord have been used as the basis for dose rate estimates. The radionuclide concentrations in water (averaged across the Abrosimov Fjord) and sediment (averaged over the area within 100 m of the dumped wastes) are given in Table XXIII for the Case A release scenario. Radionuclide concentrations in the organisms have been estimated using either the new data for concentration factors given in Table VIII of Section 3 or, in default, the data set out in Ref. [72]. The dose rate estimates are summarized in Table XXIV, with the contributions from α radiation (high LET (linear

TABLE XXII. GEOMETRICAL MODELS ADOPTED TO REPRESENT MARINE ORGANISMS

Organism	Mass (kg)	Length of major axes of the representative ellipsoid (cm)	References
Pelagic zooplankton, benthic crustaceans	1.0×10^{-6}	0.62 × 0.31 × 0.16	[104, 110]
Benthic molluscs	1.0×10^{-3}	2.5 × 1.2 × 0.62	[104, 110]
Pelagic fish	1.0	45 × 8.7 × 4.9	[104, 110]
Birds	6.0×10^{-1}	21 × 16 × 11[a]	[106]
Seals	58	180 × 35 × 19	[108]
Whales	1000	450 × 87 × 48	This study

[a] A bird was assumed to be an ellipsoid of solid tissue covered with a layer of feathers [104].

TABLE XXIII. PREDICTED PEAK RADIONUCLIDE CONCENTRATIONS IN WATER, SEDIMENT AND ORGANISMS USED TO ESTIMATE DOSE RATES

Nuclide	Water[a] (Bq/L)	Sediment[a] (Bq/kg)	Zooplankton (Bq/kg)	Molluscs (Bq/kg)	Fish (Bq/kg)	Birds (Bq/kg)	Seals (Bq/kg)	Whales (Bq/kg)
Ni-59	5.8×10^{-3}	18	5.8	12	0.58	4.1	–	–
Ni-63	8.7×10^{-3}	27	8.7	17	0.87	6.1	–	–
Co-60	9.3×10^{-3}	29	19	2.9	9.3	–	–	–
Sr-90	1.9	5.3	1.9	1.9	7.6	–	1.9	1.9
Cs-137	2.1	59	64	2.1	210	210	210	210
Pu-239	8.2×10^{-3}	24	8.2	25	0.33	0.82	2.5×10^{-2}	2.5×10^{-2}
Pu-240	4.1×10^{-3}	11	4.1	12	0.16	0.41	1.2×10^{-2}	1.2×10^{-2}
Pu-241	8.3×10^{-4}	2.3	0.83	2.5	3.3×10^{-2}	8.3×10^{-2}	2.5×10^{-3}	2.5×10^{-3}
Am-241	5.4×10^{-4}	4.1	1.1	0.22	2.7×10^{-2}	–	–	–

[a] The concentrations of radionuclides have been obtained from 3-D hydrodynamic modelling. The concentrations in water are averages across the whole of Abrosimov Fjord, and the concentrations in sediment are averages within 100 m of the waste packages.

TABLE XXIV. ESTIMATED MAXIMUM DOSE RATES TO MARINE ORGANISMS FROM RADIONUCLIDES RELEASED FROM WASTES DUMPED IN ABROSIMOV FJORD

Organism	Alpha radiation, internal (μGy/h)	Beta and gamma radiation	
		Internal (μGy/h)	External (μGy/h)
Pelagic zooplankton	4.1×10^{-2}	8.0×10^{-3}	1.5×10^{-3}
Benthic crustaceans	4.1×10^{-2}	8.0×10^{-3}	1.7×10^{-2}
Benthic molluscs	1.1×10^{-1}	1.6×10^{-3}	1.6×10^{-2}
Pelagic fish	1.6×10^{-3}	3.8×10^{-2}	6.2×10^{-4}
Birds	3.7×10^{-3}	2.9×10^{-2}	7.1×10^{-3}
Seals	1.1×10^{-4}	4.7×10^{-2}	5.5×10^{-3}
Whales	1.1×10^{-4}	8.6×10^{-2}	7.0×10^{-5}

Note: The times to peak environmental concentrations are different for different radionuclides; simple summing of the estimated peak dose rates from each radionuclide will, therefore, result in an overestimate of the maximum radiation exposure.

energy transfer)) and β and γ radiation (low LET) listed separately. The highest dose rate is predicted for molluscs from internal α emitting sources. Because the radionuclide concentrations have been estimated on the basis of whole, soft tissue concentration factors, the α particle dose rate to particular tissues or organs will have been underestimated if these show preferential accumulation of plutonium or americium. The highest dose rates from internal β and γ emitters are likely to be experienced by fish, birds, seals and whales, and arise from the accumulation of ^{137}Cs. As the greater part of the dose derives from γ radiation, the problem of non-uniform ^{137}Cs distribution in tissues and organs is of lesser significance. The dose rate from external sources is expected to be greatest for the benthic crustaceans and molluscs and reflects the significance of the ^{60}Co contamination of the underlying sediment. Previous assessments of exposures in contaminated areas have assumed a quality factor of 20 for α radiation and presented total doses from all radiations in terms of the Sv unit, although it was recognized that this approach was open to question [104]. If comparisons, where possible, are made on this basis, the maximum total dose rates estimated for the releases to Abrosimov Fjord are within the range of expected natural background and of the same order as the dose rates predicted for the waste dumped in the deep north-east Atlantic Ocean [104].

It is also relevant to consider the dose rate to marine organisms that might colonize the exposed surfaces of the dumped packages. The maximum dose rate at the surface of the caisson containing the fuel from the icebreaker has been estimated to be 0.56 μGy/h (Section 6.3.4.4 below). From extensive previous experience with disposal to the north-east Atlantic dumpsite, the dose rates at the surface of packages of low level waste, limited on the basis of handling and transport requirements, can range up to more than 2 mGy/h (although fewer than 5% of the dose rates exceeded this value) [110]. It is reasonable to assume, in the absence of specific information, that considerations of limiting the worker exposure would also have limited surface dose rates for the packages dumped in the Kara Sea to similar values.

There have been a number of reviews of the information available concerning the effects of ionizing radiation on aquatic organisms [104, 106, 111] which have concluded that dose rates below 0.4 mGy/h to the maximally exposed members of populations of aquatic organisms are very unlikely to have any detrimental effects on attributes such as morbidity, mortality, fertility, fecundity and mutation rate that may influence the maintenance of health populations. The dose rates predicted in this assessment are very much smaller than 0.4 mGy/h (by at least two orders of magnitude, even allowing for the probably increased biological effect of high LET radiation) and can affect only a very small proportion of the populations. It may reasonably be concluded, therefore, that the dumping practice can have no detrimental impact on populations of aquatic organisms.

5.7. FINAL CONCLUSIONS AND DISCUSSION

The main findings of the modelling work are:

- The IASAP team made predictions of future sea water, sediment and biota concentrations covering the requirements for radiological assessment at local, regional and global scales with the aid of various models containing different spatial and temporal parameters.
- An extensive and detailed model intercomparison has shown the levels of variations among the different models, at near source locations, to be typically less than two orders of magnitude. The level of uncertainty in predicted concentrations indicated by this variation has been propagated through to final dose estimates.
- In all cases, the estimated dose rates resulting from the different models are very low.
- At regional and local scales, the maximum individual dose rates are generally very low and are of several orders of magnitude lower than doses due to ^{210}Po from the consumption of marine foodstuffs.
- The maximum individual dose rates for the hypothetical critical military group are higher than those for the other critical groups.
- At the global scale, the collective dose estimates in the years 2050 and 3000 from all sources in the Kara Sea are very low, and all results are in good agreement (collective dose estimates for ^{14}C and ^{129}I are dealt with in Section 7 below).
- The estimated maximum total dose rates to wild organisms from predicted releases in Abrosimov Fjord are within the range of expected doses from natural background.
- Dumping has no significant radiological impact on populations of aquatic organisms.
- The transport of contaminated sediment in ice from the solid waste dumped in the Kara Sea is of low global significance when compared with the transport in water.

6. POSSIBLE REMEDIAL ACTIONS

The second objective of the IASAP was to evaluate possible remedial actions related to the dumped wastes and to advise on whether they are necessary or justified. This section evaluates the nature, feasibility and costs of remedial options for reducing the radiological consequences associated with the dumping of radioactive wastes in Russian Arctic seas. For the solid radioactive wastes dumped in the Kara Sea, these remedial options fall into three categories, i.e. the introduction of additional in situ release barriers, relocation of dumped objects within the marine environment and recovery of objects for treatment and/or disposal on land.

6.1. INITIAL CONSIDERATIONS FOR REMEDIAL MEASURES

6.1.1. Radioactive wastes dumped in the Kara Sea

According to the basic information given in the 'White Book' [5], and reproduced in Section 1 (Tables I–III), the inventory of radioactive wastes can be broken down into three general categories: high level, intermediate level and low level. Nuclear reactors with spent fuel remaining in the reactor compartment and the icebreaker fuel container are regarded as high level waste. Reactor compartments without spent fuel can be regarded as intermediate level wastes, especially taking into account the radioactive decay since the time of dumping. The White Book provides insufficient data to separate the other dumped solid materials into separate categories; therefore they are designated as combined low and intermediate level wastes. It is expected, however, that most of the dumped barrels and containers are low level waste. In the evaluation of the feasibility of remedial measures, attention was focused on the solid high level waste objects, i.e. the reactors and the container containing spent fuel. The current total inventory of radionuclides in these objects has been evaluated to be 4.7 PBq [78]. Table XXV provides information on the nature and environmental setting of the various high level objects.

6.1.2. Implications of environmental conditions for remedial measures

The environmental conditions of the Kara Sea and its fjords are described in detail in Section 3. The specific features of the climate and sea bed conditions which may affect the technical realization of possible remedial measures are only briefly dealt with here.

If any remedial measure is deemed necessary, a preliminary survey of the dump sites including all important environmental features as described in Section 6.2.1 should be carried out before a final decision is taken.

Wind

The location of the dump sites east of Novaya Zemlya, and in the fjords especially, is favourable to remedial actions at prevailing westerly winds. In winter, the prevailing wind direction in the Kara Sea is south-westerly, turning to north-westerly in spring. Normally, the wind speed does not exceed 8 m/s, reaching up to 15 m/s under storm conditions. In summer, storms usually do not last longer than six hours. The wave heights are moderate. This means that the downtime of a remedial operation due to high waves is expected to be small.

Ice

On average, the east coast of Novaya Zemlya is ice free only in August and September. Ice formation starts in October, but the ice is relatively thin until December. This thin ice can be managed with reasonable effort by ice breaking equipment. The highest and thickest coat of ice, measuring up to 1.5 m, builds up in April/May. The most critical ice formations with respect to remedial activities will be the pressure ridges close to shore (shore ridges), which may be grounded in a water depth of 10 m. The ice in the fjords freezes and melts in place without significant movement.

Tides

The maximum tidal speed, which is one of the characteristic dynamic parameters of tidal currents, occurs along the east coast of Novaya Zemlya and is about 0.2 m/s. The probability of reaching this speed is once in 18.6 years. Along the shores of Novaya Zemlya Trough the tidal amplitude reaches 0.5–0.9 m, but it is reported that, depending on the phase of the tide, the tidal range in the fjords on the east side of Novaya Zemlya can be up to 1.8 m. In these circumstances, the currents in the fjords can be significant.

TABLE XXV. AVAILABLE AND ADDITIONALLY NEEDED INFORMATION ON RADIOACTIVE WASTES DUMPED IN THE ARCTIC

Dumped object (factory number)	Submarine No. 601	Reactor compartments Nos 285 + 901	Fuel from icebreaker OK-150	Reactor No. 421
Site	Stepovoy Fjord	Abrosimov Fjord	Tsivolka Fjord	Novaya Zemlya Trough
Year of dumping	1981	1965	1967	1972
Current activity (TBq)	614	1360	2200	293
Construction and protective barriers	Ref. [78]	Information not fully available, see Ref. [78]	Ref. [78]	Ref. [78]
Weight and size of containment	Can be approximated	Can be approximated	This report	Can be approximated
Material of containment	Ref. [78]	Ref. [78]	Ref. [78]	Ref. [78]
Condition of containment	Not known	Not known	Not known	Not known
Location	Located	Located[a]	Not located	Roughly located
Water depth	30 m	15–20 m	60 m	300 m
Environmental information:	Generally known	Generally known	Generally known	Generally known
– Water current	Locally not known	Locally not known	Locally not known	Locally not known
– Ice conditions				
– Sea conditions				
– Sea bottom conditions/soil				
– Underwater visibility				

[a] Reactor compartments probably located, but identification numbers not confirmed.

Bottom sediment

Near the dumping site in the Novaya Zemlya Trough, the bottom sediment is composed of pelite with silty material. The sedimentation rate is 0.3–0.4 mm/a. In the fjords, the bottom sediments consist mainly of fine silt or mud but in some fjords there are also areas of high bottom currents, where gravel lies on the sandy silt. In the absence of actual information on the sedimentation rates in the fjords, the data from fjords situated on the west coast of Novaya Zemlya Trough can be used to describe the fjords on the east coast where the actual dumping has occurred [20]. Mean sedimentation rates there have been evaluated to be 0.15–0.8 mm/a but in certain extreme situations they can attain 1.5–10 mm/a.

6.1.3. Possible remedial measures

The IASAP team of scientists defined three major categories of options for remedial measures for further evaluation:

(1) in situ containment options;
(2) options involving relocation of sources in the marine environment; and
(3) options involving recovery of the sources for land disposal, storage or treatment.

Category (1) includes the introduction of new barriers to the release of radionuclides from the sources and encapsulation of sources in subsea repositories. Two options in this category can be specifically considered: injection into the object of additional sealing material (e.g. Furfurol(F)) to reduce the rate of corrosion and release of radionuclides; and capping of the sources by covering them with encapsulating material having retentive or absorptive properties.

Category (2) encompasses all options for achieving reduction in exposures and risks by relocating the dumped sources to new underwater sites either by isolating the sources in an underwater cavern or by relocating the sources to sites in the deep ocean where the dose consequences would be significantly lower.

Category (3) includes all options that would require the dumped sources to be recovered from the sea and delivered to a port for subsequent disposal, storage and/or treatment of the radioactive components on land.

The feasibility of these various options depends, to a large extent, on the engineering integrity of the individual sources. For example, a highly corroded object should not be disturbed in order to avoid the risk of major structural failure, which would lead to enhanced release of radionuclides or possible recriticality of the reactor, as discussed in Section 4.2.5. However, an object that has maintained its structural integrity is amenable to a much wider range of options. The structural integrity of the dumped objects has been discussed in Section 4.4.3.

Knowledge of the structural integrity of the waste packages influences the selection of a case study for potential remediation. The best characterized source from an engineering perspective is the container of spent fuel from the icebreaker. This provides greater confidence in assessing the circumstances and costs of remedial action. In addition, this container is the object containing the largest current inventory of radionuclides (2200 TBq) among all the dumped high level wastes.

Accordingly, a case study of the technical feasibility and costs associated with remedial options for the spent fuel container of the icebreaker was carried out as an illustrative example. Initially, the following five options, representing the three option categories above, were considered:

(1a) Injection of material into a dumped container to reduce corrosion and to provide additional barriers to radionuclide release;
(1b) Capping of the container in situ using concrete or other suitable material to encapsulate the object;
(2a) Creation of an underwater cavern for the isolation of dumped sources on the coast of Novaya Zemlya;
(2b) Recovery and underwater transport for relocation to a deep ocean site; and
(3) Recovery of the container to a land environment (e.g. delivery to a port or harbour on Novaya Zemlya) in a structurally sound and suitably preserved form for subsequent disposal in a land repository for high level radioactive waste.

In a further evaluation of the options for the remedial actions listed above, salvage experts screened out options (1a), (2a) and (2b). The injection of additional material as referred to in option (1) was regarded as difficult to carry out, since the reactors are already at least partially filled with Furfurol(F) and the releases to the environment would only be postponed. The creation of an underwater cavern (option (2a)) in the coast of the fjord would be too expensive (at least US $100 million) for the isolation of only one waste container. Option (2b), recovery and underwater transport of the container to a deep ocean dump site, was screened out, basically for two reasons. First, it would be doubtful if the London Convention 1972 would grant special permission for an operation which can be regarded as involving the redumping of high level radioactive waste. Secondly, underwater transport on high seas involves high risks of losing the cargo.

Therefore, the preliminary evaluation of costs related to possible remedial measures was performed for only two options, i.e. recovery and in situ capping.

6.2. CASE STUDY OF REMEDIAL MEASURES FOR THE CONTAINER OF SPENT FUEL FROM THE ICEBREAKER

The pontoon containing the nuclear fuel in special containers was dumped on 18 September 1967 in the Tsivolka Fjord, at a position of 74°26′6″ N, 58°36′9″ E, at a depth of approximately 60 m [77]. Figure 34 depicts the fuel containers A, B and C, one within the other, and the pontoon. The dumping operation was described in some detail in Section 4.1.4.3.

The dimensions of the pontoon are as follows: length 14.5 m; diameter 4.5 m and height 5.3 m. The weight of the spent fuel on the pontoon, along with various elements in which it is encased, can be estimated as follows:

containers A and B	70	tonnes [77]
container C (steel)	1	tonne[3] and
concrete around C	80	tonnes[3]

6.2.1. Preparatory survey

The following steps were identified for assessing the feasibility and costs of the remedial actions. The first step would be to locate the spent fuel container. The Norwegian–Russian expedition visiting the Tsivolka Fjord in 1993 was unable to locate the pontoon which contains the fuel from the icebreaker [24]. It was, however, found that at the time of the expedition (i.e. August/September 1993) visibility in the Tsivolka Fjord was extremely low, 0.5 m.

It is recommended to split the preparatory survey of the dump site into two stages.

Mission 1 (locating the object)

The first mission should concentrate on just locating the object with a cursory inspection of the position of the pontoon and the fuel container and using a remotely operated vehicle (ROV) to carry out preliminary measurement of the currents and radiation in the vicinity.

The vessel for the first mission should be equipped with the following:

- Side scan sonar and an impulse radar to detect submerged objects;
- ROV (position and altitude of the pontoon; conditions);
- Current measurement equipment;
- Radiation measurement equipment.

Survey period:

Summer months (i.e. August/September). Before the expedition, the prevailing ice conditions should be checked and the short term ice development forecast. The ship used should be at least of the Russian ice class L1.

Survey duration:

15 days, including travel from Kirkenes/Murmansk to Tsivolka Fjord and back.

Mission 2 (main preparatory survey)

The second mission would concentrate on studies necessary to provide the information needed before a lifting or capping operation can be planned in detail. These studies should include:

(a) Inspection of the condition, position and corrosion of the pontoon and the fuel container, especially of the cap welds;
(b) Measurement of the radiation field and the sediment radioactivity close to the pontoon;
(c) Sea current measurements; and
(d) Studies on the nature of the seabed sediment, visibility, etc.

The survey vessel to be used for the main preparatory survey mission should be equipped with the following:

- Dynamic positioning equipment;
- Side scan sonar;
- ROV (position of pontoon, condition);
- Radiation measurement equipment;
- Water sampling equipment (surface and bottom);
- Sediment sampling equipment;
- Sediment core drill;
- Current measurement equipment;
- Accurate positioning equipment;
- Diving equipment and divers (60 m water depth);
- Accommodation for the inspection team (20 persons);
- High pressure cleaning equipment;
- Underwater weld checking equipment;
- Helicopter facilities.

[3] Expert estimation.

Survey period:

The exact time of the year for carrying out the second mission should be defined after determining which month offers the best conditions of visibility. Most probably this will be during the late winter season. If this preparatory survey is going to take place during the winter (ice) season, the survey vessel should be an icebreaker (Russian ice class UL).

Survey duration:

30 days, including travel from Kirkenes/Murmansk to Tsivolka Fjord and back.

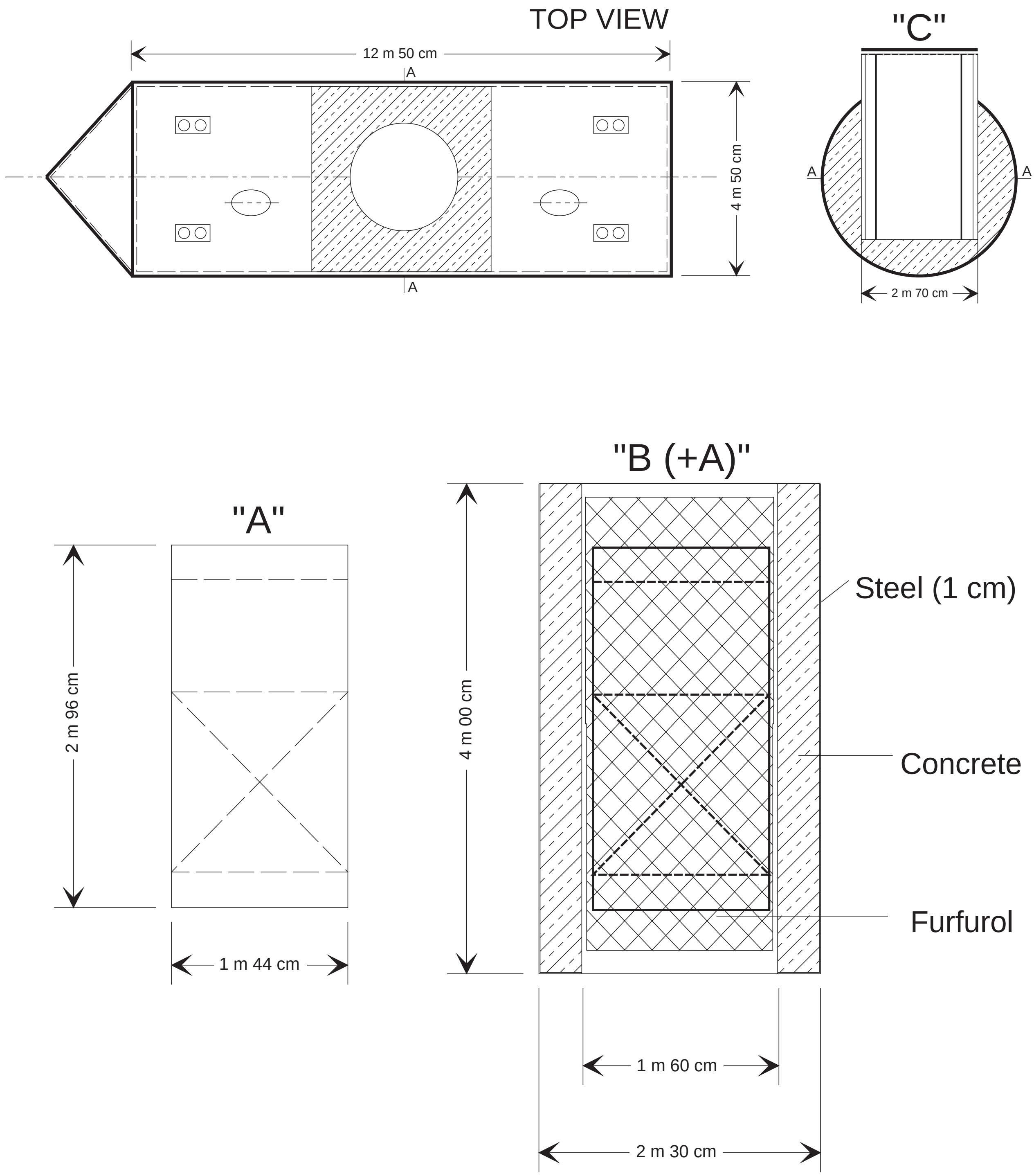


FIG. 34. Icebreaker's fuel containers A, B, and C and the pontoon [77].

The following information should be obtained during the survey:

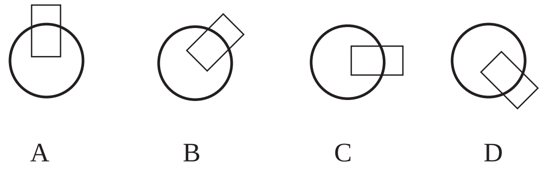
(a) Radiation dose rate and radionuclide contamination close to the pontoon;
(b) exact position and condition of pontoon, i.e. trim, list, sea growth, corrosion, penetration in seabed and integrity;
(c) Condition of welds and supports of the lid of the fuel containment;
(d) Condition of concrete in the fuel pontoon;
(e) Data on water (tidal) currents at several water depths (every 10 m);
(f) Type of sediment, sediment profile until hard bottom + additional 10 cm;
(g) Survey of seabed around the pontoon (radius: 1000 m) for location of other dumped objects;
(h) Underwater working conditions (visibility, etc.).

6.2.2. Engineering evaluation of remedial actions

6.2.2.1. Recovery

The precondition for the lifting of the fuel container is that a temporary or permanent location for its storage or disposal be identified. The recovery operation described here will include the actual lifting of the pontoon onto a cargo barge and securing it for sea transport to a nearby coastal location. If this location is on the coast of Novaya Zemlya or a nearby port, such as Murmansk, the costs of transport are included within the estimates given. Lifting and sea transport should be effected within the same season and under weather and sea conditions ensuring the highest standards of safety. Under these conditions, the additional risk of an accident during transport from the Novaya Zemlya coast to a nearby port should be of little significance.

For lifting, four different positions of the fuel pontoon have been considered, i.e. A, upright position; B, 45° list; C, 90° list and D, 135–180° list (with the lid turned downward).



The vessels required for the lifting operation are a diving support vessel, a crane barge and a cargo barge, all at least of the (Russian) ice class L1 and equipped as follows:

- Diving support vessel:
  - Dynamic positioning equipment;
  - Accurate positioning equipment;
  - Diving equipment (for a depth of 60 m);
  - Helicopter facilities;
  - Crane (capacity 10 tonnes);
  - Accommodation (minimum 30 persons);
  - Deckspace (minimum 150 m^2);
  - Satellite communication;
  - Machine shop.
- Crane barge:
  - Minimum capacity 500 tonnes;
  - Dynamic positioning equipment;
  - Seaworthy.
- Cargo barge:
  - Minimum capacity 500 tonnes;
  - Flattop type;
  - Seaworthy.

The actual lifting procedure would be as follows:

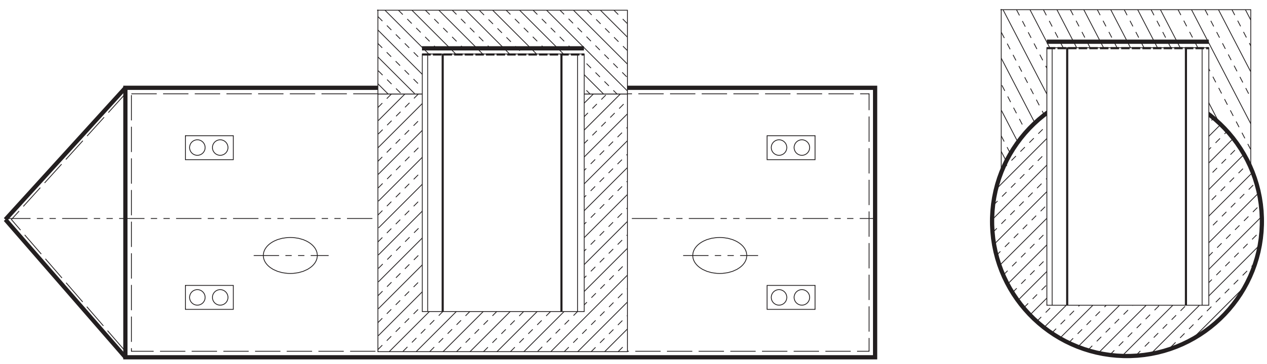


FIG. 35. Schematic depiction of the icebreaker's fuel container and pontoon when prepared for lifting ([77] and this report).

1. Securing the lid of the fuel compartment:
   - Position A:
     Installation of a circular steel casing (diameter: 3.3 m; height: 1.5 m) around the lid (see Fig. 35);
     Filling the inner space with concrete (approximately 6 m^3~12 tonnes).
   - Position B:
     The same procedure can be used as for position A.
   - Position C:
     Clearing the area surrounding the lid of sediment;
     Thereafter, the same procedure can be followed as for position A.
   - Position D:

   (1) When **no** elevated levels of radionuclides in sediment are found:
     – removing the sediment;
     – turning the pontoon to position A;
     – thereafter, the same procedure can be followed as for position A.

   (2) When **slightly** elevated levels of radionuclides in sediment are determined:
     – removing the sediment;
     – strengthening the weld of the lid in situ;
     – proceeding as described under D(1) above.

   (3) When **highly** elevated levels of radionuclides in sediment are found:
     – capping the fuel container and/or the pontoon in situ (see Section 6.2.2.2).
2. Installation of lifting plates around the pontoon in a proper position (the width of the lifting plates should be approximately 0.6 m). For installation, a crane barge and/or water jet and suction pumps can be used. If the pontoon is in a sound condition one may consider dewatering it by means of pressurized air. However, it is difficult to control this operation during refloating.
3. Raising the pontoon close to the water surface.
4. Preparing the support cradle on the cargo barge to be used for transport.
5. Moving the crane barge and lifting the pontoon (just submerged) to sheltered waters, if necessary.
6. Dewatering the pontoon (i.e. removing some 150 tonnes of water) by means of pumps. The pontoon is equipped with 8 tanks, divided between fore and aft (Fig. 36).
7. Lifting the fuel pontoon out of the water and onto the cargo barge.
8. Securing the fuel pontoon on board the cargo barge (fastening for sea journey).

6.2.2.2. In situ capping

By capping the fuel container it is possible to postpone the releases of radionuclides to the environment up to hundreds of years, thus allowing sufficient time for the decay of the most important fission product nuclides, ^{90}Sr and ^{137}Cs. However, in respect of the overall releases of long lived actinides, such as plutonium isotopes, the capping will only have a negligible effect.

Capping of the fuel container (general option)

The precondition for this option is that the structure of the fuel pontoon be intact. The option can be applied to fuel container and pontoon positions A, B, D(1) and D(2) as previously described in Section 6.2.2.1. The only vessel required for the operation is a diving support vessel of at least the (Russian) ice class L1, be equipped with the following:

- Dynamic positioning equipment;

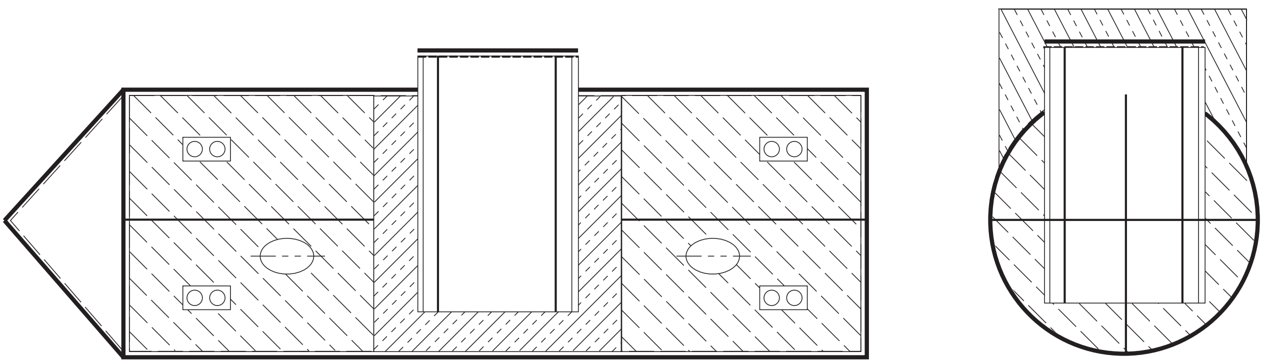


FIG. 36. Location of the water tanks in the pontoon ([77] and this report).

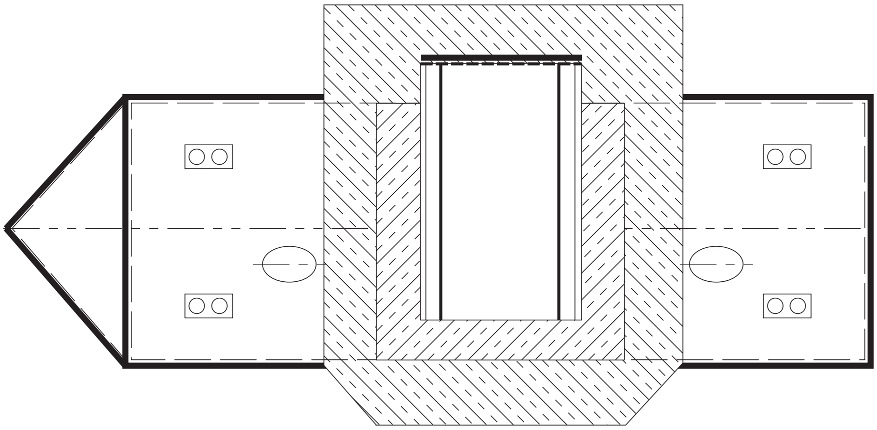


FIG. 37. Schematic depiction of cutting the pontoon and capping the icebreaker's fuel container with concrete ([77] and this report).

- Accurate positioning equipment;
- Diving equipment (for a depth of 60 m);
- Helicopter facilities;
- Crane (capacity 10 tonnes);
- Accommodation (minimum 30 persons);
- Deckspace (minimum 150 m^2);
- Satellite communications;
- Machine shop.

Procedure for capping (Fig. 37):

1. Excavation of sediment underneath the fuel pontoon to a depth of approximately 1 m below the bottom layer of the pontoon or until a solid level of sand is reached.
2. Checking the welds of the fuel container lid and rewelding if necessary.
3. Filling the excavated area with concrete.
4. Cutting off the fore and aft part of the fuel pontoon.
5. Moving the cut fore and aft parts well out of the area.
6. Installation of the casing around the fuel compartment.
7. Filling the enclosure area with concrete up to at least 50 cm above the fuel compartment.

Capping of the fuel container (special option)

The special option is applicable if the pontoon is upside down and the sediment around the container is significantly contaminated because of leakage from the containment lid. In this case, the use of atmospheric pressure diving suits is recommended for the diving team. This allows the divers to work at normal atmospheric pressure even at a depth of 60 m. After the divers have emerged, the diving suits can be decontaminated while the divers rest in normal surroundings on board the ship.

For this option, three atmospheric pressure diving suits are required, but no decompression compartment. Therefore no diving support vessel is needed and the operation can be undertaken with a normal ship equipped as above.

Since the use of atmospheric pressure diving suits allows only simple movements, activities on the sea floor at the dumpsite should not be complicated. It is therefore recommended to lift the fuel compartment together with both ends of the pontoon from the sea bottom — most likely out of the mud layer. Once raised, both ends of the pontoon are then cut off and the fuel container is lifted into a prefabricated steel box for capping with concrete.

The procedure for capping the leaking fuel compartment would be as follows:

1. Installation of the prefabricated steel box with the cradle on the seabed near the fuel pontoon.
2. Installation of the lifting plates underneath the central part (containment) of the fuel pontoon.
3. Inspection for leakage and plugging of any hole by welding.
4. Lifting the fuel pontoon from the seabed and cutting off both ends in the raised position.
5. Positioning the fuel container in the steel box.
6. Filling the box with concrete.

Capping the entire fuel pontoon

This option can be used if the fuel pontoon is corroded and does not provide adequate support for the fuel containment. The position of the fuel pontoon can be

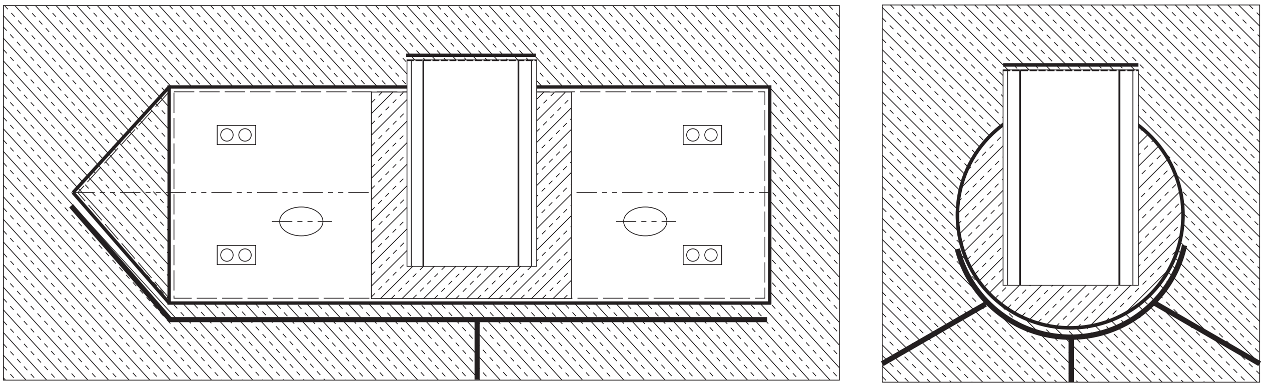


FIG. 38. Schematic depiction of providing support and capping the entire pontoon with concrete ([77] and this report).

A, B, C, D(1) or D(2), as previously described in Section 6.2.2.1. The vessels required for the operation are a diving support vessel and a crane barge. The diving support vessel should be equipped with the following:

- Dynamic positioning equipment;
- Accurate positioning equipment;
- Diving equipment (for a depth of 60 m);
- Helicopter facilities;
- Crane (capacity 10 tonnes);
- Accommodation (minimum 30 persons);
- Deckspace (minimum 150 m^2);
- Satellite communication;
- Machine shop.

The requirements for the crane barge are as follows:

- Minimum capacity 500 tonnes;
- Dynamic positioning equipped;
- Seaworthy.

Both vessels should be at least the of (Russian) ice class L1.

Procedure for capping (Fig. 38):

1. Installation of the prefabricated steel box with the cradle on the seabed close to the fuel pontoon.
2. Installation of the lifting plates underneath the central part (container) of the fuel pontoon.
3. Inspecting lid welds of the fuel container and rewelding if necessary.
4. Lifting the fuel pontoon from the seabed into the box.
5. Filling the box with concrete.

TABLE XXVI. TIME REQUIRED FOR THE CAPPING OF THE FUEL PONTOON

Diving support vessel	Number of days
Installing the steel box	1
Removal of the sediment and installation of the lifting plates	4
Cleaning of the area, inspection of the weld and rewelding if required	2
Lifting the barge into the box	1
Filling the box and fuel pontoon compartments with concrete	3
Extra margin for work	2
Working time	13
25% downtime due to the weather	3
Travel from Kirkenes/Murmansk to Tsivolka Fjord and back	7
Total	23 days
Crane barge	**Number of days**
Working time	9
25% downtime due to the weather	2
Travel from Kirkenes/Murmansk to Tsivolka Fjord and back	7
Total	18 days

6.2.3. Cost estimates

Cost estimates for the preparatory survey, lifting and capping were provided by engineering experts, including one from a major western salvage company. As an

TABLE XXVII. CRITERIA CONSIDERED IN CALCULATING THE COSTS OF FUEL PONTOON CAPPING (US $)

		Daily rate (US $)	Days	Subtotal (US $)
Diving support vessel to be equipped with:	Dynamic positioning equipment Accurate positioning equipment Diving equipment (depth of 60 m) Helicopter facilities Crane (capacity of 10 tonnes) Accommodation (30 persons) Deckspace (minimum 150 m^2) Satellite communication Machine shop Mob & demob Kirkenes/Murmansk[a]			
Remotely operated vehicle	Equipment & personnel (2+1 technician) Mob & demob Kirkenes/Murmansk[a]			
Technical experts	Technical experts & supervisor (5 persons) Mob & demob Kirkenes/Murmansk[a]			
Diving team	Equipment & personnel (8 persons) Mob & demob Kirkenes/Murmansk[a]			
Radiation protection team	Equipment & personnel (2 persons) Mob & demob Kirkenes/Murmansk[a]			
High pressure cleaning	Equipment[b] Mob & demob Kirkenes/Murmansk[a]			
Underwater welding & cutting equipment	Equipment & personnel (2 persons) Mob & demob Kirkenes/Murmansk[a]			
Suction pumps & hoses (to remove sand/mud)	Equipment[b] Mob & demob Kirkenes/Murmansk[a]			
Air jet	Equipment[b] Mob & demob Kirkenes/Murmansk[a]			
Hydraulic drilling mole (for slings)	Equipment (personnel needed?) Mob & demob Kirkenes/Murmansk[a]			
Grouting equipment (∅3.3 m; 10 mm thick)	Equipment & personnel (2 persons) Mob & demob Kirkenes/Murmansk[a]			
Lifting equipment (slings for underneath)	Construction & transport	–	–	
Crane barge	Equipment Mob & demob Tsivolka Fjord			
Supply vessel	For transport of steel box (including cradles, approximately 16 m × 6 m × 8 m) Mob & demob Tsivolka Fjord			
Steel box and cradle (including reinforced steel for concrete)	Construction	–	–	
Concrete (? m^3)	Material	–	–	
	Total			

[a] From the home port of the vessel to Kirkenes/Murmansk and back.

[b] Can be carried out by the diving team.

TABLE XXVIII. SUMMARY OF COST ESTIMATES

Operation		Range (million US $)
Preparatory survey	Mission 1	0.6–0.8
	Mission 2	1.6–3.4
Actual remediation operation:		
Lifting the fuel container	(positions A, B and C)	2.6–5.9
Lifting the fuel container	(position D)	4.3–8.5
Capping the fuel container	(general case)	2.5–3.6
Capping the fuel container	(special case)	2.5[a]
Capping the entire pontoon		3.6–5.9
Total		5–13
Preparatory survey		2.2–4.2
Actual operation		2.5–8.5

[a] Only one estimate.

example, the criteria for estimating the costs of capping the fuel pontoon are shown in Tables XXVI and XXVII. Table XXVIII contains a summary of the estimates provided for the entire project. These cost estimates vary widely (US $5–13 million). The variation results mainly from the different prices quoted for the diving support vessel; lower prices apply if not very comfortable ships are used; higher prices will be charged if the very sophisticated ships available in some companies are selected. In general, the cost estimates are very approximate. More precise cost predictions on the lifting or capping remediation can only be given after the preparatory expedition.

6.2.4. Occupational exposures during remediation

A preliminary evaluation of potential occupational exposure during the remediation was carried out by calculating the external dose rates on the surface of container C. The calculations are valid for a successful operation.

A computer model of the special container housing the damaged spent nuclear fuel and core barrel from the icebreaker's N2 reactor was developed for the purpose of calculating external dose rates. The computer code used for the calculations was the Monte Carlo code, MCNP [112], developed at Los Alamos National Laboratory and in widespread use for neutron, photon and electron transport. The photon cross-section data used with MCNP are from the ENDF/B-VI data library. The following sections explain the model, the parameters used, the assumptions and the results.

6.2.4.1. Geometry

The geometry of the container was based on several schematic illustrations contained in the report of Sivintsev [77], in particular Figs 8 and 9. The model consists of configuration A, container B, the outer caisson (container C), and the pontoon partially filled with concrete.

Configuration A is the inner part of the reactor vessel including the damaged fuel and screening assembly. The reactor is of a height of 1600 mm and a diameter of 1082 mm. However, the active fuel region was taken to be 955 mm on the basis of the drawing. Surrounding the reactor were five thermal shields of varying thicknesses, a 250 mm thick plate at the top and a 185 mm thick plate at the bottom. The entire assembly was of a height of 2965 mm and a diameter of 1440 mm.

Container B was a steel and concrete container that had been used to store configuration A for one and a half years before dumping. The container consisted of a 50 mm primary steel liner, a 30 mm secondary steel liner, a 5 mm steel outer wall, and a 515–545 mm thick annular region of concrete between the liners and the outer wall. The secondary liner was added for extra shielding just at the height of the reactor core and therefore did not extend to the entire height of the container.

Container B was placed in a specially prepared steel caisson, container C, for transport and disposal. The caisson consisted of a 12 mm steel wall and an 18 mm steel lid. The gap between container B and the caisson was filled with Furfurol(F), as was the space between configuration A and container B. The inner spaces of the reactor were also filled with Furfurol(F).

The pontoon used to transport the caisson out to sea, which was dumped along with the container C, was modelled as a steel cylinder of a length of 12.5 m and a diameter of 4.5 m. The cylinder also had a 2 m long cone on the front for towing. The walls of the pontoon were steel plates of a thickness of 16 mm.

6.2.4.2. Materials

For modelling purposes five materials — steel, concrete, UO_2, Furfurol(F), and water — were used. The carbon steel and stainless steel were modelled as iron, with a density of 7.86 g/cm^3, and the concrete was modelled as SiO_2 with a density of 2.36 g/cm^3. The formula for Furfurol(F) was taken to be $C_5H_4O_2$ with a density of 1.02 g/cm^3. Water with a density of 1.0 g/cm^3 was used to simulate the ocean water surrounding the sunken container.

The material in the 125 spent nuclear fuel assemblies was assumed to be 57% of the original 75 kg ^{235}U loading which was calculated to be 971.5 kg of UO_2 based on 5% ^{235}U enrichment and 10 g/cm^3 UO_2 fuel pellet density. The total mass of UO_2 was homogenized over the volume of the reactor core which resulted in an overall density of 1.1064 g/cm^3 in the source region. Even if the actual active volume is smaller, the mass of fuel is conserved, and the effect on the dose rate through the steel and concrete at the distances of concern should not be affected. The effect of the additional mass due to the Zr–Nb cladding was ignored, which will result in conservative estimates of dose rates.

6.2.4.3. Source term

The main source of photons from the icebreaker fuel, based on the inventory calculations of the source term group, appeared to be the fission product ^{137}Cs and the activation product ^{60}Co. The computer code requires the energy of the photons to be input; therefore, 0.662, 1.173 and 1.332 MeV, representing 94.88%, 2.56% and 2.56% of the photons considered, respectively, were used as the source term. The rest of the inventory consisted mostly of beta emitters and actinides such as ^{241}Am with low energy photons. The Monte Carlo calculation was based on a volume isotropic distribution and isotropic direction of the source photons.

The surface averaged fluence tally in MCNP was used rather than the point detector tally, which is not reliable for highly scattered systems. The fluence over the outside surface of the caisson was calculated just for the height of the cylinder in line with the source region. It was necessary to use such a large area to decrease the statistical error of the calculation. The energy dependent photon fluences were multiplied by dose conversion factors taken from ICRU Report No. 47 [113]. The tables in ICRU Report No. 47 contained fluence-to-kerma factors and kerma-to-personal dose equivalent ($H_p(10)$) factors at various energies and the product of these two factors was used in the calculation. The conversion factors are at discrete photon energies; however, MCNP uses log–log interpolation to convert all contributing energies of photons to dose equivalent. The final value, which is normalized to a single source photon, was multiplied by 1.55×10^{14} photons/h to provide the dose equivalent rate in μSv/h.

6.2.4.4. Results

The highest dose equivalent rate, 0.56 ± 0.06 μSv/h, occurs at the side of the caisson at the height of the source region. Above and below this region, the dose rate diminishes because of the greater distance from the source and the increased effective shielding thickness. In the region above the fuel beneath the top of the reactor core, the dose rate is 0.08 ± 0.01 μSv/h. The errors reported with these dose rates are only the statistical errors of the Monte Carlo calculation and do not include the uncertainties in the input parameters. The calculations were performed by using a range of material densities and source radionuclides. On the basis of the results of these calculations, it can be estimated that the total uncertainty is approximately 50%. The ‘worst case’ dose rate is below any level requiring personnel monitoring, and the actual dose rate is likely to be much less. Moreover, this dose rate is low enough that additional shielding would not be necessary to retrieve the container and bring it aboard a cargo barge.

6.2.5. Consideration of accidents

Before a decision on any remedial action is made, careful consideration should be given to the assessment of potential accidents. Some aspects are highlighted in this section. During the preliminary survey, the structural integrity of the source should be studied as well as the possible leakages of radioactive substances; the actual remedial action should be planned accordingly. It should be noted that the radiation risk to workers estimated above is valid only for successful operations. If the operation should fail, there is always a potential for destruction of some of the existing barriers and a subsequent leakage of radioactive material and weakening of containment structures. If that were to happen, resorting to divers using decompression techniques is no longer possible for underwater operations because of decontamination problems in the decompression compartment of the diving support vessel. For simple operations, however, divers using atmospheric pressure diving suits can still be called on.

With regard to possible criticality, it has been evaluated [78] that since the icebreaker fuel container contains only 20.6 kg ^{235}U in a mass of about 1000 kg UO_2 and about 30 kg as a compact mass has been regarded as needed for criticality, possible criticality of the spent fuel in the containment can be ruled out.

6.3. REMEDIATION APPLIED TO OTHER HIGH LEVEL WASTES

The analysis of feasible remedial measures for the container of spent fuel from the icebreaker has been provided as an illustrative case study. Given comparable degrees of information regarding their engineering features, similar analyses of remedial measures for other objects containing radioactive wastes dumped in the region can be conducted. However, current limitations on the information on the design and construction of these other objects would reduce the confidence with which firm conclusions can be drawn from such analyses. Nevertheless, the case study of the icebreaker spent fuel package focuses on the item containing the largest radionuclide inventory (47% of the total) situated at a nearshore location. In the case of in situ remediation, the costs of remediation for the other high level waste objects would be similar to or greater than those for the container of spent fuel from the icebreaker. This is because the other high level waste containers are larger and therefore would necessitate a broader effort and greater costs associated with in situ remediation such as capping with concrete. Only in the case of recovery of dumped wastes for land treatment and/or disposal (i.e. recovery and delivery to a coastal port) might the costs of recovery of other packages be lower than those for the icebreaker fuel package. However, as stated earlier, the downstream costs of land treatment and disposal are not included in the cost estimates of the case study. Furthermore, these land treatment costs would be similar for the disposal of all high level wastes recovered and are likely to represent the greater proportion of the overall costs of remedial actions for any of the waste packages.

6.4. CONCLUSIONS

Various options for remedial actions have been evaluated, in particular with respect to technical feasibility and associated costs. This study was carried out for the spent fuel of the icebreaker reactor as an illustrative example. The results show two options to be technically feasible, i.e. in situ capping with concrete and recovery. The latter option requires a temporary or permanent location on land for storage or disposal of the fuel containment.

If remedial actions are to be carried out, more detailed information on the exact location, situation and structural integrity of the icebreaker's spent fuel, as well as the environmental conditions, would have to be obtained through preparatory surveys (missions 1 and 2).

The estimated costs for the two remedial options discussed, including the two preparatory surveys, are in the range of US $5–13 million. The predominant reason for this range of costs pertains to the type and origin of the vessels used.

The remedial options considered for the container of spent fuel from the icebreaker, which constitutes the largest single inventory of solid radioactive waste dumped in the Kara Sea, would not result in significant radiation exposures to the personnel involved in the operations, assuming that the engineering integrity of the container and its contents are maintained.

Apart from any decision for or against remedial actions, it is recommended to carry out a site survey in order to locate all high level wastes and to obtain preliminary information on their position and conditions.

The case study of the spent fuel container from the icebreaker provides cost estimates for remediation. These can be compared and considered for the specific avertable dose for this remedial action. Then this can be expanded to consider the total dose which might be averted from remediation of all the dumped objects. These costs could then be used to evaluate the justification for any remedial action, on purely radiological grounds.

If, for other than radiological reasons, remediation should further be considered for any of the high level waste packages, attention should be devoted to a prior estimation of the dose to those persons involved in remedial actions and the probability and consequences of accidents, especially for reactors containing spent fuel. In effect, these reactors pose a higher risk of a criticality accident as a result of engineering failures.

7. ANALYSIS OF THE NEED FOR REMEDIAL ACTIONS

This section reviews the results of the radiological impact calculations (Section 5) and draws conclusions about the need for remediation from a purely radiological protection viewpoint, taking into account the technical aspects of remediation measures (Section 6).

Major inputs into such a decision are the predicted health risks to individuals and populations from the dumped materials. The liquid wastes discharged directly into the waters are considered separately from the dumped solid wastes.

7.1. LIQUID WASTES

Low level liquid wastes were dumped directly into the Barents and Kara Seas. The amount of liquid waste dumped (representing 0.88 PBq) is a small proportion (~5%) of that entering the Arctic Ocean from other sources such as global weapons testing fallout and liquid discharges from the Sellafield nuclear fuel reprocessing plant. Furthermore, the rapid flushing of the Barents and Kara Seas ensures that activity concentrations do not build up (Section 3) and the vast majority of the radionuclides disperse away.

The resultant activity concentrations from the liquid disposals are very small compared with the contributions from other artificial sources and are very much lower that the normal activity concentrations of naturally occurring radionuclides. This has been verified by environmental measurements. Measurements on environmental materials suggest that current annual doses from all artificial radionuclides in the Kara and Barents Seas are, at most, only 1 to 20 μSv (Joint Russian–Norwegian Expert Group, 1996 [6]). Such dose levels do not indicate the need for remediation (see Section 7.2.2.1). Furthermore, the only possible remedial actions are intervention in the exposure pathways, such as restricting consumption of seafood, and any reduction in radiation dose is unlikely to outweigh the disadvantages of the intervention measures from the individual’s point of view.

7.2. SOLID WASTES

7.2.1. Packaged low level wastes

Altogether, more than 6000 containers of solid low level radioactive wastes have been disposed of at eight dump sites in the years 1964–1991 (Table I). They were either dumped individually or on barges. The wastes comprised mainly film coverings, tools, personal protective devices, filters and other contaminated objects produced during maintenance work. Usually they are enclosed in metal containers, mainly made of ferritic–perlitic steel or carbon steel. It has been evaluated that the lifetime of such a container, with a wall thickness of 3–4 mm, in cold sea water with high dissolved oxygen content is from 5 to 15 years [114]. Thus, a significant fraction of the radionuclides in the low level waste containers may have already dispersed in the sea water. Indeed, analyses of some sediment samples taken close to dumped containers show elevated radionuclide concentrations (Section 3).

The total radionuclide content of low level waste containers (<0.6 PBq, at the time of dumping) can be compared with the amount of high level wastes dumped at the same sites (37 PBq). It can also be compared with a similar type of low level wastes dumped at the north-east Atlantic dump sites under the London (Dumping) Convention provisions (42.3 PBq).

Therefore, for the following reasons, it can be concluded that no remedial actions are required for the packaged low level wastes. First, on the basis of the assessment for high level waste described below, it is evident that the releases of radionuclides from low level wastes with much smaller inventories would not pose a significant hazard to human health or to the environment. Secondly, owing to the great number of containers scattered on the sea floor and their questionable condition, any remedial action would be difficult if not impossible.

7.2.2. High level wastes

These wastes comprise reactors complete with fuel, together with one consignment of spent reactor fuel from a nuclear powered icebreaker.

Radionuclides will be released to the environment from these wastes over a period of many years, and thus modelling studies must be undertaken to assess possible doses and risks to individuals and populations in the future. These studies are described in Section 5. The most important aspects of radiological impact for the purposes of deciding on the need for remedial actions are as follows (see also Section 2):

(1) the doses and risks to the most exposed individuals if no action is taken and whether their situation is improved by taking action; and

(2) the total health impact on exposed populations and how much of this can be avoided by taking remedial actions.

These two aspects of radiological impact are considered separately.

7.2.2.1. Individual dose

Doses were calculated for individuals in four population groups bordering on the Kara and Barents Seas, and in a group of military personnel on Novaya Zemlya. Three of the four population groups represent critical groups of subsistence communities of seafood consumers. These groups are located on the Yamal peninsula, the Taymyr peninsula and the estuary of the Ob and Yenisey rivers. The fourth population group represents average seafood consumers in the nearest major population centre on the Kola peninsula. There are no indigenous population groups on Novaya Zemlya itself; however, doses were calculated for military personnel patrolling the banks of the fjords containing dumped radioactive wastes.

Different models and modelling approaches were used in calculating future doses (Section 5), and a range of results was obtained. The range reflects, inter alia, the spatial resolution of the models in the areas of interest.

A first stage in deciding on the need for remediation is to evaluate the radiological consequence on members of the public of the normal gradual releases of radionuclides from the solid wastes. These releases occur owing to the gradual corrosion of the waste form and are represented in the study as Scenario A (Section 4). The calculated doses from the releases to the individuals in four population groups bordering on the Kara and Barents Seas are summarized in Table XXIX. The range of results reflects the results of the individual models used in the assessment. Doses are presented for each source area, e.g. Stepovoy Fjord or Abrasimov Fjord. The summed total for each critical group is a summation of the highest calculated dose to that critical group from each source. In summing doses, the fact that peak doses may occur in different years has been ignored; therefore, the totals are conservative in this respect. However, it is interesting to note that the critical group doses calculated by models which were run for all sources simultaneously (Table XXIX) are within an order of magnitude of the doses summed source by source.

Annual individual doses in each of the four population groups are all less than 0.1 µSv. This value is significantly lower than the normal variations in background radiation from place to place that are experienced in western countries – these can be 300 µSv/a excluding the much greater variations in doses from ^{222}Rn. Furthermore, annual doses from natural ^{210}Po via consumption of fish caught in the Barents Sea are of the order of 400 µSv.

No subsistence communities live on the island of Novaya Zemlya itself. The island has been used for nuclear weapons testing and is under military control. Thus, dose calculations have been undertaken for a hypothetical critical group of military personnel patrolling beaches on the fringes of the fjords containing radioactive wastes (Section 5). These calculated doses are generally higher than those calculated for the population groups bordering the Kara and Barents Seas; the highest annual dose of about 570 µSv is calculated for military personnel patrolling Tsivolka Fjord while releases are occurring from the icebreaker fuel (Table XXX). The annual calculated doses for the other locations are about one order of magnitude, or more, lower. The calculations assume that military personnel patrol the beaches directly bordering each fjord for 100 hours — given the harsh environmental conditions, this assumption is likely to be conservative.

Calculations were also undertaken for two possible future situations where accelerated releases can occur: rupture of the box containing the icebreaker fuel in 2050 followed by accelerated release of radionuclides (Scenario B); and widespread glaciation of Novaya Zemlya followed by glacial scouring of the fjords, crushing of the waste forms and eventual simultaneous release of radionuclides in the Kara Sea in the year 3000 (Scenario C). Scenario B is discussed in Section 7.3 in connection with the specific example of possible remedial actions on the icebreaker fuel.

Considering the subsistence communities, the highest doses from Scenario C were to the critical groups of seafood consumers on the Yamal and Taymyr peninsulas (Table XXXI). However, the annual doses to these groups were less than 1 µSv and thus can be regarded as having no significance. The highest doses to the military personnel in Scenario C are around 3 mSv/a.

7.2.2.2. Collective dose

A second aspect of radiological impact relevant to an evaluation of the necessity for remedial actions is the collective dose. This quantity is the sum of the doses from the source in question to all individuals in the exposed population. The collective dose to a given population from a given source represents the radiological consequences to this population from this source. However, long lived radionuclides, in particular ^{14}C in the context of this study, may cause exposures over many hundreds or thousands of years into the future. There are, however, increasing uncertainties in calculating doses over such time periods. Therefore, in this study, for the reason given in Section 7.4, collective doses were evaluated up to the year 3000. This time period covers the times of the peak release rates for the normal, gradual corrosion release scenario (Scenario A). Furthermore, collective doses delivered after the year 3000 are likely to be dominated by the contribution from the long lived, mobile radionuclide ^{14}C. As will be discussed in Section 7.4, this radionuclide requires special consideration when evaluating the significance of projected collective doses for a decision on remediation.

Collective doses have been evaluated for the best estimate Scenario (A). Doses arising from the dispersion of radionuclides in the world's oceans were estimated using a number of models (see Section 5). The collective doses were evaluated for two time periods, up to the year 2050 and up to the year 3000. The calculated collective doses are given in Tables XIX and XX. For each time period, the collective doses estimated by the various models are similar and small, with all values being below 1 man·Sv. The waste inventory also contains two radionuclides, ^{14}C and ^{129}I, that are long lived and mobile. These two radionuclides circulate globally in the aquatic, atmospheric and terrestrial environments. Appropriate global circulation models have been used to calculate collective doses for these two radionuclides [115]. Assuming that the entire ^{14}C inventory in the wastes is released around the year 2000 and integrating doses to the world's population (10^{10} individuals) up to the year 3000 yields a collective dose of about 8 man·Sv. The corresponding value for ^{129}I is much lower, at 0.0001 man·Sv.

Thus, the total collective dose to the world's population up to the year 3000 from the dumped radioactive waste is approximately 10 man·Sv. In comparison, the annual collective dose to the world's population from natural radionuclides in the ocean is about three orders of magnitude higher [91].

For comparative purposes, the collective dose from ^{14}C over the next 10 000 years is about 37 man·Sv and over all time is about 50 man·Sv. The corresponding values for ^{129}I are 0.002 man·Sv and 0.2 man·Sv. This leads to a total collective dose for all time of about 60 man·Sv when the contributions from other radionuclides are included. This estimate assumes current environmental conditions and population habits.

The greater part of the collective dose arises from global circulation of ^{14}C, and the dose from this radionuclide will be spread throughout the world's population, thus leading to a vanishingly small level of individual risk. Furthermore, the calculated collective dose is very small in comparison with the annual dose from natural background to the world's population of approximately 24 million man·Sv (assuming the average 2.4 m·Sv/a for a world population of 10^{10} people); about 120 000 man·Sv of which arises from natural ^{14}C produced in the upper atmosphere.

It is informative to compare the collective doses estimated for the dumping in the Kara Sea with the collective doses reported in other assessments for radionuclides in marine waters. The European Commission's project Marina [94] estimated the radiological impact of radionuclides in northern European marine waters. Calculations were undertaken of the collective doses, arising from various sources, to the population of European Communities up to the year 2500. The calculations give values of 4600 man·Sv for liquid discharges up to the mid 1980s from the Sellafield nuclear fuel reprocessing plant, and 1600 man·Sv for weapons test fallout (excluding the contribution from ^{14}C which was not reported and could be up to an additional several hundred man·Sv). The collective dose to the European Community population up to 2500 from past dumpings of solid radioactive waste in the north-east Atlantic is estimated to be 50 man·Sv, of which the majority is due to ^{14}C. The collective dose to the world's population from this source is reported to be 3000 man·Sv up to approximately the year 3000. Thus, it is clear that the overall radiological impact of waste dumped in the Kara Seas is very low compared with other sources including controlled releases of radionuclides.

7.3. ICEBREAKER FUEL

The dumped wastes are distributed in several areas each of which would require a separate examination of the technical feasibility of retrieval or some other form of remediation. As an example, the technical feasibility of taking remedial action on the icebreaker spent fuel container at Tsivolka Fjord has been evaluated in Section 6. Two possible remedial actions were considered, i.e. retrieval of the waste and provision of various types of additional barriers in situ.

The first stage in evaluating whether remedial action is indicated is to investigate the radiological implications of taking action. From the discussion in Section 6.4, it is clear that no overall action is indicated on radiological protection grounds when considering the normal gradual releases from the fuel. The possibility of an elevated release of radionuclides from the fuel due to loss of containment barriers has also been considered (Scenario B, Table XXX). This scenario covers the possibility of damage to the waste form due to, e.g. a munitions accident in the vicinity. The calculated annual

TABLE XXIX. SUMMARY OF CRITICAL GROUP DOSES — SCENARIO A, BEST ESTIMATE

Population group	1			2	3
Location	Yamal (Sv)	Taymyr (Sv)	Ob/Yenisey (Sv)	Military (Sv)	Kola (Sv)
Novaya Zemlya Trough	$2 \times 10^{-11} - 2 \times 10^{-10}$	$5 \times 10^{-11} - 9 \times 10^{-9}$	$2 \times 10^{-12} - 5 \times 10^{-10}$	1×10^{-14} [a]	$2 \times 10^{-12} - 3 \times 10^{-10}$
Techeniye Fjord[b]	$2 \times 10^{-16} - 2 \times 10^{-11}$	$2 \times 10^{-16} - 4 \times 10^{-13}$	$2 \times 10^{-16} - 1 \times 10^{-11}$	8×10^{-12} [a]	$5 \times 10^{-19} - 7 \times 10^{-17}$
Tsivolka Fjord	$2 \times 10^{-10} - 3 \times 10^{-8}$	$3 \times 10^{-10} - 7 \times 10^{-8}$	$3 \times 10^{-10} - 5 \times 10^{-9}$	$2 \times 10^{-8} - 6 \times 10^{-4}$	$7 \times 10^{-12} - 6 \times 10^{-10}$
Stepovoy Fjord	$1 \times 10^{-12} - 3 \times 10^{-12}$	$5 \times 10^{-14} - 6 \times 10^{-10}$	$1 \times 10^{-13} - 4 \times 10^{-11}$	8×10^{-12} [a]	$7 \times 10^{-14} - 3 \times 10^{-13}$
Abrosimov Fjord	$2 \times 10^{-10} - 6 \times 10^{-8}$	$2 \times 10^{-10} - 1 \times 10^{-8}$	$2 \times 10^{-10} - 3 \times 10^{-9}$	$2 \times 10^{-9} - 7 \times 10^{-5}$	$5 \times 10^{-11} - 2 \times 10^{-9}$
Totals (sum of maximum results for each source)	9×10^{-8}	9×10^{-8}	8.5×10^{-9}	6.7×10^{-4}	2.9×10^{-9}
Totals (running models using all sources)	$3 \times 10^{-10} - 6 \times 10^{-8}$	$3 \times 10^{-10} - 8 \times 10^{-9}$	$3 \times 10^{-10} - 3 \times 10^{-9}$	$3 \times 10^{-8} - 2 \times 10^{-5}$	$5 \times 10^{-12} - 2 \times 10^{-9}$

[a] Only one set of results.

[b] It should be noted that dose rates of the order of magnitude 10^{16}–10^{19} correspond to only a few radioactive decays in a human lifetime and are thus radiologically meaningless. For comprehensiveness, the calculated figures are, however, included in the report.

TABLE XXX. SUMMARY OF CRITICAL GROUP DOSES AS A RESULT OF RELEASE FROM ICEBREAKER FUEL CONTAINER IN 2050 — SCENARIO B

Population group	1			2	3
Location	Yamal (Sv)	Taymyr (Sv)	Ob/Yenisey (Sv)	Military (Sv)	Kola (Sv)
Tsivolka Fjord only[a]	$3 \times 10^{-10} - 7 \times 10^{-7}$	$3 \times 10^{-10} - 3 \times 10^{-7}$	3×10^{-10} [b]	$1 \times 10^{-7} - 4 \times 10^{-3}$	$1 \times 10^{-10} - 2 \times 10^{-8}$
All other sources[a] according to Scenario A and Tsivolka Fjord Scenario B	$1 \times 10^{-7} - 3 \times 10^{-7}$	$4 \times 10^{-8} - 1 \times 10^{-7}$	1×10^{-7} [b]	$1 \times 10^{-5} - 4 \times 10^{-4}$	$4 \times 10^{-10} - 3 \times 10^{-9}$

[a] In some cases the range of results for ‘all sources’ is lower than that for ‘Tsivolka Fjord only’ because all the models were not used in both cases.

[b] Only one set of results.

TABLE XXXI. SUMMARY OF CRITICAL GROUP DOSES — SCENARIO C

Population group		1		2	3
Location	Yamal (Sv)	Taymyr (Sv)	Ob/Yenisey (Sv)	Military (Sv)	Kola (Sv)
Novaya Zemlya Trough	$8 \times 10^{-11} - 4 \times 10^{-9}$	$2 \times 10^{-11} - 4 \times 10^{-9}$	8×10^{-11} [a]	$1 \times 10^{-14} - 2 \times 10^{-6}$	$3 \times 10^{-12} - 8 \times 10^{-11}$
Techeniye Fjord	$1 \times 10^{-16} - 3 \times 10^{-11}$	$1 \times 10^{-16} - 3 \times 10^{-11}$	1×10^{-16} [a]	$1 \times 10^{-12} - 3 \times 10^{-12}$	$4 \times 10^{-17} - 6 \times 10^{-13}$
Tsivolka Fjord	$2 \times 10^{-9} - 2 \times 10^{-7}$	$1 \times 10^{-9} - 2 \times 10^{-7}$	2×10^{-9} [a]	$1 \times 10^{-7} - 3 \times 10^{-3}$	$2 \times 10^{-10} - 5 \times 10^{-9}$
Stepovoy Fjord	$2 \times 10^{-9} - 3 \times 10^{-8}$	$1 \times 10^{-10} - 3 \times 10^{-8}$	2×10^{-9} [a]	$5 \times 10^{-8} - 7 \times 10^{-8}$	$3 \times 10^{-11} - 6 \times 10^{-10}$
Abrosimov Fjord	$2 \times 10^{-10} - 2 \times 10^{-8}$	$2 \times 10^{-10} - 2 \times 10^{-8}$	2×10^{-10} [a]	$5 \times 10^{-9} - 2 \times 10^{-4}$	$2 \times 10^{-11} - 5 \times 10^{-10}$
Totals (sum of maximum results for each source)	2.5×10^{-7}	2.5×10^{-7}	4.3×10^{-9}	3.2×10^{-3}	6.2×10^{-9}
Totals (running models using all sources)	$4 \times 10^{-9} - 3 \times 10^{-7}$	$4 \times 10^{-9} - 3 \times 10^{-7}$	$4 \times 10^{-9} - 3 \times 10^{-7}$	$8 \times 10^{-10} - 2 \times 10^{-4}$	$6 \times 10^{-10} - 6 \times 10^{-9}$

[a] Only one set of results.

doses to members of the public are slightly higher, at less than 1 μSv, than for Scenario A, but are still very small compared with those experienced by individuals from natural sources. Calculated doses to the critical group or military personnel patrolling the banks of Tsivolka Fjord are also higher, at up to 4 mSv/a, than for Scenario A. However, the dose is not dissimilar to general natural background levels. Furthermore, as noted in Section7.2.2.1, this critical group is a hypothetical one and the assumptions concerning beach occupancy are conservative.

Turning to collective doses, calculations were not undertaken for each source separately, but from an inspection of the radionuclide inventories it can be concluded that releases from the icebreaker fuel box will make only a relatively small contribution to the total collective dose from the wastes dumped in the Kara Sea. Specifically, the icebreaker fuel contributes less than 20% to the total ^{14}C in wastes dumped in the Kara Sea. Therefore, the collective dose to the year 3000 from the dumped icebreaker fuel will be less than 2 man·Sv, and the collective dose over all time will be about five times higher.

7.4. THE NEED FOR REMEDIATION

There are a number of factors that require consideration in a decision concerning the need for remedial actions. These factors are outlined in Section 2.4. From a radiological protection perspective, the most important are the health risks to individuals and populations if no remediation is undertaken.

Turning to the health risks to individuals, remediation should be undertaken if the health risks from the attributable radiation doses are unacceptable. It may be indicated at lower doses but is obviously not indicated if the doses and risks are trivial. In this regard, the doses to individuals in the population groups bordering the Kara and Barents Seas from all the scenarios considered are less than 1 μSv. This dose level is at least an order of magnitude below that which is considered trivial for regulatory purposes [9, 116]. The risk of fatal cancer from a dose of 1 μSv is about 5×10^{-8} a risk level also around an order of magnitude lower than risk levels commonly regarded as trivial [116]. Thus, members of indigenous population groups are not being exposed to unacceptable risks from the dumped wastes; the risks they are being exposed to can be considered trivial.

The doses to the hypothetical critical group of military personnel on Novaya Zemlya, 3.3 mSv/a, are higher than those to members of the public but are similar to average doses from natural background radiation. In evaluating the significance of such exposures within an intervention framework (see Section 2), it is usual to derive an action level — a dose level above which remedial action should be taken. Action levels have not been derived for this particular type of situation. However, for comparison, the dose level implied by the internationally recommended action level for ^{222}Rn in homes [117] is 3–10 mSv/a.

With account taken of the fact that the doses to the hypothetical military personnel could be controlled if required, none of the calculated individual doses indicates a need for remedial action. This conclusion is supported by the comparisons made in earlier sections between the calculated individual doses and doses from other sources. It is based on the summed individual doses from all dumped materials, and therefore the conclusion applies to any one of them.

The second important factor that requires consideration is the health risk to populations. This is considered to be proportional to the collective dose from the dumped materials. In considering collective doses in a remediation decision, it is the collective dose *saved* by the remediation that is important. In simple terms, although the risks to each individual may be trivial when summed over a population, a significant number of health effects may be predicted to arise as a result of the additional exposure. If the costs of the proposed remedial actions are considered worth while when set against the health effects averted, then the remedial action may be indicated. In reality, the decision process will be more complex than this, and many other factors will have to be taken into account (see Section 2).

Taking collective dose into account in a remediation decision is often not straightforward, and problems arise because the collective doses from long lived radionuclides can be delivered over long times into the future. There are increasing uncertainties in calculating collective doses over such time periods. These uncertainties arise from our unavoidable lack of knowledge of the habits, size and location of future populations, together with possible future climate changes. All of these will have an influence on the collective doses. Furthermore, when using collective dose as an indicator of possible health effects over long times into the future, the unpredictable changes in environmental, social and medical conditions will have a profound effect on the relationship between a radiation dose and its corresponding health effects and introduce further uncertainties. Such considerations have led to a proposal that “in decision-making, less significance should be attached to collective doses relating to periods beyond 500 years into the future than to those relating to shorter time periods” [118]. A similar approach has been followed in the EC’s project Marina (see Section 7.2.2.2), where collective doses up to the year 2500 were calculated [94]. In the present study, emphasis is placed on collective doses delivered up to the year 3000. This is twice the time period considered in the previous study but, nevertheless, avoids the possibility of basing a remediation decision on estimates of collective dose of doubtful validity. Since the peak individual doses will occur in some cases only in 2500–2700, the chosen approach seems to be justified.

The collective dose to the world's population up to the year 3000 from all materials dumped in the Kara Sea is about 10 man·Sv. This calculated collective dose is small but, nevertheless, should be considered further in a decision concerning possible remediation. One approach to considering collective dose within a decision framework for deciding on the need for remediation is to assign a monetary value to the health detriment saved by the remedial measures. Monetary values for unit collective dose have been proposed in the range of US $10 000 to 100 000 per man·Sv [119, 120]. Using this range, the calculated collective dose corresponds to approximately US $100 000 to US $1 000 000. These figures provide a yardstick for evaluating possible remedial measures. Measures costing more than US $1 000 000 may not represent a radiologically optimum solution even if they avert all of the collective dose. If the collective dose over all time is considered to be important, the range of monetary value becomes US $1 million to 10 million. Furthermore, there are many other factors such as occupational dose levels and risks during remediation which should be taken into account in the decision making process (see Section 2.4).

Remediation of the icebreaker spent fuel container is the specific example considered in the technical analysis in Section 6. The collective dose up to the year 3000 from this package is about 2 man·Sv, which corresponds to a range of US $20 000 to US $200 000. A simple cost–benefit analysis of the type described in the previous paragraph strongly indicates that the costs of the remedial measures (Section 6) which are in the range of US $5–13 million are unlikely to be warranted by the savings in collective dose. This conclusion would be the same even if collective doses delivered beyond the year 3000 are considered.

In evaluating the relevance of calculated collective doses in a remediation decision, it is important to consider the collective dose saved by the remediation. In this respect, it is important to note that the greater part of the collective dose from the dumped waste arises from the long lived, mobile radionuclide ^{14}C. This radionuclide is not easily contained in a land based disposal facility, although its eventual migration to the biosphere may be delayed.

Therefore, a consideration of collective doses indicates that remedial measures are not warranted.

Section 5 showed that the predicted dose rates to wild organisms are very low, with peak dose rates at about 0.1 μGy/h — a dose rate which is considered unlikely to entail any detrimental effects on morbidity, mortality, fertility, fecundity and mutation rate that may influence the maintenance of a healthy population.

Overall, from the radiological protection viewpoint, remedial action on the radioactive materials dumped in the Kara Sea is not warranted. However, this conclusion depends upon the maintenance of some form of institutional control over Novaya Zemlya and the surrounding waters. This control is largely required to prevent the possibility of inadvertent disturbance or inadvertent retrieval of some part of the dumped waste. Repopulation of the island could also result in higher doses than have been considered in this analysis. The assumption of institutional control is not unreasonable as the island is a former nuclear weapons test site; institutional control may be required for reasons other than the dumped materials.

8. CONCLUSIONS AND RECOMMENDATIONS

8.1. CONCLUSIONS

The following conclusions can be drawn from this study:

(1) Monitoring has shown that releases from identified dumped objects are small and localized to the immediate vicinity of the dumping sites. Overall, the levels of artificial radionuclides in the Kara and Barents Seas are low and the associated radiation doses are negligible when compared with those from natural sources.
Environmental measurements suggest that current annual individual doses from all artificial radionuclides in the Barents and Kara Seas are at most 1–20 µSv. The main contributors are global fallout from nuclear weapons testing, discharges from nuclear fuel reprocessing plants in western Europe and fallout from the Chernobyl nuclear accident. These doses can be compared with worldwide average annual individual dose from natural radiation of 2400 µSv.

(2) Projected future doses to members of the public in typical local population groups arising from radioactive wastes dumped in the Kara Sea are very small. Projected future doses to a hypothetical group of military personnel patrolling the foreshores of the fjords in which wastes have been dumped are higher and comparable in magnitude to doses from natural sources.
These conclusions are drawn from a consideration of the high level solid waste which contains the vast majority of the dumped radionuclides. The radionuclide inventories of dumped waste objects were estimated on the basis of the design and operating histories of the nuclear reactors from which they were derived. The predicted future rates of radionuclide release to the environment from these sources were combined with mathematical models of radionuclide behaviour to calculate radiation doses to humans.
The predicted future maximum annual doses to typical local population groups are less than 1 µSv while those to the hypothetical group of military personnel are higher, up to 4 mSv but still of the same order as the average natural background dose.

(3) Doses to marine organisms are insignificant in the context of effects on populations.
These doses are delivered to only a small proportion of the population and, furthermore, are orders of magnitude below those at which detrimental effects on populations of marine organisms might be expected to occur.

(4) It is concluded that, on radiological grounds, remediation is not warranted. Controls on the occupation of beaches and the use of coastal marine resources and amenities in the fjords of Novaya Zemlya must, however, be maintained.
The condition is specified to take account of concerns regarding the possible inadvertent disturbance or recovery of high level waste objects and the radiological protection of the hypothetical group of individuals occupying the beaches adjacent to the fjords of Novaya Zemlya in which dumping has taken place. Efforts should be made to locate and identify all the high level waste objects whose locations are at present not known.

8.2. RECOMMENDATIONS

It is recommended that:

(1) Efforts should be made to locate and identify all high level waste objects.

(2) Institutional control should be maintained over access and activities in the terrestrial and marine environments in and around the fjords of Novaya Zemlya in which dumping has occurred.

(3) If, at some time in the future, it is proposed to terminate institutional control over areas in and around these fjords, prior assessment should be made of doses to any new groups of individuals who may be potentially at risk.

(4) In order to detect any changes in the condition of the dumped high level wastes, a limited environmental monitoring programme at the dump sites should be considered.

REFERENCES

[1] INTERNATIONAL ATOMIC ENERGY AGENCY, Inventory of Radioactive Material Entering the Marine Environment: Sea Disposal of Radioactive Waste, IAEA-TECDOC-588, Vienna (1991).

[2] INTERNATIONAL ATOMIC ENERGY AGENCY, Provisional Definition of High Level Radioactive Waste Unsuitable for Dumping at Sea and Recommendations, IAEA-INFCIRC/205/add.1, IAEA (1974).

[3] INTERNATIONAL ATOMIC ENERGY AGENCY, Revised Version of the Definition and Recommendations for the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter IAEA-INFCIRC/205/add.1/rev.1, IAEA (1978).

[4] INTERNATIONAL ATOMIC ENERGY AGENCY, Definition and Recommendations for the Convention on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, 1972, 1986 Edition, Safety Series No. 78, IAEA, Vienna (1986).

[5] Facts and Problems Related to Radioactive Waste Disposal in the Seas Adjacent to the Territory of the Russian Federation, Materials for a Report by the Governmental Commission on Matters Related to Radioactive Waste Disposal at Sea, Established by Decree No. 613 of the Russian Federation President, Moscow, 24 October 1992 (1993) (Original in Russian, English translation by International Maritime Organization, LC16/INF.2 (1993)).

[6] Dumping of Radioactive Waste and Investigation of Radioactive Contamination in the Kara Sea, Results from Three Years of Investigations (1992–1994) in the Kara Sea, Joint Norwegian–Russian Expert Group for Investigation of Radioactive Contamination in the Northern Areas, Norwegian Radiation Protection Authority, Oslo (1996).

[7] A Survey of Artificial Radionuclides in the Kara Sea, Results from the Russian–Norwegian 1992 Expedition to the Barents and Kara Seas, Joint Russian–Norwegian Expert Group for Investigation of Radioactive Contamination in the Northern Seas, Norwegian Radiation Protection Authority, Oslo (1993).

[8] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION (ICRP), 1990 Recommendations of the International Commission on Radiological Protection, Publication No. 60, Annals of the ICRP, Pergamon Press, Oxford and New York (1990).

[9] FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, INTERNATIONAL ATOMIC ENERGY AGENCY, INTERNATIONAL LABOUR ORGANISATION, OECD NUCLEAR ENERGY AGENCY, PAN AMERICAN HEALTH ORGANIZATION, WORLD HEALTH ORGANIZATION, International Basic Safety Standards for Protection against Ionizing Radiation and for the Safety of Radiation Sources, Safety Series No.115, IAEA, Vienna (1996).

[10] FRENCH, S., Decision Theory: An Introduction to the Mathematics of Rationality, Ellis Horwood, Chichester (1988).

[11] GOODWIN, P., WRIGHT, G., Decision Analysis for Management Judgement, Wiley, Chichester (1992).

[12] VON WINTERFELDT, D., EDWARDS, W., Decision Analysis and Behavioral Research, Cambridge University Press, Cambridge (1986).

[13] OSTENSO, N.A., “Arctic Ocean”, in The Encyclopedia of Oceanography: Encyclopedia of Earth Sciences, Vol. 1 (FAIRBRIDGE, R.W., Ed.), Reinhold, New York (1966) 49–55.

[14] SCHLOSSER, P., SWIFT, J.H., LEWIS, D., PFIRMAN, S.L., The role of the large-scale Arctic Ocean circulation in the transport of contaminants, Deep-Sea Res. II, **42** (1995) 1341–1367.

[15] PAVLOV, V.K., Oceanographical description of the Kara and Barents Seas, IAEA-IASAP Working Material No. 2, Arctic and Antarctic Research Institute, St. Petersburg (1994).

[16] AAGAARD, K., CARMACK, E.C., The Role of Sea Ice and Other Fresh Water in the Arctic Circulation, J. Geophys. Res. **94** (1989) 14 485–14 498.

[17] INTERNATIONAL ATOMIC ENERGY AGENCY, Modelling of the Radiological Impact of Radioactive Waste Dumping in the Arctic Seas — Report of the Modelling and Assessment Working Group of the International Arctic Seas Assessment Project (in preparation).

[18] SAWYER HOPKINS, T., The GIN Sea. A synthesis of its physical oceanography and literature review (1972–1985), Earth-Sci. Rev. **30** (1991) 175–318.

[19] MAXIMOV, I.W., On summer water flow from the Barents to the Kara Sea through the Kara Gate, Arctic Problems **4** 21 (1940) (in Russian).

[20] IVANOV, V., Sedimentological description of the Barents and Kara Seas, IAEA-IASAP Working Material No. 3, All-Russian Research Institute for Geology and Mineral Resources of the World Oceans, St. Petersburg (1994).

[21] PAVLOV, V.K., PFIRMAN, S.L., Hydrographic structure and variability of the Kara Sea: Implications for pollutant distribution, Deep-Sea Res. II **42** (1995) 1369–1390.

[22] PFIRMAN S.L., EICKEN, H., BAUCH, D., WEEKS, W.F., The potential transport of pollutants by Arctic sea ice, Sci. Total Environ. **159** (1995) 129–146.

[23] FOYN, L., SEMENOV, A., The Joint Norwegian/Russian Expedition to the Kara Sea, Joint Russian–Norwegian Expert Group (cruise report) Moscow–Oslo (1992).

[24] FOYN, L., NIKITIN, A., The Joint Norwegian/Russian Expedition to the Dump Sites for Radioactive Waste in the open Kara Sea, the Tsivolki Fjord and the Stepovogo Fjord, September–October 1993 (cruise report) Bergen (1993).

[25] FOYN, L., NIKITIN, A., Joint Norwegian/Russian Expedition to the Dump Sites for Radioactive Waste in the Abrosimov Fjord and Stepovogo Fjord, August–September 1994 (cruise report) Bergen (1994).

[26] JOINT NORWEGIAN–RUSSIAN EXPERT GROUP, Radioactive Contamination at Dumping Sites for Nuclear Waste in the Kara Sea, Results from the Russian– Norwegian 1993 expedition to the Kara Sea, NRPA, Østerås (1994).

[27] POVINEC, P.P., OSVATH, I., BAXTER, M.S., HARMS, I., "*In-situ* monitoring of gamma radioactivity in the Kara Sea" (Proc. Workshop Monitoring of Nuclear Contamination in Arctic Seas, Washington 1995), NRL, Washington (1995) 2–17.

[28] STRICKLAND, J.D.H., "Production of organic matter in the primary stages of the marine food chain", Chemical Oceanography, Vol. 1 (RILEY J.P., SKIRROW, G., Eds), Academic Press, London (1965) 477–610.

[29] JONES, E.P., NELSON, D.M., TREGUER, P., "Chemical oceanography", Polar Oceanography (SMITH, W.O. Jr., Ed.), Academic Press, London (1990) 407–476.

[30] WALLACE, D.W.R., JONES, E.P., MOORE, R.M., Ventilation of the Arctic Ocean cold halocline: Rates of diapycnal and isopycnal transport, oxygen utilization and primary production inferred using chlorofluoromethane distributions, Deep-Sea Res. **34** (1987) 1957–1979.

[31] SMITH, W.O. Jr., SAKSHAUG, E., "Polar Phytoplankton", Polar Oceanography (SMITH, W.O. Jr., Ed.), Academic Press, London (1990) 477–525.

[32] SMITH, W.O. Jr., Phytoplankton dynamics in marginal ice zones, Oceanogr. Mar. Biol. **25** (1987) 11–38.

[33] DIVOKY, G.J., "The pelagic and nearshore of the Alaskan Beaufort Sea: Biomass and trophics", The Alaskan Beaufort Sea: Ecosystems and Environments (BARNES, P.W., SCHELL, D.M., REIMNITZ, E., Eds), Academic Press, Orlando (1984) 417–433.

[34] BRADSTREET, M.S.W., Occurrence, habitat use and behaviour of seabirds, marine mammals, and Arctic cod at the Pond Inlet ice edge, Arctic **35** (1982) 28–40.

[35] AINLEY, D.G., DEMASTER, D.P., "The upper trophic levels in polar marine ecosystems", Polar Oceanography, Part B (SMITH, W.O. Jr., Ed.), Academic Press, London (1990) 599–630.

[36] BRADSTREET, M.S.W., et al., Aspects of the biology of the Arctic cod (*Boreogadus saida*) and its importance in Arctic marine food chains, Can. Tech. Rep. Fisheries and Aquatic Sciences **1491** (1986) 1–193.

[37] STIRLING, I., CLEATOR, H., SMITH, T.G., "Marine mammals", Polynyas in the Canadian Arctic, Occas. Pap. No. 49 (STIRLING, I., CLEATOR, H., Eds), Can. Wildl. Ser., Ottawa (1981) 45–58.

[38] DIVOKY, G.J., "Sea ice as a factor in seabird distribution and ecology in the Beaufort, Chukchi, and Bering Seas", Conservation of Marine Birds of Northern North America, Wildl. Res. Rep. 11 (BARTONEK J.C., NETTLESHIP, D.N., Eds), U.S. Fish Wildl. Ser., Washington DC (1979) 9–18.

[39] LOWRY, L.F., FROST, K.J., "Feeding and trophic relationships of phocid seals and walruses in the eastern Bering Sea", The Eastern Bering Sea Shelf: Oceanography and Resources, Vol. 2 (HOOD, D.W., CALDER, A.D., Eds), Univ. of Washington Press, Seattle (1981) 813–824.

[40] BRADSTREET, M.S.W., CROSS, W.E., Trophic relationships at high Arctic ice edges, Arctic **35** (1982) 1–12.

[41] BELOPOP'SKII, L.O., Ecology of Sea Colony Birds of the Barents Sea, Israel Program Sci. Transl., Jerusalem (1961).

[42] SAKSHAUG, E., SKJOLDAL, H.R., Life at the ice edge, Ambio **18** (1989) 60–67.

[43] HOPKINS, C.C.E., NILSSEN, E.M., "The rise and fall of the Barents Sea capelin (*Mallotus villosus*): A multivariate scenario" (Proc. Pro Mare Symp. Polar Marine Technology, Trondheim, 1990), (SAKSHAUG, E., HOPKINS, C.C.E., ORITSLAND, N.A., Eds), Polar Res. **2** (1991).

[44] MEHL, S., "The Arctic cod stock's place in the Barents Sea ecosystem in the 1980s: An overview", ibid., Polar Res. **2** (1991).

[45] HAMRE, J., "Interrelation between environmental changes and fluctuating fish populations in the Barents Sea", Long-term Variability of Pelagic Fish Population and Their Environment (KAWASAKI, T., TANAKA, S., TOBA, Y., TANIGUCHI, A., Eds), Pergamon Press, Tokyo (1991) 259–270.

[46] MIQUEL, J.C., Environment and biology of the Kara Sea: A general view for contamination studies, Marine Poll. Bull. (in press).

[47] KRINITSYN, V.S., Peculiarities of biology and distribution of commercial fish in the Yenisey Gulf, GOSNIORH: Collected Scientific Papers **296** (in Russian) (1989) 130–141.

[48] POVINEC, P.P., et al., Global Marine Radioactivity Database (GLOMARD) — Radioactivity of the Arctic Seas, Rep. R4/96, IAEA-MEL, Monaco (1996).

[49] VAKULOVSKY, S.M., Radioactive Contamination of the Arctic Seas (TYPHOON Report, Obninsk, Russian Federation) (1993).

[50] KERSHAW P.J., BAXTER, A.J., "Sellafield as a source of radioactivity to the Barents Sea" (Proc. Int. Conf. Environmental Radioactivity in the Arctic and Antarctic, Kirkenes, 1993), 161–176.

[51] KERSHAW, P.J., GURBUTT, P.A., WOODHEAD, D.S., LEONARD, K.S., REES, J.M., "Estimates of fluxes of caesium-137 in northern waters from recent measurements", ibid., pp. 95–97.

[52] NAVAL OCEANOGRAPHIC OFFICE, Arctic Environmental Isotope Study, Informal Report No. 69–4,OO, Washington, DC (1969).

[53] BAXTER, M.S., et al., "Non-local radiological consequences of nuclear waste dumping in the Arctic Seas: a preliminary assessment" (Proc. Conf. Radioactivity and Environmental Security in the Oceans, Woods Hole, 1993) 329–341.

[54] STRAND, P., et al., "Survey of artificial radionuclides in the Kara Sea" (Proc. Int. Conf. Environmental Radioactivity in the Arctic and Antarctic, Kirkenes, 1993) 53–65.

[55] HAMILTON, T.H., et al., Radiometric investigations of Kara Sea sediments and preliminary radiological assessment related to dumping of radioactive wastes in the Arctic Seas, J. Environ. Radioact. **25** (1994) 113–134.

[56] STRAND, P., et al., Survey of artificial radionuclides in the Barents Sea and the Kara Sea, J. Environ. Radioact. **25** (1994) 99–112.

[57] BAXTER, M.S., et al., "Marine radioactivity studies in the vicinity of sites with potential radionuclide releases" (Proc. Int. Symp. Environmental Impact of Radioactive Releases, Vienna 1995) 125–141.

[58] NIKITIN, A.I., et al., "Joint Russian–Norwegian collaboration on radioactive contamination from dumped nuclear wastes in the Kara Sea — 3 years with expeditions to the dumping sites: goals and investigation" (Proc. Int. Conf. Environmental Radioactivity in the Arctic, Oslo) (1995) 125–128.

[59] OSVATH, I., et al., "IAEA-MEL's contribution to the investigation of the Kara Sea dumping sites", ibid., pp. 139–144.

[60] SALBU, B., et al., Joint Russian–Norwegian collaboration on radioactive contamination from dumped nuclear wastes in the Kara Sea — Results from the 1994 expedition to Abrosimov and Stepovogo Fjords" (Proc. Int. Conf. Environmental Radioactivity in the Arctic, Oslo) (1995) 129–134.

[61] AARKROG, A., Radioactivity in polar regions — Main sources, J. Environ. Radioact. **25** (1994) 21–35.

[62] INTERNATIONAL ATOMIC ENERGY AGENCY, Marine Environment Laboratory Monaco, unpublished results (1996).

[63] POVINEC, P.P., OSVATH, I., BAXTER, M.S., Underwater gamma spectrometry with HPGe and NaI(Tl) detectors, Appl. Radiat. Isot. **47** (1996) 1127–1133.

[64] UNITED NATIONS SCIENTIFIC COMMITTEE ON THE EFFECTS OF ATOMIC RADIATION, Ionizing Radiation: Sources and Biological Effects, UN, New York (1993).

[65] MATISHOV, G.G., MATISHOV, D.G., SZCZYPA, J., "Radionuclides in the bottom sediments and biota of the shelf and Barents Sea coasts" (Proc. Int. Conf. Environmental Radioactivity in the Arctic and Antarctic, Kirkenes 1993) 305–326.

[66] SMITH, J.N., et al., "Radionuclide sources in the Barents and Kara Seas" (Proc. Int. Conf. Environmental Radioactivity in the Arctic, Oslo 1995) 179–185.

[67] POVINEC P.P., et al., IAEA-MEL's contribution to the investigation of Kara Sea radioactivity and radiological assessment, Marine Poll. Bull. (in press).

[68] MATISHOV, D.G., MATISHOV, G.G., RISSANEN, K., "Peculiarities of radionuclides' accumulation in benthic organisms and fish of the Barents and Kara Seas" (Proc. Int. Conf. Environmental Radioactivity in the Arctic, Oslo 1995) 233–237.

[69] BAXTER M.S., et al., "IAEA programmes related to the radioactive waste dumped in the Arctic Seas: Part 2. Arctic Seas Assessment Project at the IAEA-MEL" (Proc. Int. Conf. Environmental Radioactivity in the Arctic and Antarctic, Kirkenes 1993) 93–96.

[70] BAXTER, M.S., FOWLER, S.W., POVINEC, P.P., Observations on plutonium in the oceans, Appl. Radiat. Isot. **46** 11 (1995) 1213–1223.

[71] FOWLER, S.W., et al., "International radiological assessment programs related to radioactive waste dumping in the Arctic Seas: IAEA-MEL's role" (Proc. Workshop Arctic Contamination, Anchorage 1993) 92–100.

[72] INTERNATIONAL ATOMIC ENERGY AGENCY, Sediment K_ds and Concentration Factors for Radionuclides in the Marine Environment, Technical Reports Series No. 247, IAEA, Vienna (1985).

[73] INTERNATIONAL ATOMIC ENERGY AGENCY, Environmental and Radiological Description of the Arctic Ocean — Report for the International Arctic Seas Assessment Project (in preparation).

[74] SIVINTSEV, Y.V., Study of Nuclides Composition and Characteristics of Fuel in Dumped Submarine Reactors and Atomic Icebreaker "LENIN": Part 1 – Atomic Icebreaker, IAEA–IASAP Rep. No. 1, Russian Research Center "Kurchatov Institute", Moscow, Russian Federation (1993).

[75] SIVINTSEV, Y.V., Study of Nuclides Composition and Characteristics of Fuel in Dumped Submarine Reactors and Atomic Icebreaker "LENIN": Part 2 — Nuclear Submarines, IAEA-IASAP Rep. No. 5, Russian Research Center "Kurchatov Institute", Moscow, Russian Federation (1994).

[76] YEFIMOV, E., Radionuclides Composition, Characteristics of Shielding Barriers, and Analyses of Weak Points of the Dumped Reactors of Submarine N 601, IAEA–IASAP Rep. No. 6, State Scientific Center of Russian Federation, Institute of Physics and Power Engineering, Obninsk, Russian Federation (1994).

[77] SIVINTSEV, Y.V., Description of Shielding Barriers at the Nuclear Reactors Dumped in the Arctic, IAEA–IASAP Rep. No. 8, Russian Research Center "Kurchatov Institute", Moscow, Russian Federation (1995).

[78] INTERNATIONAL ATOMIC ENERGY AGENCY, Predicted Radionuclide Release from Marine Reactors

Dumped in the Kara Sea, Report of the Source Term Working Group of the International Arctic Seas Assessment Project (IASAP), IAEA-TECDOC-938, Vienna (1997).

[79] RUBTSOV, P.M., RUZHANSKII, P.A., Estimate of the radiation characteristics of spent fuel from submarine and 'Lenin' icebreaker reactors scuttled in the region of the Archipelago Novalya Zemlya, At. Energy, **81** 3 (in English) (1997) 656–662.

[80] WARDEN, J., et al., Potential radionuclide release rates from marine reactors dumped in the Kara Sea, Sci. Total Environ. **202** (1997) 225–236.

[81] HEISER, J.H., SOO, P., Corrosion of Barrier Materials in Sea Water Environment, Brookhaven National Laboratory, Upton, NY, USA (1995).

[82] BARTH, C.H., SHELDON, R.B., Corrosion Rates of Structural Materials on the Ocean Floor, Rep. KAPL-4701, General Electric Co., Schenectady, NY, USA (1989).

[83] SOO, P., Memorandum on Review of Corrosion Rates Specified by the International Arctic Seas Assessment Project (IASAP), Brookhaven National Laboratory, Upton, NY, USA (1995).

[84] CARTER, J.H., "Corrosion of LENIN Reactor Compartment Materials", Department of Nuclear Science and Technology, Royal Naval College, Greenwich, London, Technical Memorandum RNC/NS/TM431 (1994).

[85] ALEXANDROV, V.P., SIVINTSEV, Y.V., Composition and Properties of Compound Furfurol(F), Russian Research Center "Kurchatov Institute", Rep. 31/7522, Moscow, Russian Federation (1995).

[86] ALEXANDROV, V.P., SIVINTSEV, Y.V., Application of the Compound "Furfurol" to the Isolation of Spent Nuclear Fuel, Russian Research Center "Kurchatov Institute", Rep. 31/7705, Moscow, Russian Federation (1996).

[87] US ENERGY RESEARCH AND DEVELOPMENT ADMINISTRATION, Operational Accidents and Radiation Exposure Experience within the U.S. AEC, Rep. WASH-1192 Dept. of Energy (1975).

[88] HIBLER, W.D. III, A Dynamic Thermodynamic Sea Ice Model, J. Phys. Oceanogr. **9** (1979) 815–864.

[89] HIBLER, W.D. III, Modeling a Variable Thickness Sea Ice Cover, Monthly Weather Review **198** (1980) 1942–1973.

[90] CIGNA, A., DELFANTI, R., SERRO. E., (Eds), The radiological exposure of the population of the European Community to radioactivity in the Mediterranean Sea, Marina Med. Project (Proc. Seminar, Rome), Rep. EUR-15564 (1994).

[91] INTERNATIONAL ATOMIC ENERGY AGENCY, Sources of Radioactivity in the Marine Environment and their Relative Contributions to Overall Dose Assessment from Marine Radioactivity (MARDOS), IAEA-TECDOC-838, Vienna (1995).

[92] FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS, Fishery Statistics **70–77**, FAO, Rome (1992).

[93] INTERNATIONAL COUNCIL FOR EXPLORATION OF THE SEA, Bull. Stat. Peches Mar. **70-73**, ICES, Copenhagen (1988–1992).

[94] COMMISSION OF THE EUROPEAN COMMUNITIES, The Radiological Exposure of the Population of the European Community from Radioactivity in North European Marine Waters, Project "Marina", Rep. EUR-12483-EN, 571P, Luxembourg (1990).

[95] INTERNATIONAL ATOMIC ENERGY AGENCY, CRP on Modelling of the Radiological Impact of Radioactive Waste Dumping in the Arctic Seas, Benchmarking Scenario, Working Material of the International Arctic Seas Assessment Project, IAEA–IASAP-7, Vienna (1994).

[96] BARRACLOUGH, I.M., ROBB, J.D., ROBINSON, C.A., SMITH, K.R., COOPER, J.R., The use of estimates of collective dose to the public, J. Radiol. Prot. **16** 2 (1996) 73–80.

[97] SCOTT, E.M., et al., Benchmarking of numerical models describing the dispersion of radionuclides in the Arctic Seas, Sci. Total Environ. **202** (1997) 123–134.

[98] WOLLENBURG, I., Sediment Transport by Arctic Sea Ice: The Recent Load of Lithogenic and Biogenic Material, Berichte zur Polarforschung, 127, Alfred-Wegener-Institut, Hamburg (1993).

[99] NÜRNBERG, D., et al., Sediments in Arctic Sea ice: Implications for entrainment, transport and release, Marine Geol. **119** (1994).

[100] CHENG, A., PRELLER, R.H., The Development of an Ice–Ocean Coupled Model for the Northern Hemisphere, NRL Formal Report, Rep. NRL/FR/7322–95-9627, Naval Research Lab., Washington (1966).

[101] PFIRMAN, S., LANGE, M.A., WOLLENBURG, I., SCHLOSSER, P., "Sea ice characteristics and the role of sediment inclusions in deep-sea depositions: Arctic – Antarctic, comparisons in geological history of the polar oceans", Arctic versus Antarctic, Dordrecht (BLEIL, U., THIEDE, J., Eds.), Kluwer Academic, Netherlands (1990).

[102] MEESE, D., et al., Caesium-137 Contamination in Sea Ice (Proc. Workshop Monitoring Nuclear Contamination in the Arctic Seas), Naval Research Laboratory, Washington DC (1995).

[103] COLONY, R., THORNDIKE, A.S., Sea ice motion as drunkard's walk, J. Geogr. Res. **90** (1985).

[104] INTERNATIONAL ATOMIC ENERGY AGENCY, Assessing the Impact of Deep Sea Disposal of Low Level Radioactive Waste on Living Marine Resources, Technical Reports Series No. 288, IAEA, Vienna (1988).

[105] INTERNATIONAL ATOMIC ENERGY AGENCY, Methodology for Assessing Impacts of Radioactivity on Aquatic Ecosystems, Technical Reports Series No. 190, IAEA, Vienna (1979).

[106] NATIONAL COUNCIL ON RADIATION PROTECTION AND MEASUREMENTS, Effects of Ionizing

Radiation on Aquatic Organisms, Rep. NCRP 109, Bethesda, MD (1991).

[107] WOODHEAD, D.S., The radiation exposure of black-headed gulls *(Larus ridibundus)* in the Ravenglass Estuary, Cumbria, UK: A preliminary assessment, Sci. Total Environ. **58** (1986) 273–281.

[108] CALMET, D., WOODHEAD, D., ANDRÉ, J.M., ^{210}Pb, ^{137}Cs and ^{40}K in three species of porpoises caught in eastern tropical Pacific Ocean, J. Environ. Radioact. **15** (1992) 153–169.

[109] BROWNELL, G.L., ELLETT, W.H., REDDY, A.R., Absorbed fractions for photon dosimetry, J. Nucl. Med. Suppl. **1** (1968) 27–39.

[110] NUCLEAR ENERGY AGENCY OF THE OECD, Review of the Continued Suitability of the Dumping Site for Radioactive Waste in the North-East Atlantic, OECD/NEA, Paris (1985).

[111] INTERNATIONAL ATOMIC ENERGY AGENCY, Effects of Ionizing Radiation on Plants and Animals at Levels Implied by Current Radiation Protection Standards, Technical Reports Series No. 332, IAEA, Vienna (1992).

[112] BRIESMEISTER, J.F. (Ed.), MCNP — A General Monte Carlo N-Particle Transport Code Version 4A, Rep. LA-12625-M., Los Alamos Scientific Lab., NM, USA (November 1993).

[113] INTERNATIONAL COMMISSION ON RADIATION UNITS AND MEASUREMENTS, Measurement of Dose Equivalents from External Photon and Electron Radiations, Rep. ICRU 47, ICRU, Bethesda, MD, USA (1992).

[114] LYSTSOV, V.N., "Radioecological assessment for solid radioactive waste (SRW) dumped into the Kara Sea by the former USSR" (Proc. Int. Sem. Radioactivity and Environmental Security in the Oceans: New Research and Policy Priorities in the North Atlantic, June, 1993), Woods Hole Oceanographic Institution, Woods Hole, MA, USA (1993).

[115] TITLEY, J.G., CABIANCA, T., LAWSON, G., MOBBS, S.F., SIMMONDS, J.R., Improved global dispersion models for Iodine-129 and Carbon-14, EC, Rep. EUR 15880 EN, Luxembourg (1995).

[116] INTERNATIONAL ATOMIC ENERGY AGENCY, Principles for the Exemption of Radiation Sources and Practices from Regulatory Control, Safety Series No. 89, IAEA, Vienna (1988).

[117] INTERNATIONAL COMMISSION ON RADIOLOGICAL PROTECTION, Protection Against Radon-222 at Home and at Work, Publication No. 65, Annals of the ICRP **23** 2, Pergamon Press, Oxford and New York (1994).

[118] BARRACLOUGH, I.M., ROBB, J.D., ROBINSON, C.A, SMITH, K.R., COOPER, J.R., The use of estimates of collective doses to the public, J. Radiat. Prot. **16** 2 (1996) 73–80.

[119] INTERNATIONAL ATOMIC ENERGY AGENCY, Intervention Criteria in a Nuclear or Radiation Emergency, Safety Series No. 109, IAEA, Vienna (1994).

[120] THE NORDIC RADIATION PROTECTION AUTHORITIES, The Monetary Value of Collective Dose Reduction (α–value), Statement from a Meeting of the Nordic Radiation Protection Authorities at Reykjavik (14 June 1991).

CONTRIBUTORS TO DRAFTING AND REVIEW

Aarkrog, A.	Risø National Laboratory, Denmark
Baxter, M.S.	IAEA Marine Environment Laboratory, Monaco
Bewers, J.M.	Department of Fisheries and Oceans, Canada
Boisson, F.	IAEA Marine Environment Laboratory, Monaco
Carrol, J.-L.	IAEA Marine Environment Laboratory, Monaco
Charmasson, S.	IPSN/DPRE, France
Cigna, A.A.	ENEA, Italy
Cooper, J.	National Radiological Protection Board, United Kingdom
Dyer, R.	US Environmental Protection Agency, United States of America
Fisher, N.	Marine Sciences Research Centre, United States of America
Fowler, S.W.	IAEA Marine Environment Laboratory, Monaco
Føyn, L.	Institute of Marine Research, Norway
Gurbutt, P.	Ministry of Agriculture, Fisheries and Food, United Kingdom
Gussgard, K.	Norwegian Radiation Protection Authority, Norway
Harms, I.	Universität Hamburg, Germany
Heling, R.	KEMA Nederland B.V., Netherlands
Holm, E.	Lund University, Sweden
Johansson, G.	National Institute of Radiation Protection, Sweden
Kinehara, Y.	Research Center for Safety Engineering, Japan
Kuznetsov, Y.	Khlopin Radium Institute, Russian Federation
Lynn, N.	Royal Naval College, United Kingdom
Mork, O.	Mork Marine a.s., Norway
Mount, M.	Lawrence Livermore National Laboratory, United States of America
Nauke, M.	International Maritime Organization, United Kingdom
Newstead, C.	State Department, United States of America
Nielsen, S.P.	Risø National Laboratory, Denmark
Nies, H.	Bundesamt für Seeschiffahrt und Hydrographie, Germany
Nyeffeler, F.	Université de Neuchâtel, Switzerland
Osvath, I.	IAEA Marine Environment Laboratory, Monaco
Povinec, P.P.	IAEA Marine Environment Laboratory, Monaco
Preller, R.	Naval Research Laboratory, United States of America
Rakov, N.A.	Ministry of Atomic Energy of the Russian Federation, Russian Federation
Rissanen, K.	Finnish Centre for Radiation and Nuclear Safety, Finland
Roubtsov, P.M.	The State Institute for Applied Ecology, Russian Federation
Ruksha, V.	Murmansk Shipping Company, Russian Federation
Salbu, B.	Agricultural University of Norway, Norway
Salo, A.	Private Consultant, Finland
Sazykina, T.	SPA “Typhoon”, Russian Federation
Schwarz, J.	Hamburg Ship Model Basin, Germany
Scott, M.	University of Glasgow, United Kingdom
Sivintsev, Y.	Russian Research Center “Kurchatov Institute”, Russian Federation
Sjoeblom, K-L.	International Atomic Energy Agency
Strand, P.	Norwegian Radiation Protection Authority, Norway
Tanner, J.	Pacific Northwest National Laboratory, United States of America

Templeton, W.	Battelle Pacific NW National Laboratory, United States of America
Teyssie, J.-L.	IAEA Marine Environment Laboratory, Monaco
Timms, S.	Royal Naval College, United Kingdom
Wada, A.	Nihon University, Japan
Walenkamp, H.J.G.	Smit International, Netherlands
Warden, J.	Royal Naval College, United Kingdom
Woodhead, D.	Directorate of Fisheries Research, United Kingdom
Yefimov, E.	Institute of Physics and Power Engineering, Russian Federation
Zuur, E.	Limnocéane, Université de Neuchâtel, Switzerland